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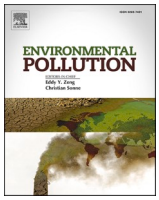

Microplastic shape influences fate in vegetated wetlands☆

Hayley K. McIlwraith [a,b], Penelope K. Lindeque [a], Anastasia Miliou [c], Trevor J. Tolhurst [b], Matthew Cole [a,*]

[a] Marine Ecology & Biodiversity, Plymouth Marine Laboratory, Prospect Place, Plymouth, PL1 3DH, UK
[b] University of East Anglia, School of Environmental Sciences, Norwich Research Park, Norwich, NR4 7TJ, UK
[c] Archipelagos Institute of Marine Conservation, Pythagorio, Samos, 83103, Greece

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ABSTRACT

Coastal areas are prone to plastic accumulation due to their proximity to land based sources. Coastal vegetated habitats (e.g., seagrasses, saltmarshes, mangroves) provide a myriad of ecosystem functions, such as erosion protection, habitat refuge, and carbon storage. The biological and physical factors that underlie these functions may provide an additional benefit: trapping of marine microplastics. While microplastics occurrence in coastal vegetated sediments is well documented, there is conflicting evidence on whether the presence of vegetation enhances microplastics trapping relative to bare sites and the factors that influence microplastic trapping remain understudied. We investigated how vegetation structure and microplastic type influences trapping in a simulated coastal wetland. Through a flume experiment, we measured the efficiency of microplastic trapping in the presence of branched and grassy vegetation and tested an array of microplastics that differ in shape, size, and polymer. We observed that the presence of vegetation did not affect the number of microplastics trapped but did affect location of deposition. Microplastic shape, rather than polymer, was the dominant factor in determining whether microplastics were retained in the sediment or adhered to the vegetation canopy. Across the canopy, microfibre concentrations decreased from the leading edge to the interior which suggests that even on a small-scale, vegetation has a filtering effect. The outcome of this study enriches our understanding of coastal vegetation as a microplastics sink and that differences among microplastics informs where they are most likely to accumulate within a biogenic canopy.

1. Introduction

Plastic debris is found in high concentrations in coastal waters, which include highly valued vegetated ecosystems such as mangroves, saltmarshes, and seagrasses (Harris et al., 2021; Ouyang et al., 2022; Paduani, 2020). Plastic debris is persistent, harmful, and can breakdown into smaller pieces called microplastics (<5 mm in size), contributing to environmental accumulation and harm (Browne et al., 2007; Cole et al., 2011). Coastal wetlands have many ecosystem functions that benefit both human and wildlife communities, including coastal protection, erosion control, habitat refuge, and carbon storage (Barbier, 2013; Barbier et al., 2011; Chmura et al., 2003; Spalding et al., 2014), with vegetation canopies dampening wave action, decreasing turbulence, and promoting sediment deposition (Gacia et al., 1999; Infantes et al., 2012; Möller et al., 2014; Terrados and Duarte, 2000). Similarly, coastal vegetation might play a role in the entrapment of macroplastic and microplastic debris (Waldschläger et al., 2022; Yao et al., 2019). Understanding microplastics distribution and interactions in coastal systems is key for protecting these important areas.

Microplastics can enter coastal vegetative habitats from both riverine and marine sources, or by the degradation of trapped macroplastics (Biltcliff-Ward et al., 2022; Weinstein et al., 2016). A number of studies have suggested such habitats may act as a microplastics sink (e.g., de los Santos et al., 2021; Huang et al., 2020; Kreitsberg et al., 2021; Lloret et al., 2021; Navarrete-Fernández et al., 2022), and several field studies have demonstrated vegetated coastal wetlands trap more microplastics than unvegetated sites (Huang et al., 2023, 2020; Jones et al., 2020; Ogbuagu et al., 2022; Pinheiro et al., 2022; Zhao et al., 2022). Yet the mechanisms by which microplastics become entrapped remain poorly elucidated. These studies postulate that because vegetation reduces local

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* Corresponding author.
E-mail address: mcol@pml.ac.uk (M. Cole).

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turbulence and bed shear stress thereby promoting sediment deposition, the same mechanisms also promote microplastics deposition (Waldschläger et al., 2022). It has also been observed that microplastics can adhere to the vegetation itself (Goss et al., 2018; Jones et al., 2020). However, other field studies have found no difference in microplastic concentration between bare and vegetated sites (Cozzolino et al., 2020; Unsworth et al., 2021; Wright et al., 2023). A meta-analysis by Biltcliff-Ward et al. (2022) found a negligible difference in microplastic concentration between bare and vegetated sites. There are many factors that may influence whether a difference is observed, such as where samples are collected within a study area. For instance, many studies sample from the dense interior of a canopy (Cozzolino et al., 2020; Unsworth et al., 2021; Wu et al., 2020). It is known, however, that sediment and nonmotile fauna typically accumulate on the edge of a canopy (Bologna and Heck, 2002; Leonard et al., 2002). Studies investigating spatial variation of microplastics in coastal vegetation found a similar pattern to sediment deposition – increasing microplastics with proximity to the vegetation edge (Helcoski et al., 2020; Navarrete-Fernández et al., 2022; Yao et al., 2019), although the inverse pattern was observed in one study in Southern Brazil (Pinheiro et al., 2022). Other factors that may influence microplastics accumulation in bare vs. vegetated sites include: nearby population density and proximity to plastic sources; water depth; flow; tidal flux, phase and cycle; bathymetry; sampling and laboratory methodology (Jones et al., 2020; Lloret et al., 2021; Wu et al., 2020).

Given the myriad of drivers that can influence microplastic deposition, some insight can be drawn from well-established sedimentology research (Lofty et al., 2023). There are many additional factors that are known to influence sedimentation that have yet to be tested for microplastics, such as canopy structural complexity, frontal area of vegetation, vegetation species and morphology, local hydrodynamics, and topography (Bouma et al., 2013; Chapman et al., 2015; Chen et al., 2018; Hendriks et al., 2010; Wilkie et al., 2012). Half of all plastic types have a lower density than seawater (Ballent et al., 2013). Due to their low density and slow settling velocity, it is generally assumed that microplastics act most like fine grained sediments (e.g., clay, silt) and organic matter (e.g., leaves, wood, algal debris)(Harris, 2020). However, there is a discrepancy in the literature on whether a relationship exists between microplastic abundance and fine sediments and organic matter (Enders et al., 2019). Particle size, shape, and density will affect particle transport and deposition (Waldschläger and Schüttrumpf, 2019), and microplastics come in diverse shapes and sizes that vary from sediment grains (e.g., fibres). It is for these reasons that microplastic dynamics are likely different to those for natural sediments (Bridge and Bennett, 1992; Horton and Dixon, 2018; Mendrik et al., 2023).

Controlled laboratory flume experiments have started to identify the drivers of microplastic entrapment, these include: water flow, plant density, presence of infauna, microplastic polymer type and size, and bed roughness (Cozzolino et al., 2022; de los Santos et al., 2021; Ogbuagu et al., 2022). These studies used live macrophytes taken directly from the environment, which maintains environmental relevance but makes it difficult to isolate individual drivers. For instance, the physical barrier of the plant structure and the epibiont and biofilm coverage on vegetation blades are both factors that could influence microplastics trapping but are difficult to unpack from each other with living plants (de Smit et al., 2021; Ogbuagu et al., 2022). Many of these studies used large industrial pellets as a proxy for microplastics, which are less commonly found in the environment and may act more similarly to fine grained sediments (de los Santos et al., 2021; Harris, 2020; Ogbuagu et al., 2022). In contrast, fibres and small particles <2 mm are more abundant in the environment and are less likely to act like sediment, and so should be incorporated into more studies (Athey and Erdle, 2022; Harris, 2020).

