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Microplastics interact with benthic biostabilization processes

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Abstract

Marine microplastics (MPs) accumulate in sediments but impacts on ecosystem functions are poorly understood. MPs interactions with stabilizing benthic flora/fauna or biostabilization processes, have not been fully investigated, yet this is critical for unraveling MPs effects on ecosystem-scale processes and functions. This is also vital for understanding feedback processes that may moderate the stock and flow of MPs as they are transported through estuaries. The relationships between sedimentary MPs, biota, environmental properties and sediment stability from field sediments, were examined using variance partitioning (VP) and correlation analyses. VP was used to identify common and unique contributions of different groups of variables (environmental, fauna and microplastic variables) to sediment stability. The influence of microplastic presence (fragment/fiber abundances and microplastic diversity) on sediment stability (defined using erosion thresholds and erosion rates) was demonstrated. Furthermore, MPs appeared to mediate the biostabilizing effects of environmental properties (including microorganisms) and fauna. Environmental properties and sediment stability could also explain the variation in MPs across sites suggesting biostabilizing properties may mediate the abundance, type and diversity of MPs that accumulate in the bed. The potential for MPs to influence biota and biostabilization processes and mediate microplastic resuspension dynamics within estuaries is discussed.

1. Introduction

Soft sediments are a sink for marine microplastics (MPs; Brandon *et al* 2019), but many studies have struggled to link MP distribution to environmental properties and processes associated with intertidal habitats (Browne *et al* 2010, Alomar *et al* 2016). Despite recent studies revealing functional effects of MPs (Green *et al* 2017, Seeley *et al* 2020, Hope *et al* 2020a), their potential impact on biostabilization and sediment transport processes has not been examined. As benthic microalgae (BMA), bacteria and fauna are central to many functions (Pinckney 2018, Schenone and Thrush 2020, Hope *et al* 2020b), MP effects on biota and processes associated with them may have major consequences for entire ecosystems (Horton and Barnes 2020, Stubbins *et al* 2021). This has the potential to effect the ecosystem services (Sridharan *et al* 2021). This is critical to understand, considering the importance of coastal ecosystems, the projected increase MP pollution in coastal zones (Hale *et al* 2020), increasing storms, precipitation and sea level rise that threaten coastal zones (Lehmann *et al* 2015, McEvoy *et al* 2021) and the significant role biostabilization plays in mediating sediment dynamics (Malarkey *et al* 2015, Parsons *et al* 2016, Hope *et al* 2020c).

MPs can alter the BMA community structure, biomass or functional roles in a variety of ways. Toxic effects on cells from leaching additives and pollutants, are complicated by differences in MP size, structure and charge (Garrido *et al* 2019, Capolupo *et al* 2020, Nava and Leoni 2021). Low doses of additives can cause hormesis and/or reductions in cellular metabolic activity and the lipid content of cells but can also induce extracellular polymeric substances (EPS) production from the cells (Seoane *et al* 2019, Song *et al* 2020). Reductions in algae biomass and their photosynthetic capacity have been documented in the presence of MPs (Mao *et al* 2018, Nava and Leoni 2021), and biogeochemical processes associated with biofilms can be altered (Seeley *et al* 2020, Galgani and Loiselle 2021). Several studies have demonstrated sub-lethal and species-specific microorganism responses (Hope *et al* 2020a, Nava and Leoni 2021), but the mechanistic effects inducing these responses often remain elusive and variable.

Non-toxic effects may be linked to changes in carbon quality and quantity or changes to the structure and function of microbial biofilms (both on the seafloor and on floating particles). By acting as an alternative carbon substrate, the recalcitrant carbon backbone of MPs can benefit some microorganisms and negatively affect others (Romera-Castillo *et al* 2018, Zhu *et al* 2020) instigating a shift in metabolic processes. If this MP carbon replaces BMA-derived carbon in the bed, this will alter BMA-bacteria interactions and feedbacks that underpin several ecosystem functions. It will also alter the formation of stabilizing biofilms, as BMA-derived carbon secretions are fundamental to biofilm structure and biostabilization (Hope *et al* 2020c). Finally, as fauna also play a functional role in biostabilization (Joensuu *et al* 2018), changes to faunal-BMA-bacteria interactions and/or particular feeding and burrowing traits due to MP presence or ingestion (Green *et al* 2017) will have additional effects on biostabilization processes.