Using a laboratory flume, we test the hypothesis that the presence and physical structural complexity of vegetation will affect microplastic trapping and that differences in microplastic type will affect their depositional patterns. We predict that more complex vegetation will trap higher loads of microplastics than less complex vegetation. Furthermore, the effect of microplastic shape, size, and polymer type on vegetative interaction and depositional patterns were assessed. The results provide insight on some of the potential variables affecting microplastic trapping in aquatic vegetation canopies and will help to inform potential hotspots of contamination.

2. Methods

2.1. Flume tank

Experiments were conducted using a closed-loop flume system (Fig. 1, Armfield Sediment Transport Channel S8 MKII), comprising a linear test section (1.5 m length, 0.08 m width, 0.11 m height). The pump provided a constant mean flow of 14.7 cm/s, consistent with similar flume set-ups (Cozzolino et al., 2022; de Smit et al., 2021). Flow rates within the flume were measured in triplicate using a flowmeter (Valeport801) at three points in the middle of the water column: upstream, midstream, and downstream (represented by black dots in Fig. 1). To ensure microplastics were kept suspended when transporting through the system, two submersible pumps (Boyu FP-350) were inserted into the influent tank and one in the discharge tank (Boyu FP-100; shown in Fig. 1).

2.2. Vegetation scenarios

To compare the influence of both the absence and presence of vegetation and the complexity of vegetation on microplastic deposition, three treatments were employed: (1) flat sand with no vegetation (control, N = 3); (2) grassy vegetation (N = 3), comprising artificial plants with a 0.2 cm diameter stem and 8 flat blades protruding upwards; and (3) branched vegetation (N = 3), comprising artificial plants with a 0.2 cm diameter stem and 6 branches with 12 round leaves each (Figure S1 and Figure S2). Artificial plants, constructed from polyethylene, were used in lieu of real vegetation to ensure that all plants were a uniform size and shape and to minimise other biological influences on microplastic behaviour. Surface area of plants was calculated by measuring the length and width of each part of the plant. The stems and branches were assumed to be cylinders, grass blades as rectangles, and rounded leaves as circles. Total surface area was the summation of the stem, branches, and both sides of the leaves and blades. Only the submerged part of each plant was included in the measurements. To maintain the stability of the artificial plants, they were inserted into 2 mm pre-drilled holes within an acrylic sheet (0.08 m long x 0.5 m wide x 3 mm thick), placed on the base of the central area of the flume. The holes were drilled in an irregular pattern but were spread across the length of the acrylic sheet (shown in Fig. 1). The sheet was placed in the same orientation for each experimental run and 23 plants were inserted for each flume run to achieve a vegetation density of 575 plants m^{-2} which is a density found within 5 m of a salt marsh edge (Neumeier and Amos, 2006). Acrylic sheets without plant inserts were used during the control runs to account for potential confounds. Prior to each run, 2.4 L of well-sorted sand was added to the flume (mode grain size: 262 μm, measured with a Malvern Mastersizer, 2000), covering the acrylic sheet to a depth of 2 cm. To prevent the accidental introduction of microplastics to the test system, sand was baked at 500 °C for a minimum of 4 h to combust any polymers present.

2.3. Microplastics

To account for the diverse array of microplastics found in environmental samples, the experimental system was spiked with five types of easily identifiable microplastics varying in size, shape, and density (Table 1, Figure S3). Microplastics were dosed at a concentration of 300 particles L^{-1} for each microplastic type, with a total microplastic

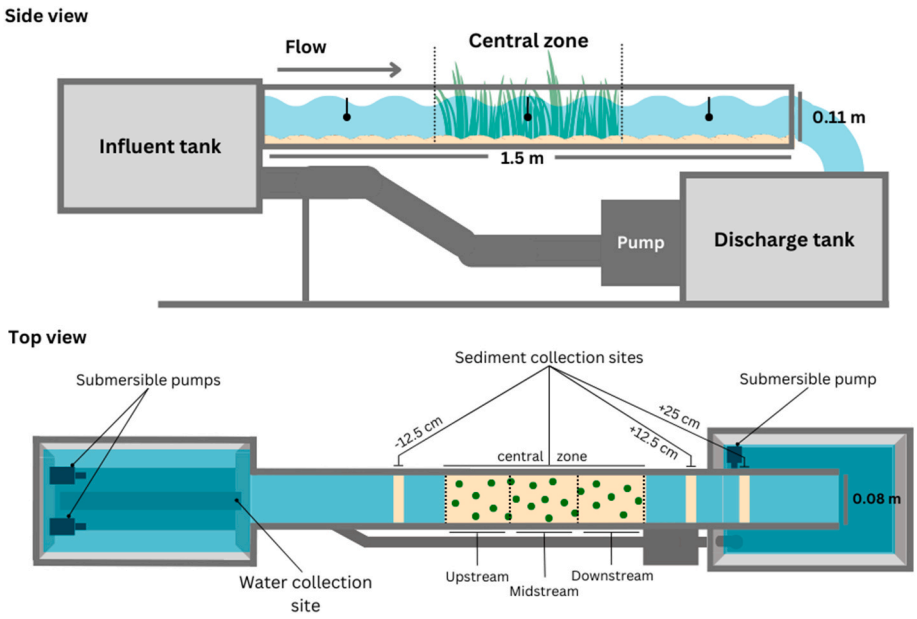

Fig. 1. Schematic of close-looped flume tank set up with artificial vegetation and sand. Flow rate was measured at three points represented by black dots in the upper panel (side view). Green dots in lower panel (top view) represent placement of artificial vegetation. Submersible pumps were used to maintain flow of microplastics through the flume tank. Water was sampled throughout the experiment; sediment and vegetation were sampled at the end to quantify microplastic trapping. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

Table 1
Characteristics of microplastics included in the experiment.

Polymer	Shape	Size (μm)	Density (g/cm^3)	Colour
PVC	Flake	1000	1.38	Purple
PET	Flake	375	1.38	Blue
PET	Fibre	100–1000 (median: 610)	1.38	Red
PA6,6	Fibre	100–1000 (median: 525)	1.14	Purple
PA6,6	Fragment	1000	1.14	White

concentration of 1500 particles L^{-1}. While these concentrations exceed environmental concentrations, they ensured that adequate levels of microplastics could be captured within samples.

Polyethylene terephthalate (PET) and nylon (PA6,6) fibres were created by shaving and cutting fleece-like fabrics of each polymer type with a stainless-steel scalpel and scissors. The shavings were rinsed into a glass jar and vacuum filtered through a 1000 μm nylon mesh filter, and the filtrate vacuum filtered through a 100 μm nylon mesh filter. The 100–1000 μm fibres collected on the mesh filter were rinsed with ultrapure water into a sterile glass jar and stored at room temperature prior to use. Fibre length was ascertained by measuring the length of fibres across five 0.1 mL subsamples of the stock solutions with ImageJ software (version 2.3.0). For PET and nylon (PA6,6) fibres, median length was 610 μm and 525 μm, respectively. Polyvinyl chloride (PVC) and PET flakes and nylon fragments were procured from domestic suppliers, weighed, and added directly to a 300 mL glass jar. Then, aliquots of both fibre types were added to the mixtures.

Pilot tests using microplastics showed particles tended to float, flocculate, and adhere to sides of the flume. Therefore, microplastic mixtures were prepared in advance following a modified protocol from Ramsperger et al. (2020), by incubating them in filtered (2 μm, polycarbonate filter) lake water for 7 days at 30 °C and 100 RPM (VWR Incubating Orbital Mini Shaker) to allow a biofilm to form, though presence of a biofilm was not confirmed. Prior to experimental use, microplastics were filtered out using a 0.45 μm mixed cellulose ester filter, rinsed with ultrapure water, and stored in glass jars with 150 mL of ultrapure water. Following incubation, microplastic readily mixed within the treatment water and flocculation and adherence was visibly reduced.

2.4. Experimental runs

For each run, the flume was filled with 30 L of deionized (DI) water and the system turned on for 10 min prior to adding any plastics to allow the pump to prime. Microplastics were added to the flume by swirling the 300 mL glass jar containing the particles with 150 mL of DI water and releasing the contents of the jar into the influent tank in the direction of the test section. To track waterborne microplastic concentrations throughout the run, 200 mL water samples were taken 5 min after the addition of microplastics (T_5), with subsequent samples collected every 15 min up to 1 h (T_{20}, T_{35}, T_{50}, T_{65}). Water samples were collected by using a glass beaker to take dip-samples from the influent tank (adjacent to the entrance of the flume test section; Fig. 1), which were thrice-rinsed into pre-labelled 300 mL glass jars.

At the end of the run, the pump was turned off and the water was left to drain out for 2 h, simulating the drainage of an intertidal area on an ebb tide. To determine the deposition of microplastics across the test section, sediment samples were collected: (i) upstream of the central zone (−12.5 cm); (ii) within the central zone; and (iii) at two points downstream of the central zone (+12.5 cm, and +25 cm; Fig. 1). Samples were collected by scooping-up small sections of the sand (8 cm × 4 cm for upstream and downstream points; 50 cm × 8 cm for the central zone, with the entire depth collected from the sediment surface to the flume base) with a stainless-steel spoon and placing these samples in pre-labelled 300 mL glass jars. Due to erosion, transport and deposition during the experiment, depth of sand within and across sections varied from 0 cm–5 cm depending on location and treatment run. Due to this variation, sediment across the entire central zone was collected. There was less variation in depth in the upstream and downstream points, so smaller sections were collected.

To ascertain whether microplastics were adhering to the surface of the plants, three plants were sampled. For each run, the vegetation section was divided into three equal zones (17 cm × 8 cm with 7–8 plants per zone), and a single plant was selected from: (i) the upstream; (ii) midstream; and (iii) downstream areas (Fig. 1). Each plant was carefully removed from the acrylic base and rinsed thoroughly with DI

water over a pre-labelled 300 mL glass jar to collect any adhered microplastics.

Following each run, the flume was thoroughly cleaned by twice flushing the system out with DI water. A negative control test was run three times at random points throughout the study to ensure there was no contamination from previous runs of the flume and to ensure there was no airborne contamination of similar microplastics throughout collection and processing. For these tests, the flume was run without any plastics, sediments, or vegetation and two water samples were taken at 5 and 10 min after priming the pump. Samples from the blank runs contained 1–3 PA fibres and 0–1 PET fibres, and no flakes or fragments were present. Sample processing is described in Supplementary Materials.

2.5. Data analysis

Data was assessed for normality (Shapiro-Wilks Test) and homogeneity of variances (Bartlett's test). When assumptions were met, ANOVA tests with post-hoc Tukey test were used. Alternatively, a Kruskal-Wallis test was used for non-parametric data. We tested whether there was a relationship between vegetation treatment and flow rate. To compare differences in flow rate across treatments and flume sections, individual tests were run for each section of the flume tank (i.e., upstream, midstream, downstream). Flow rate results are reported in the Supplementary Materials (Table S1; Figure S4). To measure the rate of microplastic deposition across each vegetation treatment, we calculated the microplastic loss rate from the water column by the equation:

$$\frac{Final\ MP\ concentration\ (T_{65}) - Initial\ MP\ concentration\ (T_5)}{Total\ time}$$

We tested whether there were effects on the microplastic loss rate from vegetation treatment or microplastic type. Only microfibres were included in the statistical tests because they were consistently present in the water samples, whereas the other microplastic types were not. Microplastics found on the vegetation were reported as microplastics per cm^2. We compared microplastic concentrations across vegetation treatment and across vegetation zone. We also tested a two-way ANOVA with both vegetation treatment and zone as factors. However, AIC model selection distinguished the one-way ANOVA with vegetation zone as a factor was the best fit model. Microplastics in sediment were standardized by microplastics per cm^2 and data was log transformed to maintain normality. We tested the relationship between vegetation treatment, sediment section, and microplastic concentration. We also calculated the percent relative abundance of each microplastic type across matrices and treatments by dividing the sum of each microplastic type by the total number of microplastics within matrix and treatment, multiplied by 100. Data was analysed using R statistical software (version 4.2.1) in an RStudio environment (version 2022.02.0 + 443), using significance level $p < 0.05$.

3. Results

3.1. Microplastic concentrations decreased in the water column over time

Microfibre concentration decreased over the course of the flume run in all treatments (Fig. 2; Figure S5). The mean ± standard deviation (s.d.) rate of microfibre loss in particles min^{-1} from the water column was 5.5 ± 1.5 for control, 6.2 ± 1.3 for grass, and 5.0 ± 0.9 for branched treatments. There was no significant interaction between the effects of treatment and microfibre type ($F_{2,\ 12} = 0.1$, $p = 0.9$). Furthermore, there was no significant difference in microfibre rate of loss across treatments ($p = 0.4$) or between microfibre types ($p = 0.3$; Fig. 2). The other microplastic types were generally found in low abundance in the water column, and the concentration of flakes in the branched treatment had a large variation at t20 and t35, but no notable pattern was discerned (Figure S5). Notably, flakes and fragments were observed to settle out of the water column quickly and travel along the sediment bed by saltation, rolling, and sliding.

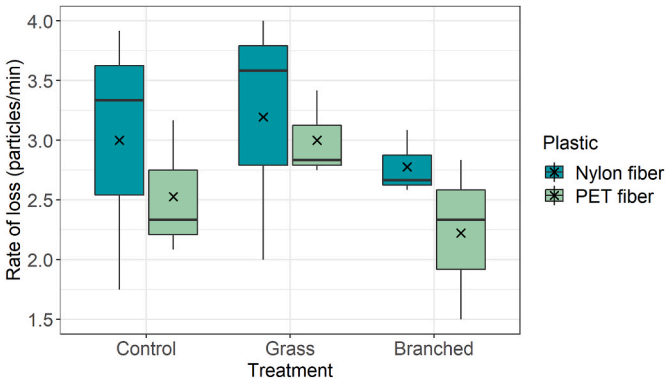

Fig. 2. Rate of microfibre loss from the water column over time, for each vegetation treatment. X represents the mean.

3.2. Microplastics were trapped on vegetation blades

The number of microplastics adhered to the artificial plants did not differ significantly between the two vegetation treatments (Fig. 3a). Grass and branched plants had a mean ± s.d. of 3.1 ± 1.8 and 3.3 ± 1.7 microplastics cm^{-2}, respectively ($F_{1,16} = 0.05$, $p = 0.8$). However, variance of the microplastic concentration was best explained by vegetation zone. There was a significant difference between plants positioned upstream than those positioned downstream (ANOVA: $F_{2,15} = 4.7$, $p = 0.03$; Tukey: $p = 0.03$). Plants positioned in the middle of the patch were not significantly different from the other groups ($p > 0.05$). However, when both treatment and vegetation zone were included as interacting parameters, neither the interaction ($F_{2,12} = 0.3$, $p = 0.8$) nor treatment ($p = 0.8$) was significant, and vegetation zone ($p = 0.05$) was borderline significant. The relative abundance of different microplastic types was similar across the two vegetation treatments, but the two types of flakes show a gradual decrease across vegetation zone for the branched treatment, and the grass treatment show an increase of both types of flakes in the midstream section (Fig. 3b).