If MPs are influencing biostabilizing biota and resuspension dynamics, this emerging contaminant also has the potential to mediate its own redistribution dynamics as sediments become more or less cohesive. There has been limited consideration of the physical processes that influence MP transport to date (Kane and Clare 2019, Harris 2020), but no evaluation of the role of biostabilization for MP retention or remobilization. As long-term monitoring data is still lacking, it is difficult to fully comprehend whether MPs are immobilized once deposited on the bed, or frequently resuspended and transported between systems. Nonetheless, we are starting to understand the unique characteristics of MPs that may influence these dynamics (Waldschläger and Schüttrumpf 2020). Additionally, a greater understanding of the processes that may influence these dynamics, such as biostabilization, is vital to facilitate future modeling efforts. This is particularly important given MP are ubiquitous, but research is still in its infancy and methodological and characterization methods are not yet standardized.

The objectives of this present study were therefore to (a) evaluate whether MP accumulation has the potential to negatively affect biota and key interactions associated with biostabilization processes and (b) evaluate whether environmental properties, sediment stability and fauna influence MP accumulation on the bed. The hypothesis was that different MPs would be associated with the bio-physical properties of the beds and that MP accumulation would negatively influence sediment stability.

2. Methods

2.1. Site selection

The Waitemata is a moderately sized (80 km^2) Harbor, situated on the East Coast of New Zealand. The Harbor is predominately surrounded by Auckland city, the largest city in New Zealand (population of 1.5 million). The harbor drains a catchment of approximately 427 km^2, has a spring tidal range of 2.7 m, and a fluvial discharge of 8.9×105 m^3 per tidal cycle. Due to its proximity to Auckland city, the Waitemata is one of the more human impacted harbors in New Zealand.

2.2. Experimental design & sample collection

Samples were collected from six intertidal sites in the Waitemata Harbor, New Zealand (Lat: Long provided in table 1). Sediment cores (2.6 cm ID, 1 cm depth) were collected for biochemical analysis of the sediment (with 3 $\times$ pooled cores per replicate, and three true, independent replicate MP and biochemical samples per site). Biochemical cores were kept dark on dry ice, until flash frozen in liquid nitrogen on shore. Surface sediment (1 cm depth) was also collected from randomly placed quadrats (25 cm $\times$ 25 cm) within 3 m of the cores for MP extractions, where all surface material within the quadrat was transferred to pre-rinsed 500 ml glass jars using a metal spoon. Additional cores (2 $\times$ pooled cores, three replicates, 2.6 cm ID, 3 cm depth) were collected and transported on ice for later extraction in the laboratory to determine porewater ammonium concentrations. Finally, triplicate intact sediment cores (10 cm diameter, 10 cm depth) were retrieved from the midshore to erode in the laboratory (<8 h).

2.3. Microplastic extraction, quantification and characterization

MPs were extracted using a two-step extraction process (following Claessens *et al* 2013); elutriation through a custom-built unit, followed by the removal of organic matter, density separation using NaCl and filtration (Hope *et al* 2021). MP particles (>70 μm) were isolated, characterized and quantified using a Leica MS5 microscope (40$\times$ magnification) and assigned particles to five different morphologies; fragments, fibers, beads, film or fiber bundles. Identification largely followed Hidalgo-Ruz *et al* (2012), with ambiguous plastic polymers (mostly fibers) distinguished from pieces of remaining algae using a hot needle test (Directive 2013, De Witte

Table 1. The mean ± SE of key environmental characteristics from the six sites (GPS co-ordinates) in the Waitemata Harbor, New Zealand. MGS—mean grain size, chl a—chlorophyll a content, OM—organic matter content, U_{crit}—erosion threshold of the sediment surface, ER—erosion rate for surface sediment, m_e—erosion constant for subsurface erosion.