3.3. Depositional patterns of microplastics in sediment varied

Total microplastic concentration in the sediment was not significantly different across treatments ($p = 0.5$), although there was more variability in the grass and branched treatments compared to the control (Fig. 4a). The sediment section where microplastics deposited was significantly different ($p = 0.009$) and the interaction between treatment and sediment section was significant ($F_{4,18} = 5.6$, $p = 0.001$). For the grass treatment, microplastic concentrations between the vegetated central section and sections −12.5 cm upstream and +25 cm downstream of the central sections was significantly different; significant differences were not observed between upstream and either downstream sections, or between the central section and the +12.5 cm downstream section (Fig. 4a; Table S2). In the branched treatment, microplastic concentrations were significantly different between the central section and the +12.5 cm downstream section (Fig. 4a; Table S2). With no vegetation (control), there was a gradual increase in microplastic deposition across the sediment bed with the peak accumulation zone at +12.5 cm downstream and then a decrease at +25 cm downstream, but these differences were not significant (Fig. 4a).

When microplastic concentrations in sediment are split across microplastic type, the patterns within and across treatments vary (Fig. 4b–f). Within the control treatment, microplastic fibre and fragment concentrations gradually increased and PET flakes decreased along the sediment bed, while PVC flakes were most abundant in the central section (Fig. 4). The patterns observed in the grass treatment are largely

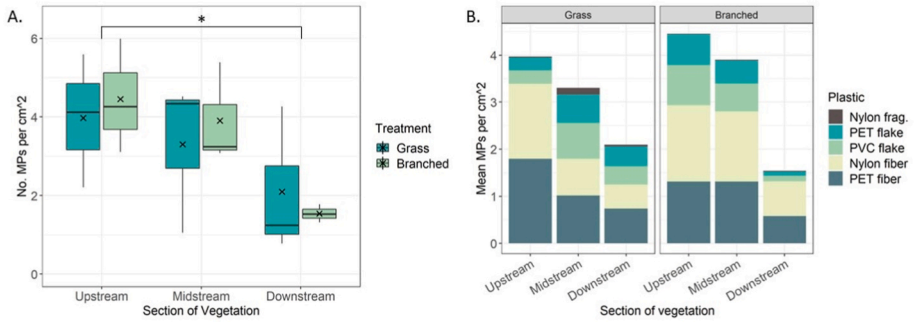

Fig. 3. (A) Number of microplastics (MPs) per cm^2 adhered to vegetation for each treatment and vegetation section. X represents the mean. Microplastic concentrations differed significantly between the upstream and downstream sections ($F_{2,15} = 4.7$, $p = 0.03$), but not by treatment. (B) Mean number of microplastic types found at each vegetation zone and for each treatment.

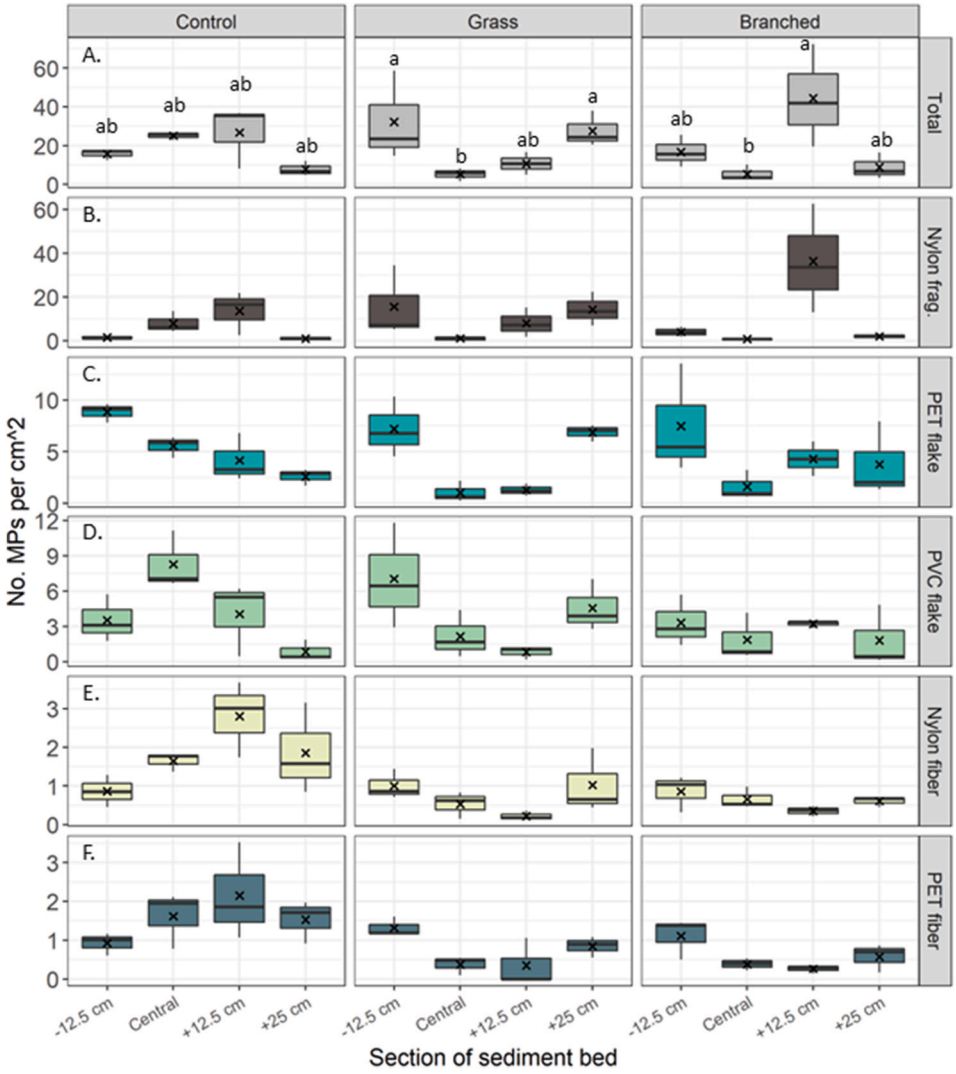

Fig. 4. Number of microplastics (MPs) per cm^2 in sediment for each vegetation treatment and sediment section, separated by (A) total microplastic and (B–F) individual microplastic type. X denotes the mean and letters denote significance groups based on post-hoc Tukey test.

the same across plastic types. All show a decrease in abundance along the sediment bed until they accumulate again at +25 cm downstream (Fig. 4). Additionally, the two types of fibres show the same pattern in the branched treatment as in the grass treatment and oppose the pattern observed in the control treatment, whereas the flakes and nylon fragments show different depositional patterns to the grass treatment. All show a decrease in the central section and an uptick in the +12.5 cm downstream position, which is most noticeable with the nylon fragments (Fig. 4). This increase in abundance in the downstream sections of the vegetation patch was observed during the grass and branched flume runs, where nylon fragments and some PVC and PET flakes were buried within a bedform just after the vegetation (Figure S6).

3.4. Relative abundance of each microplastic type differed across matrices

Microplastic concentrations in water, adhered to vegetation, or deposited in the sediment varied depending on the type of microplastic. There was a higher proportion of fibres in the water and adhered to the plants than in sediment. The two types of flakes were mainly found on the vegetation and in sediment and nylon fragments were primarily found in the sediment (Fig. 5a). Generally, these differences did not change between treatments, though the branched plants did have a higher proportion of the two flakes in the water column than the other treatments (Fig. 5b). These patterns are grouped mainly by shape (e.g., fibre vs. flake) rather than polymer type (e.g., PET vs. Nylon).

4. Discussion

In this experimental flume study, we observed that microplastic shape was an influencing factor in determining microplastic fate. For instance, a higher proportion of fibres were caught on the above ground canopy than deposited directly in sediment, while the opposite was true for flakes and fragments. Moreover, microfibre concentrations decreased from the leading edge of the vegetation canopy to the downstream end, indicating a filtering effect. Vegetation presence and complexity affected location of microplastic deposition. Burial of microplastics downstream of the vegetation patch was observed in both vegetation treatments but not the control and suggests that accumulation may occur over longer periods of time because of a potential for reduced resuspension.

4.1. Vegetated habitats as a microplastic sink

Microfibres settled out of the water column consistently over time and this decline was unaffected by the vegetation (Figure S5). Several studies have considered that plants act to reduce hydrodynamic flow causing microplastics to settle out (Cozzolino et al., 2022; de los Santos et al., 2021; de Smit et al., 2021; Ogbuagu et al., 2022; Waldschläger et al., 2022). In this study, the grass treatment did show a reduction in flow within the canopy, however it may not have been sufficient to promote higher rates of microplastic settlement as compared to the control treatment. This suggests that increased trapping of microplastics requires other factors in addition to purely vegetation presence. This may include canopy size, canopy density, presence of infauna and epibiont coverage on plant blades, and bed roughness (Cozzolino et al., 2020; de los Santos et al., 2021; Jones et al., 2020; Ogbuagu et al., 2022; Zhao et al., 2022).

While microplastic abundance in the sediment bed did not vary across vegetation treatments, the sites of highest microplastic deposition varied. Typically, highest microplastic concentrations were found prior to or after the vegetated canopies, as compared with the vegetated area itself. Obstacles in flowing water locally alter velocity and generate turbulence, which can cause increased sediment scour (increased erosion) at low obstacle density and skimming flow (less erosion) at high obstacle density (Mayaud et al., 2016). In the flume, we hypothesize that microplastics did not readily settle within the vegetation patches because both types of artificial plant increased flow velocity within the vegetation, and this was observed through increased scour at the base of the plants. At higher plant densities we hypothesize that skimming flow might result in a greater settling of microplastics within vegetation patches. The increased sediment scour is typical for the marsh grass *Spartina alterniflora*, where emergent stems with a narrow base followed by upper branching often create a maximum flow velocity near the bed (Leonard and Croft, 2006; Leonard and Luther, 1995; Nepf, 2012). As we measured flow velocity at a single point in the middle of the canopy and these changes in velocity are small, this maximum velocity would not have been measured. Downstream of the vegetation patch, water velocity decreases in the absence of obstructions, thereby creating a deposition zone outside of the vegetation (Figure S6) (Chen et al., 2012; Follett and Nepf, 2012). On a larger scale, this phenomenon occurs with grasses aiding the creation of sand dunes (Olson, 1958). While the concentration of microplastics in the downstream sections were not significantly different between the control and vegetative treatments, this downstream deposition zone caused by the vegetation resulted in the burial of microplastics. This burial indicates that vegetation could help prevent resuspension of microplastics rather than solely promoting deposition (Figure S6). Over time, this may enhance the accumulation of microplastics on the edges of vegetated sediments. This burial will be different for more muddy sediments which have different properties from sand, such as cohesion, reduced mobility and propensity for flocculation processes that promote microplastic adherence (Grabowski et al., 2011; Murray, 1977). However, Xu et al. (2023) also found that the reduction in sediment erosion by mangroves was the primary determinant of microplastic abundance rather than sediment accretion. The upstream accumulation of microplastics may be explained by a decrease in turbulent energy as water begins to flow through the vegetation, thereby causing sediments (and microplastics) to settle, and has been observed for sediments along tidal marsh creekbanks and edges (Leonard et al., 2002; Neubauer et al., 2002).

The accumulation of microplastics at the edges of vegetated canopies has been observed in tidal wetlands and mangrove forests (Duan et al., 2021; Helcoski et al., 2020; Yao et al., 2019) and is in accordance with fine sediment behaviour (Soler et al., 2020). However, flow conditions in the field are more complex and multidirectional (e.g., waves), whereas flow within a flume is restricted to unidirectional flow (Tinoco et al., 2020); and the flume used in this study is smaller than previous studies and may be more affected by constraints from flume width (Williams, 1970). While flumes are useful for determining specific drivers of microplastic trapping on a fine scale, it is only representative of the parameters used. Across larger spatial and temporal scales, other driving forces may have a greater influence on microplastic trapping than those observed here. For instance, water depth and velocity change across tidal periods creating variable flow patterns and much more

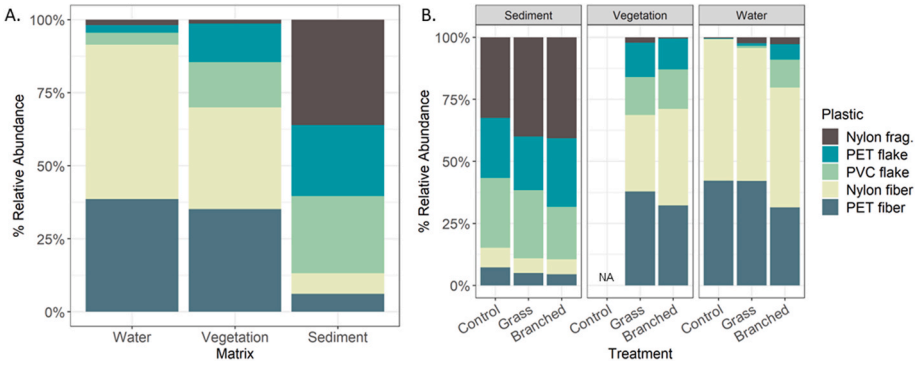

Fig. 5. Percent relative abundance of microplastic types across (A) matrices and (B) across vegetation treatments.

dynamic conditions (Neumeier and Ciavola, 2004; Tinoco et al., 2020). This work is a first step at isolating specific variables (i.e., physical structure of vegetation and microplastic type) that could be driving microplastics trapping in vegetated canopies, though there are many other drivers that can and have been explored (Cozzolino et al., 2022; de los Santos et al., 2021; de Smit et al., 2021; Ogbuagu et al., 2022).

4.2. Adherence to vegetation

Branched and grassy artificial plants both had microplastics adhered to their surface, with higher concentrations of microplastics adhered to plants on the leading edge of the canopy than at the downstream end (Fig. 3). This filtering pattern has been observed in the field with both macroplastics and microplastics (Navarrete-Fernández et al., 2022; Stead et al., 2020; Yao et al., 2019), where some have suggested that these coastal wetlands act as an estuarine 'filter' that mitigates plastic transport into the open marine environment (Biltcliff-Ward et al., 2022). Fibres were the predominant type of microplastic found attached to both types of plants. This is in alignment with what is found in the field, with a recent review showing microplastics have been identified on 10–100% of aquatic plant canopies and macroalgae sampled, with 76% of the microplastics identified being fibrous (Huang et al., 2023). Additionally, we observed some sand grains adhered to the surface of the artificial plants, usually at the node of where two blades connected or at leaf-stem nodes. Physical blockage, entanglement or hydrophobic attraction were likely the primary drivers of trapping in this simulated system. In living plants, microplastics adherence could further stem from entrapment on epibionts (e.g., algae, hydroids, bryozoans) that increase surface area and roughness, microbial biofilms that secrete extracellular polymeric substances (EPS), algal mucus layers with polysaccharide compounds, and root systems that can affect vertical migration of microplastics (Gutow et al., 2016; Jones et al., 2020; Li et al., 2023; Sfriso et al., 2021; Sundbæk et al., 2018; Zhao et al., 2022). Others have found that the submergence level, surface roughness, rigidity and flexibility, and complexity of aquatic canopies can affect trapping (Cozzolino et al., 2020; de Smit et al., 2021). As we used rigid and emergent stems, the microplastics were 'forced' to flow through and interact with the plants. In contrast, flexible and submerged vegetation may provide more movement and potentially fewer microplastic-plant interactions. Still, the adherence of microplastics to vegetated canopies suggests that only quantifying microplastics in the sediment bed may not tell the full story of microplastic fate and field studies that collect more than one sample type (e.g., sediment, vegetation, water) would be beneficial for comparisons within a geographic area.