	Site names (GPS co-ordinates)					
	Big Shoal (174.7676, −36.8036)	Kelvin Strand (174.6564, −36.8274)	Little Shoal (174.7413, −36.8179)	Lwr Pt Chev (174.6951, −36.8687)	Motorway (174.7516, −36.8205)	Scout Club (174.6642, −36.7792)
Mud content (%)	6.0 ± 0.5	0.5 ± 0.1	4.5 ± 0.4	9.1 ± 2.1	2.9 ± 0.2	10.7 ± 0.3
MGS (μm)	168 ± 3	221 ± 9	166 ± 4	182 ± 5	209 ± 1	211 ± 8
Chl a (μg g DW sed^{-1})	28.9 ± 2.3	8.3 ± 0.4	28.1 ± 1.8	39.0 ± 1.5	16.0 ± 0.2	22.9 ± 1.5
OM (%)	1.95 ± 0.05	0.92 ± 0.06	1.44 ± 0.03	1.75 ± 0.05	0.96 ± 0.03	1.71 ± 0.08
Fauna Shannon's diversity index (0–1)	0.83 ± 0.07	0.85 ± 0.08	0.83 ± 0.03	0.78 ± 0.01	0.49 ± 0.02	0.70 + 0.01
Density Large bioturbators	527 ± 190	125 ± 50	377 ± 87	603 ± 190	753 ± 190	678 ± 345
MP Fragment abund (m^{-2})	128 ± 46	32 ± 24	101 ± 37	101 ± 11	69 ± 32	69 ± 19
MP Fiber abund (m^{-2})	75 ± 21	160 ± 24	94 ± 63	69 ± 11	149 ± 46	101 ± 35
MP Shannon's diversity index (0–1)	0.86 ± 0.03	0.82 ± 0.04	0.84 ± 0.04	0.92 ± 0.02	0.90 ± 0.05	0.92 ± 0.07
U_{crit} (Nm^{-2})	0.43 ± 0.03	1.00 ± 0.06	0.47 ± 0.09	0.80 ± 0.10	1.17 ± 0.07	1.07 ± 0.19
ER (g sed m^{-2} s^{-1})	0.21 ± 0.06	0.00 ± 0.00	0.12 ± 0.02	0.02 ± 0.01	0.00 ± 0.00	0.02 + 0.01
$m_{e\,()}$ (g N^{-1} s^{-1})	1.97 ± 043	11.90 ± 2.21	8.17 ± 3.33	2.20 ± 0.50	9.80 ± 0.38	8.23 ± 2.57

et al 2014) or isolated for chemical characterization using Fourier transform infrared (FTIR) spectroscopy using Primpke *et al* (2018) as a spectral reference database. A random selection of MPs (across 21 sites used as part of a wider study) were also examined using FTIR to confirm they were indeed synthetic. All brown and dark green 'fibers' were removed from the analysis as these were found to be of natural origin. For the remaining particles, MP color/type combinations were defined as individual MP 'categories' following Hope *et al* (2021), e.g. all 'green fragments' were distinguished as a single MP category and 'blue fibers' was another category.

2.4. MPs quality assurance/quality control

During all steps of sample collection and processing, field and lab workers avoided synthetic clothing and the use of plastic equipment, where possible. In order to account for potential contamination of samples during laboratory processing, positive and negative controls were carried out throughout each step. During elutriation, the unit was maintained in an unoccupied room, with minimal airflow. Large wet filter papers were exposed to the atmosphere during the elutriation of samples, with the elutriation unit and 63 μm collection sieve were covered during processing to minimize exposure to airborne fibers. During the subsequent density separation and digestion of samples, three procedural blank samples (filtered seawater only) were processed with each batch of samples following Lusher *et al* (2017) and Brander *et al* (2020). Filtered seawater samples were treated the same as the sediment samples (i.e. density separated in NaCl twice, digested in 15% H_2O_2 and filtered). The average number of particles (typically fibers) observed in the blanks was subtracted from the corresponding batch of samples. Samples were kept covered when possible. Additional wet filter papers were exposed to the atmosphere during processing, when samples were uncovered, to account for atmospheric contamination. Sample separation and filtration were carried out in a fume hood with the benchtop, filtration equipment & glassware all rinsed twice with MilliQ water before use. Natural clothing was worn during sample collection and laboratory analysis where possible, however each analysts clothing was also noted and compared to identified particles on counted filter papers for that day. During microscopic examination, new filter papers were exposed to the air adjacent to the sample under the microscope. These were examined for contamination before and after each sample count with samples adjusted accordingly. MP abundance was corrected by subtracting contamination on the negative controls before expressing the final counts as particle number per meter.