4.3. Microplastic pathways

Here, we observed differences in microplastic fate primarily based on their shape (Fig. 5). In the environment, Helcoski et al. (2020) found a higher proportion of fragments in the sediment at the vegetation edge of a tidal wetland as compared to the interior of the vegetation. However, Huang et al. (2020) did not find a difference between microplastic shape in sediment inside and outside a seagrass canopy. A limitation of many existing field studies is that samples are taken from one area of a vegetated bed, so it is difficult to compare whether there are differences in concentrations of microplastic type across a bed. From flume studies, others have found differences based on polymer, noting that polymers with higher densities (PET and Nylon) are more likely to become trapped in a seagrass patch, but they did not compare differences among shapes (de los Santos et al., 2021). In sediments, grain shape does not differ as greatly as microplastic shape (e.g., fragments, fibres, flakes, spheres) and the differences between densities of commonly employed polymer particles (0.9–1.5 g/cm^3) is narrower compared to sediment and organic matter particle densities (0.9–3.0 g/cm^3; Harris, 2020). Due to this variability, microplastic shape may have a more meaningful influence on transport. However, Waldschläger and Schüttrumpf (2019) did not observe a large effect from microplastic shape on erosion thresholds as compared to microplastic densities, but did note that while not statistically tested, spheres and fibres showed particularly different erosion behaviour from other shapes. Another study investigating microplastic dispersion across a German estuary noted that high density fibres had similar dispersion patterns to non-fibrous low-density polymers (Enders et al., 2019).

5. Conclusions

Shape was the primary factor determining the fate of microplastics in a modelled coastal vegetative system. Presence of vegetation and differences in vegetation structure affected which area of the sediment bed microplastics deposited. Fibres were found in greatest abundance in the water and adhered to plants, whereas flakes were observed adhered to plants and deposited in sediment, and fragments were primarily found in the sediment. This pattern is indicative of how each microplastic shape travelled in this system: fibres were largely transported as suspended load, flakes travelled as suspended load and bedload, and nylon fragments were primarily transported as bedload. Here, we show that microplastic transport and deposition should not be generalised across all plastics, but rather differences in microplastic characteristics will affect where they accumulate and whether they are retained within a vegetated bed.

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CRediT authorship contribution statement

Hayley K. McIlwraith: Writing – original draft, Visualization, Methodology, Investigation, Formal analysis, Conceptualization. **Penelope K. Lindeque:** Writing – review & editing, Supervision, Methodology, Conceptualization. **Anastasia Miliou:** Writing – review & editing, Conceptualization. **Trevor J. Tolhurst:** Writing – review & editing, Supervision, Resources, Methodology, Conceptualization. **Matthew Cole:** Writing – review & editing, Supervision, Methodology, Funding acquisition, Conceptualization.

Declaration of competing interest

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

Data availability

Data will be made available on request.

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Appendix A. Supplementary data

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References

Athey, S.N., Erdle, L.M., 2022. Are we underestimating Anthropogenic microfiber pollution? A critical review of occurrence, methods, and reporting. Environ. Toxicol. Chem. 41, 822–837. https://doi.org/10.1002/etc.5173.

Ballent, A., Pando, S., Purser, A., Juliano, M.F., Thomsen, L., 2013. Modelled Transport of Benthic Marine Microplastic Pollution in the Nazaré Canyon 14.

Barbier, E.B., 2013. Valuing ecosystem services for coastal wetland protection and restoration: progress and challenges. Resources 2, 213–230. https://doi.org/10.3390/resources2030213.

Barbier, E.B., Hacker, S.D., Kennedy, C., Koch, E.W., Stier, A.C., Silliman, B.R., 2011. The value of estuarine and coastal ecosystem services. Ecol. Monogr. 81, 169–193. https://doi.org/10.1890/10-1510.1.

Biltcliff-Ward, A., Stead, J.L., Hudson, M.D., 2022. The estuarine plastics budget: a conceptual model and meta-analysis of microplastic abundance in estuarine systems. Estuar. Coast Shelf Sci. 275, 107963 https://doi.org/10.1016/j.ecss.2022.107963.

Bologna, P.A.X., Heck, K.L., 2002. Impact of habitat edges on density and secondary production of seagrass-associated fauna. Estuaries 25, 1033–1044. https://doi.org/10.1007/BF02691350.

Bouma, T.J., Temmerman, S., van Duren, L.A., Martini, E., Vandenbruwaene, W., Callaghan, D.P., Balke, T., Biermans, G., Klaassen, P.C., van Steeg, P., Dekker, F., van de Koppel, J., de Vries, M.B., Herman, P.M.J., 2013. Organism traits determine the strength of scale-dependent bio-geomorphic feedbacks: a flume study on three intertidal plant species. Geomorphology 180–181, 57–65. https://doi.org/10.1016/j.geomorph.2012.09.005.

Bridge, J.S., Bennett, S.J., 1992. A model for the entrainment and transport of sediment grains of mixed sizes, shapes, and densities. Water Resour. Res. 28, 337–363. https://doi.org/10.1029/91WR02570.

Browne, M.A., Galloway, T., Thompson, R., 2007. Microplastic—an emerging contaminant of potential concern? Integrated Environ. Assess. Manag. 3, 559–561. https://doi.org/10.1002/ieam.5630030412.

Chapman, J.A., Wilson, B.N., Gulliver, J.S., 2015. Drag force parameters of rigid and flexible vegetal elements. Water Resour. Res. 51, 3292–3302. https://doi.org/10.1002/2014WR015436.

Chen, Y., Li, Y., Thompson, C., Wang, X., Cai, T., Chang, Y., 2018. Differential sediment trapping abilities of mangrove and saltmarsh vegetation in a subtropical estuary. Geomorphology 318, 270–282. https://doi.org/10.1016/j.geomorph.2018.06.018.

Chen, Z., Ortiz, A., Zong, L., Nepf, H., 2012. The wake structure behind a porous obstruction and its implications for deposition near a finite patch of emergent vegetation. Water Resour. Res. 48 https://doi.org/10.1029/2012WR012224.

Chmura, G.L., Anisfeld, S.C., Cahoon, D.R., Lynch, J.C., 2003. Global carbon sequestration in tidal, saline wetland soils. Global Biogeochem. Cycles 17. https://doi.org/10.1029/2002GB001917.

Cole, M., Lindeque, P., Halsband, C., Galloway, T.S., 2011. Microplastics as contaminants in the marine environment: a review. Mar. Pollut. Bull. 62, 2588–2597. https://doi.org/10.1016/j.marpolbul.2011.09.025.

Cozzolino, L., Nicastro, K.R., Seuront, L., McQuaid, C.D., Zardi, G.I., 2022. The relative effects of interspecific and intraspecific diversity on microplastic trapping in coastal biogenic habitats. Sci. Total Environ. 848, 157771 https://doi.org/10.1016/j.scitotenv.2022.157771.

Cozzolino, L., Nicastro, K.R., Zardi, G.I., de los Santos, C.B., 2020. Species-specific plastic accumulation in the sediment and canopy of coastal vegetated habitats. Sci. Total Environ. 723, 138018 https://doi.org/10.1016/j.scitotenv.2020.138018.

de los Santos, C.B., Krång, A.-S., Infantes, E., 2021. Microplastic retention by marine vegetated canopies: simulations with seagrass meadows in a hydraulic flume. Environ. Pollut. 269, 116050 https://doi.org/10.1016/j.envpol.2020.116050.

de Smit, J.C., Anton, A., Martin, C., Rossbach, S., Bouma, T.J., Duarte, C.M., 2021. Habitat-forming species trap microplastics into coastal sediment sinks. Sci. Total Environ. 772, 145520 https://doi.org/10.1016/j.scitotenv.2021.145520.

Duan, J., Han, J., Cheung, S.G., Chong, R.K.Y., Lo, C.-M., Lee, F.W.-F., Xu, S.J.-L., Yang, Y., Tam, N.F., Zhou, H.-C., 2021. How mangrove plants affect microplastic distribution in sediments of coastal wetlands: case study in Shenzhen Bay, South China. Sci. Total Environ. 767, 144695 https://doi.org/10.1016/j.scitotenv.2020.144695.

Enders, K., Käppler, A., Biniasch, O., Feldens, P., Stollberg, N., Lange, X., Fischer, D., Eichhorn, K.-J., Pollehne, F., Oberbeckmann, S., Labrenz, M., 2019. Tracing microplastics in aquatic environments based on sediment analogies. Sci. Rep. 9, 15207 https://doi.org/10.1038/s41598-019-50508-2.

Follett, E.M., Nepf, H.M., 2012. Sediment patterns near a model patch of reedy emergent vegetation. Geomorphology 179, 141–151. https://doi.org/10.1016/j.geomorph.2012.08.006.

Gacia, E., Granata, T.C., Duarte, C.M., 1999. An approach to measurement of particle flux and sediment retention within seagrass (Posidonia oceanica) meadows. Aquat. Bot. 65, 255–268. https://doi.org/10.1016/S0304-3770(99)00044-3.

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.

Grabowski, R.C., Droppo, I.G., Wharton, G., 2011. Erodibility of cohesive sediment: the importance of sediment properties. Earth Sci. Rev. 105, 101–120. https://doi.org/10.1016/j.earscirev.2011.01.008.

Gutow, L., Eckerlebe, A., Giménez, L., Saborowski, R., 2016. Experimental evaluation of seaweeds as a vector for microplastics into marine food webs. Environ. Sci. Technol. 50, 915–923. https://doi.org/10.1021/acs.est.5b02431.

Harris, P.T., 2020. The fate of microplastic in marine sedimentary environments: a review and synthesis. Mar. Pollut. Bull. 158, 111398 https://doi.org/10.1016/j.marpolbul.2020.111398.

Harris, P.T., Westerveld, L., Nyberg, B., Maes, T., Macmillan-Lawler, M., Appelquist, L.R., 2021. Exposure of coastal environments to river-sourced plastic pollution. Sci. Total Environ. 769, 145222 https://doi.org/10.1016/j.scitotenv.2021.145222.

Helcoski, R., Yonkos, L.T., Sanchez, A., Baldwin, A.H., 2020. Wetland soil microplastics are negatively related to vegetation cover and stem density. Environ. Pollut. 256, 113391 https://doi.org/10.1016/j.envpol.2019.113391.

Hendriks, I.E., Bouma, T.J., Morris, E.P., Duarte, C.M., 2010. Effects of seagrasses and algae of the Caulerpa family on hydrodynamics and particle-trapping rates. Mar. Biol. 157, 473–481. https://doi.org/10.1007/s00227-009-1333-8.

Horton, A.A., Dixon, S.J., 2018. Microplastics: an introduction to environmental transport processes. WIREs Water 5, e1268. https://doi.org/10.1002/wat2.1268.

Huang, S., Jiang, R., Craig, N.J., Deng, H., He, W., Li, J.-Y., Su, L., 2023. Accumulation and re-distribution of microplastics via aquatic plants and macroalgae - a review of field studies. Mar. Environ. Res., 105951 https://doi.org/10.1016/j.marenvres.2023.105951.

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.

Infantes, E., Orfila, A., Simarro, G., Terrados, J., Luhar, M., Nepf, H., 2012. Effect of a seagrass (Posidonia oceanica) meadow on wave propagation. Mar. Ecol. Prog. Ser. 456, 63–72. https://doi.org/10.3354/meps09754.

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.

Kreitsberg, R., Raudna-Kristoffersen, M., Heinlaan, M., Ward, R., Visnapuu, M., Kisand, V., Meitern, R., Kotta, J., Tuvikene, A., 2021. Seagrass beds reveal high abundance of microplastic in sediments: a case study in the Baltic Sea. Mar. Pollut. Bull. 168, 112417 https://doi.org/10.1016/j.marpolbul.2021.112417.

Leonard, L.A., Croft, A.L., 2006. The effect of standing biomass on flow velocity and turbulence in Spartina alterniflora canopies. Estuar. Coast. Shelf Sci., Salt Marsh Geomorphology: Physical and ecological effects on landform 69, 325–336. https://doi.org/10.1016/j.ecss.2006.05.004.

Leonard, L.A., Luther, M.E., 1995. Flow hydrodynamics in tidal marsh canopies. Limnol. Oceanogr. 40, 1474–1484. https://doi.org/10.4319/lo.1995.40.8.1474.

Leonard, L.A., Wren, P.A., Beavers, R.L., 2002. Flow dynamics and sedimentation in Spartina alterniflora and Phragmites australis marshes of the Chesapeake Bay. Wetlands 22, 415–424. https://doi.org/10.1672/0277-5212(2002)022[0415:FDASIS]2.0.CO;2.

Li, T., Wang, Y., Jiao, M., Zhao, Z., Li, R., Qin, C., 2023. Distinct microplastics abundance variation in root-associated sediments revealed the underestimation of mangrove microplastics pollution. Sci. Total Environ. 899, 165611 https://doi.org/10.1016/j.scitotenv.2023.165611.

Lloret, J., Pedrosa-Pamies, R., Vandal, N., Rorty, R., Ritchie, M., McGuire, C., Chenoweth, K., Valiela, I., 2021. Salt marsh sediments act as sinks for microplastics and reveal effects of current and historical land use changes. Environ. Adv. 4, 100060 https://doi.org/10.1016/j.envadv.2021.100060.

Lofty, J., Valero, D., Wilson, C.A.M.E., Franca, M.J., Ouro, P., 2023. Microplastic and natural sediment in bed load saltation: Material does not dictate the fate. Water Res. 243, 120329 https://doi.org/10.1016/j.watres.2023.120329.

Mayaud, J.R., Wiggs, G.F.S., Bailey, R.M., 2016. Dynamics of skimming flow in the wake of a vegetation patch. Aeolian Res 22, 141–151. https://doi.org/10.1016/j.aeolia.2016.08.001.

Mendrik, F., Fernández, R., Hackney, C.R., Waller, C., Parsons, D.R., 2023. Non-buoyant microplastic settling velocity varies with biofilm growth and ambient water salinity. Commun. Earth Environ. 4, 1–9. https://doi.org/10.1038/s43247-023-00690-z.

Möller, I., Kudella, M., Rupprecht, F., Spencer, T., Paul, M., van Wesenbeeck, B.K., Wolters, G., Jensen, K., Bouma, T.J., Miranda-Lange, M., Schimmels, S., 2014. Wave attenuation over coastal salt marshes under storm surge conditions. Nat. Geosci. 7, 727–731. https://doi.org/10.1038/ngeo2251.

Murray, W.A., 1977. Erodibility of Coarse sand-Clayey silt mixtures. J. Hydraul. Div. 103, 1222–1227. https://doi.org/10.1061/JYCEAJ.0004856.

Navarrete-Fernández, T., Bermejo, R., Hernández, I., Deidun, A., Andreu-Cazenave, M., Cózar, A., 2022. The role of seagrass meadows in the coastal trapping of litter. Mar. Pollut. Bull. 174, 113299 https://doi.org/10.1016/j.marpolbul.2021.113299.

Nepf, H.M., 2012. Flow and transport in regions with aquatic vegetation. Annu. Rev. Fluid Mech. 44, 123–142. https://doi.org/10.1146/annurev-fluid-120710-101048.