2.5. Sediment erosion measurements

Sediment erosion potential was measured in the large cores using a portable EROMES-device (Schunemann and Kuhl 1991) following Andersen (2001). Further details on the set up and methods are provided in the supplementary material (available online at stacks.iop.org/ERL/16/124058/mmedia). To allow three erosion potential measurements to be determined; the erosion threshold (U_{crit}, N m^{-2}), the erosion rate (ER, g m^{-2} s^{-1}) and the subsurface erosion constant (m_e, g N^{-1} s^{-1}), the ER (g m^{-2} s^{-1}) was first plotted as a function of nominal bed shear stress (BSS), following Andersen (2001). The U_{crit}

at which 0.1 g m^{-2} s^{-1} had occurred was selected to describe the initial erosion of sediment after the removal of unconsolidated, organic particles. This was defined as 'Type-Ib' erosion (Amos *et al* 1992). ER was defined at a commonly used, fixed BSS of 0.5 Nm^{-2} (Andersen 2001, Harris *et al* 2016) and the subsurface erosion constant (m_e), was determined from the slope of the linear relationship between ER and BSS after erosion had initiated (between 1.0 and 1.6 Nm^{-2}; Harris *et al* 2016). This was used to describe the change in ER with increasing BSS deeper in the bed (Mitchener and Torfs 1996), as a measure of 'Type-II' erosion. As U_{crit} refers to the shear stress required to initiate the movement of sediment grains, a higher U_{crit} defines a more stable sediment bed (higher shear stress is required), while a higher ER indicates less stable sediment (sediments are eroded from the bed more rapidly) and a higher m_e denotes a more rapidly eroding subsurface.

Water samples were extracted immediately after visible erosion had occurred in each core (>10% increase in turbidity) and filtered with the filtrates and filter papers frozen to determine the flow-induced, bentho-pelagic exchange of dissolved NH_4^+ (Eroded (E)—NH_4^+) and BMA cells (Eroded (E)—chl a), respectively. After erosion measurements were completed, the remaining sediment core was sieved (500 μm) to collect macrofauna which were preserved in 70% isopropyl alcohol (IPA) stained with rose Bengal.

2.6. Biochemical and physical characteristics of sediment and resuspended material

Fauna were identified to the lowest taxonomic level possible (typically species) at the National Institute of Water and Atmospheric Research, New Zealand. Microalgal pigments (chlorophyll a and pheophytins, 90% acetone method; Lorenzen 1967), colloidal carbohydrate fraction of (EPS-carbohydrates, Phenol-sulfuric assay; Dubois *et al* 1956) and labile organic matter content (OM, loss on ignition, Parker 1983) were determined from freeze dried sediments and standardized by sediment weight (μg g^{-1} dry weight (DW) sediment). Porewater was extracted by centrifugation and filtrates frozen until dissolved ammonium (NH_4^+) concentrations (corrected for porosity and dilutions) were determined using a Lachat QuickChem 8500+ FIA nutrient autoanalyzer (Zellweger Analytics Inc. Milwaukee, Wisconsin, 53 218, USA). Particle size distribution was determined from digested sediments (6% hydrogen peroxide) that were homogenized and run through a Malvern Mastersizer 3000 (range 0.05–2000 μm) to obtain mean grain size (MGS) and mud content (%) (Singer *et al* 1988).

2.7. Statistical analysis

Community indices (richness, abundance, evenness, Shannon's diversity (H') and Simpson's diversity) were applied to the MP 'category' matrix for further analysis alongside the abundance of different MP types and sizes. To determine categorical differences in key biochemical, physical and MP characteristics sites, one-way permutational ANOVA tests (PERMANOVA) were performed on individual environmental parameters (PRIMER, v.7; Anderson *et al* 2008). Site was treated as a random factor (six levels), and 9999 permutations used to provide Pseudo-F statistics and p-values. Spearman's rank correlation coefficients were calculated to examine the relationships between key biochemical and physical characteristics, MP characteristics and erosion measurements using the Hmisc package in R statistical software (version 3.1.1; R Development Core Team 2014) and the R studio graphical interface (v. 0.98.1083). To understand the relative influence of MP accumulation on biostabilization, the variation in sediment stability measures (U_{crit}, ER, m_e) explained by combinations of: (a) MP characteristics (fragment/fiber abundance and overall diversity); (b) key environmental variables associated with biostabilization (chl a, OM, MGS, cyanobacteria); and (c) faunal community metrics (presence of large cockles, all large bioturbators, species richness and species diversity) were evaluated using the variance partitioning analysis in the 'varpart' function of the 'vegan' R package (Oksanen *et al* 2019). This allowed us to estimate the individual and shared contribution of predictor matrices (a, b and c). This package automatically conducts variation partitioning of a response table with respect to tables of explanatory variables using redundancy analysis (RDA)-adjusted r^2 values This allowed us to determine the significance of adjusted r^2 values for each set of explanatory (MPs, environmental, faunal & stability) variables, using partial redundancy analysis and 999 permutations (Borcard *et al* 1992, Peres-Neto *et al* 2006). As the MP matrix and environmental variable matrix shared explained variance, the analyses were performed again to compare the explained variance with the exclusion of the MP matrix and the environmental variable matrix respectively, to evaluate the relative importance of MP effects in addition to environmental properties and biota for sediment stability. MP accumulation, retention and remobilization may also be influenced by ecosystem structure and function, therefore the potential influence of environmental properties, stability and fauna on the observed MPs characteristics (i.e. the MP matrix as the response) was also evaluated.

3. Results

3.1. Habitat characteristics

Sediments were largely non-cohesive (MGS; 166–220 μm and mud content; 0.5%–10.5%; table 1), but both MGS and mud content varied significantly across the sites (Pseudo-F = 16.51, $p < 0.001$

and Pseudo-F = 17.99, $p < 0.001$ respectively). OM content, BMA biomass and porewater NH_4^+ (Pseudo-F = 73.94, $p < 0.001$; Pseudo-F = 51.63, $p < 0.001$ and Pseudo-F = 73.44, $p < 0.001$) also varied across the sites and generally increased with mud content ($r_s = 0.78$, $p < 0.001$; $r_s = 0.66$, $p < 0.01$ and $r_s = 0.94$, $p < 0.001$).

3.2. Biostabilization

The sediment erosion threshold (U_{crit}) was higher at sites containing coarser sediment particles (supplementary figure S1(a); $r_s = 0.70$, $p < 0.01$). Despite their role in biostabilization, OM content did not correlate with the erosion threshold (U_{crit}), and U_{crit} was higher when there was less BMA biomass (chl a content; $r_s = -0.47$, $p < 0.05$) on the sediment surface. Additionally, faunal richness ($r_s = -0.46$, $p = 0.05$) and diversity (Simpson's; $r_s = -0.47$, $p = 0.05$) were higher in surface sediment with lower thresholds, and richness greater in more mobile sediments (higher ER; $r_s = 0.61$, $p < 0.01$). Biostabilization properties had more of an effect on the stability of deeper sediment layers. For example, lower ER and less subsurface erosion (m_e) were observed in beds with higher OM content ($r_s = -0.71$, $p < 0.001$ and $r_s = -0.74$, $p < 0.001$ respectively) and higher BMA biomass ($r_s = -0.67$, $p < 0.01$ and $r_s = -0.79$, $p < 0.001$ respectively, supplementary figure S1(b) and table S2).

3.3. Erodibility & MPs

Erodibility measures and a number of biophysical properties known to moderate sediment stability correlated with observed MP characteristics (supplementary table S2). Fragments were more abundant in sediments with higher BMA biomass ($r_s = 0.51$, $p = 0.05$), and where the underlying sediment were more stable (slower subsurface erosion, m_e; supplementary figure S2(a), $r_s = -0.59$, $p < 0.01$). Even with these stable underlying sediments, higher fragment numbers related to lower surface erosion thresholds ($r_s = -0.48$, $p < 0.05$; supplementary figure S2(b)) and rapid erosion immediately after the threshold had been surpassed (supplementary figure S3; $r_s = 0.54$, $p < 0.05$). Fibers on the other hand, were more abundant in sediments with lower BMA biomass (supplementary figure S4(a); $r^2 = -0.26$, $p < 0.05$), less OM ($r^2 = -0.44$, $p < 0.01$), and less stable subsurface sediments (higher m_e, $r_s = 0.65$, $p < 0.01$, supplementary figure S4(b)).