Neubauer, S.C., Anderson, I.C., Constantine, J.A., Kuehl, S.A., 2002. Sediment Deposition and Accretion in a Mid-Atlantic (U.S.A.) Tidal Freshwater Marsh. Estuar. Coast Shelf Sci. 54, 713–727. https://doi.org/10.1006/ecss.2001.0854.

Neumeier, U., Amos, C.L., 2006. The influence of vegetation on turbulence and flow velocities in European salt-marshes. Sedimentology 53, 259–277. https://doi.org/10.1111/j.1365-3091.2006.00772.x.

Neumeier, U., Ciavola, P., 2004. Flow Resistance and Associated Sedimentary Processes in a Spartina maritima Salt-Marsh. J. Coast Res. 20, 435–447. https://doi.org/10.2112/1551-5036(2004)020[0435:FRAASP]2.0.CO;2.

Ogbuagu, C.C., Kassem, H., Udiba, U.U., Stead, J.L., Cundy, A.B., 2022. Role of saltmarsh systems in estuarine trapping of microplastics. Sci. Rep. 12, 15546 https://doi.org/10.1038/s41598-022-18881-7.

Olson, J.S., 1958. Lake Michigan Dune Development 2. Plants as Agents and Tools in Geomorphology. J. Geol. 66, 345–351.

Ouyang, X., Duarte, C.M., Cheung, S.-G., Tam, N.F.-Y., Cannicci, S., Martin, C., Lo, H.S., Lee, S.Y., 2022. Fate and Effects of Macro- and Microplastics in Coastal Wetlands. Environ. Sci. Technol. 56, 2386–2397. https://doi.org/10.1021/acs.est.1c06732.

Paduani, M., 2020. Microplastics as novel sedimentary particles in coastal wetlands: A review. Mar. Pollut. Bull. 161, 111739 https://doi.org/10.1016/j.marpolbul.2020.111739.

Pinheiro, L.M., Britz, L.M.K., Agostini, V.O., Pérez-Parada, A., García-Rodríguez, F., Galloway, T.S., Pinho, G.L.L., 2022. Salt marshes as the final watershed fate for meso- and microplastic contamination: A case study from Southern Brazil. Sci. Total Environ. 838, 156077 https://doi.org/10.1016/j.scitotenv.2022.156077.

Sfriso, A.A., Tomio, Y., Juhmani, A.-S., Sfriso, A., Munari, C., Mistri, M., 2021. Macrophytes: A Temporary Sink for Microplastics in Transitional Water Systems. Water 13, 3032. https://doi.org/10.3390/w13213032.

Soler, M., Colomer, J., Folkard, A., Serra, T., 2020. Particle size segregation of turbidity current deposits in vegetated canopies. Sci. Total Environ. 703, 134784 https://doi.org/10.1016/j.scitotenv.2019.134784.

Spalding, M.D., McIvor, A.L., Beck, M.W., Koch, E.W., Möller, I., Reed, D.J., Rubinoff, P., Spencer, T., Tolhurst, T.J., Wamsley, T.V., van Wesenbeeck, B.K., Wolanski, E., Woodroffe, C.D., 2014. Coastal Ecosystems: A Critical Element of Risk Reduction. Conserv. Lett. 7, 293–301. https://doi.org/10.1111/conl.12074.

Stead, J.L., Cundy, A.B., Hudson, M.D., Thompson, C.E.L., Williams, I.D., Russell, A.E., Pabortsava, K., 2020. Identification of tidal trapping of microplastics in a temperate salt marsh system using sea surface microlayer sampling. Sci. Rep. 10, 14147 https://doi.org/10.1038/s41598-020-70306-5.

Sundbæk, K.B., Koch, I.D.W., Villaro, C.G., Rasmussen, N.S., Holdt, S.L., Hartmann, N.B., 2018. Sorption of fluorescent polystyrene microplastic particles to edible seaweed Fucus vesiculosus. J. Appl. Phycol. 30, 2923–2927. https://doi.org/10.1007/s10811-018-1472-8.

Terrados, J., Duarte, C.M., 2000. Experimental evidence of reduced particle resuspension within a seagrass (Posidonia oceanica L.) meadow. J. Exp. Mar. Biol. Ecol. 243, 45–53. https://doi.org/10.1016/S0022-0981(99)00110-0.

Tinoco, R.O., San Juan, J.E., Mullarney, J.C., 2020. Simplification bias: lessons from laboratory and field experiments on flow through aquatic vegetation. Earth Surf. Process. Landforms 45, 121–143. https://doi.org/10.1002/esp.4743.

Unsworth, R.K.F., Higgs, A., Walter, B., Cullen-Unsworth, L.C., Inman, I., Jones, B.L., 2021. Canopy Accumulation: Are Seagrass Meadows a Sink of Microplastics? Oceans 2, 162–178. https://doi.org/10.3390/oceans2010010.

Waldschläger, K., Brückner, M.Z.M., Carney Almroth, B., Hackney, C.R., Adyel, T.M., Alimi, O.S., Belontz, S.L., Cowger, W., Doyle, D., Gray, A., Kane, I., Kooi, M., Kramer, M., Lechthaler, S., Michie, L., Nordam, T., Pohl, F., Russell, C., Thit, A., Umar, W., Valero, D., Varrani, A., Warrier, A.K., Woodall, L.C., Wu, N., 2022. Learning from natural sediments to tackle microplastics challenges: A multidisciplinary perspective. Earth Sci. Rev. 228, 104021 https://doi.org/10.1016/j.earscirev.2022.104021.

Waldschläger, K., Schüttrumpf, H., 2019. Erosion Behavior of Different Microplastic Particles in Comparison to Natural Sediments. Environ. Sci. Technol. 53, 13219–13227. https://doi.org/10.1021/acs.est.9b05394.

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, 1632–1640. https://doi.org/10.1002/etc.3432.

Wilkie, L., O'Hare, M.T., Davidson, I., Dudley, B., Paterson, D.M., 2012. Particle trapping and retention by Zostera noltii: A flume and field study. Aquat. Bot. 102, 15–22. https://doi.org/10.1016/j.aquabot.2012.04.004.

Williams, G.P., 1970. Flume Width and Water Depth Effects in Sediment-Transport Experiments. U.S. Government Printing Office.

Wright, J., Hovey, R.K., Paterson, H., Stead, J., Cundy, A., 2023. Microplastic accumulation in Halophila ovalis beds in the Swan-Canning Estuary, Western Australia. Mar. Pollut. Bull. 187, 114480 https://doi.org/10.1016/j.marpolbul.2022.114480.

Wu, F., Pennings, S.C., Tong, C., Xu, Y., 2020. Variation in microplastics composition at small spatial and temporal scales in a tidal flat of the Yangtze Estuary, China. Sci. Total Environ. 699, 134252 https://doi.org/10.1016/j.scitotenv.2019.134252.

Xu, N., Zhu, Z., Li, S., Ouyang, X., Zhu, Q., Gao, W., Cai, Y., Yang, Z., 2023. The role of bio-geomorphic feedbacks in shaping microplastic burial in blue carbon habitats. Sci. Total Environ. 861, 160220 https://doi.org/10.1016/j.scitotenv.2022.160220.

Yao, W., Di, D., Wang, Z., Liao, Z., Huang, H., Mei, K., Dahlgren, R.A., Zhang, M., Shang, X., 2019. Micro- and macroplastic accumulation in a newly formed Spartina alterniflora colonized estuarine saltmarsh in southeast China. Mar. Pollut. Bull. 149, 110636 https://doi.org/10.1016/j.marpolbul.2019.110636.

Zhao, L., Ru, S., He, J., Zhang, Z., Song, X., Wang, D., Li, X., Wang, J., 2022. Eelgrass (Zostera marina) and its epiphytic bacteria facilitate the sinking of microplastics in the seawater. Environ. Pollut. 292, 118337 https://doi.org/10.1016/j.envpol.2021.118337.