Chl a (proxy of BMA resuspension) and dissolved NH_4^+ were released into the water column during erosion measurements, however eroded-NH_4^+ did not correlate with any measured sediment or MP characteristics. In contrast, a greater flow-induced release of BMA cells (higher eroded-chl a concentration) was observed from more mobile subsurface sediments (higher m_e; $r_s = 0.48$, $p < 0.05$), that contained greater fiber numbers ($r_s = 0.49$, $p < 0.05$, supplementary table S2).

3.4. Properties influencing sediment stability

Partitioning the variance of the sediment stability matrix between MP, environmental and fauna matrices demonstrated that when considering the two other explanatory variable groups, faunal community dynamics explained the largest proportion of variation in sediment erodibility measures (32%, $p < 0.05$), followed by MPs (28%, $p < 0.05$) and environmental variables (7%, $p < 0.05$) (figure 1(a)). The MP matrix shared 29% of the explained variance with environmental properties, so variance partitioning (VP) was run again, first excluding MPs from the analysis and again with environmental properties excluded to evaluate the role of these respective properties.

The results were examined together with the initial full model and demonstrated that the exclusion of MP variables reduced the total explained variance from 67% to 48%; with environmental properties and fauna explaining just 35% and 13% of the variation in erodibility measures, respectively in the reduced model ($p < 0.05$; figure 1(b)). The MP data matrix therefore not only increased the total explained variance, but also altered the relative effect of faunal and environmental variables on sediment stability (figures 1(a) and (b)). Specifically, the inclusion of MPs in the model strengthened the contribution of fauna, and MPs and fauna explained 57% and 30% of the variance respectively when environmental properties were removed ($p < 0.05$, figure 1(c)). The lack of shared variance between MPs and fauna, the high shared variance between MPs and environmental variables, and the greatest total explained variance (87%) when the environmental matrix was replaced with the MP matrix in the analysis, suggests the contribution of MPs is closely associated to environmental properties such as BMA and MGS.

3.5. Effects on microplastic accumulation

VP MP data between environmental, faunal and stability matrices also helped demonstrate the impact sediment stability may have on MP accumulation in the bed. Together, these properties explained 40% of the variance in MP characteristics (figure 2(a)). Environmental properties contributed a further 18% explained variance, and while the faunal community explained 13% of the variance this matrix did not significantly contribute to the variation in MPs ($P > 0.05$, RDA test). The sequential removal of fauna (figure 2(b)) and then stability (figure 2(c)) did not improve the explained variance, but highlighted the importance of including all three explanatory matrices in the analysis.

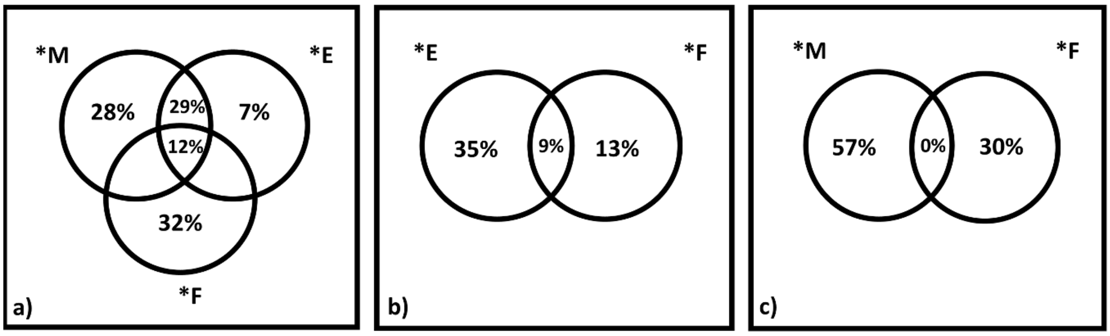

Figure 1. Venn diagrams illustrating (a) the relative contributions (% of variance explained) of environmental (E), faunal (F), and microplastics (M) matrices in explaining variation in sediment stability, and the shared components (overlaps). (b) The change in explained variance when microplastics (M) are removed from the analysis. (c) The change in explained variance when environmental variables (E) are removed from the analysis. Significance of the pure components (E), (F) and (M) were tested with partial redundancy analysis (pRDA) using 999 random permutations. Significant values ($p < 0.05$) are represented by '*'.

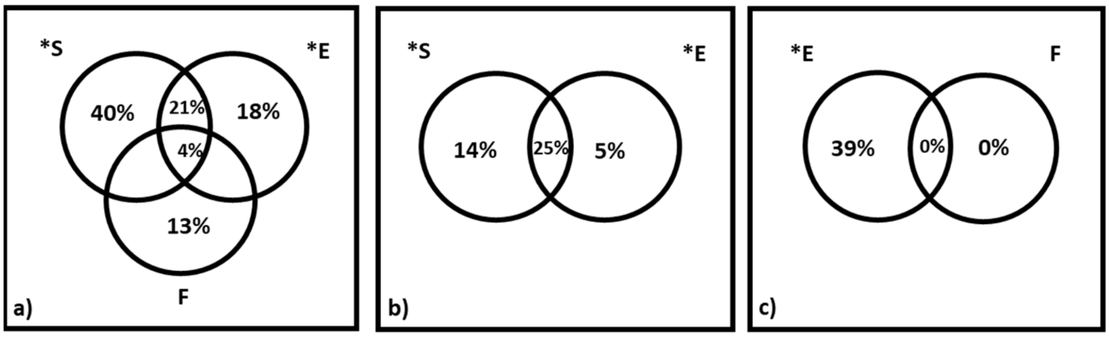

Figure 2. Venn diagrams illustrating (a) the relative contributions (% of variance explained) of environmental (E), faunal (F), and stability (S) matrices in explaining variation in MP abundance and diversity, and the shared components (overlaps). (b) The change in explained variance when fauna (F) are removed from the analysis. (c) The change in explained variance when stability variables (S) are removed from the analysis. Significance of the pure components (E), (F) and (S) were tested with partial redundancy analysis (pRDA) using 999 random permutations. Significant values ($p < 0.05$) are represented by '*'.

4. Discussion

To our knowledge, this is the first study to relate MP pollution to biostabilization and sediment stability, despite the importance of these processes for coastal sediment dynamics, habitat formation and biogeochemical cycling (Paterson *et al* 2018, Hope *et al* 2020c). The inherent complexity of sediments, and the infancy of MP field studies, renders MP effects on functions difficult to disentangle in the real world (Nava and Leoni 2021). Nonetheless, recognizing interactions and feedback processes in field situations is fundamental to understanding MP effects on functionality (Ladewig *et al* 2021) and the ecological risk they may pose. The findings provide context for a number of laboratory exposure studies that suggest MPs affect sediment ecosystem structure and function (Green *et al* 2017, Seeley *et al* 2020, Hope *et al* 2020a). By utilizing VP, the potential for MPs to affect the functional role of biota was highlighted, and the fact that MPs may interfere with processes that underpin biostabilization, as well as sediment stabilization potentially influencing MP accumulation in the bed.

These results need to be evaluated in greater detail through more in-depth laboratory analyses and long-term monitoring studies, however the relationships observed suggest there may be implications for both direct mechanistic MP effects (on organism fitness etc), and indirect cascading effects on various processes and feedbacks as MPs interact with the biological and physical components of the seabed. For example, if MPs alter biostabilization potential this may, in turn, modulate MP distribution and fate as more or less physical disturbance is required to mobilize sediments, MPs and other associated contaminants.

4.1. Influence of MPs on sediment stability

The findings suggest that even when a poor direct correlation was observed between fauna and stability, the inclusion of MP characteristics in the model, increased the variance in stability explained by fauna. This may suggest that as MPs increase, their effects on different fauna may indirectly influence functions such as sediment stability, perhaps down to their activity levels being altered, which has been seen with bivalves (Bour *et al* 2018, Hope *et al* 2020a) and worms (Green *et al* 2017) with exposure in the laboratory. The reduction in the total explained variance when the MP matrix was removed

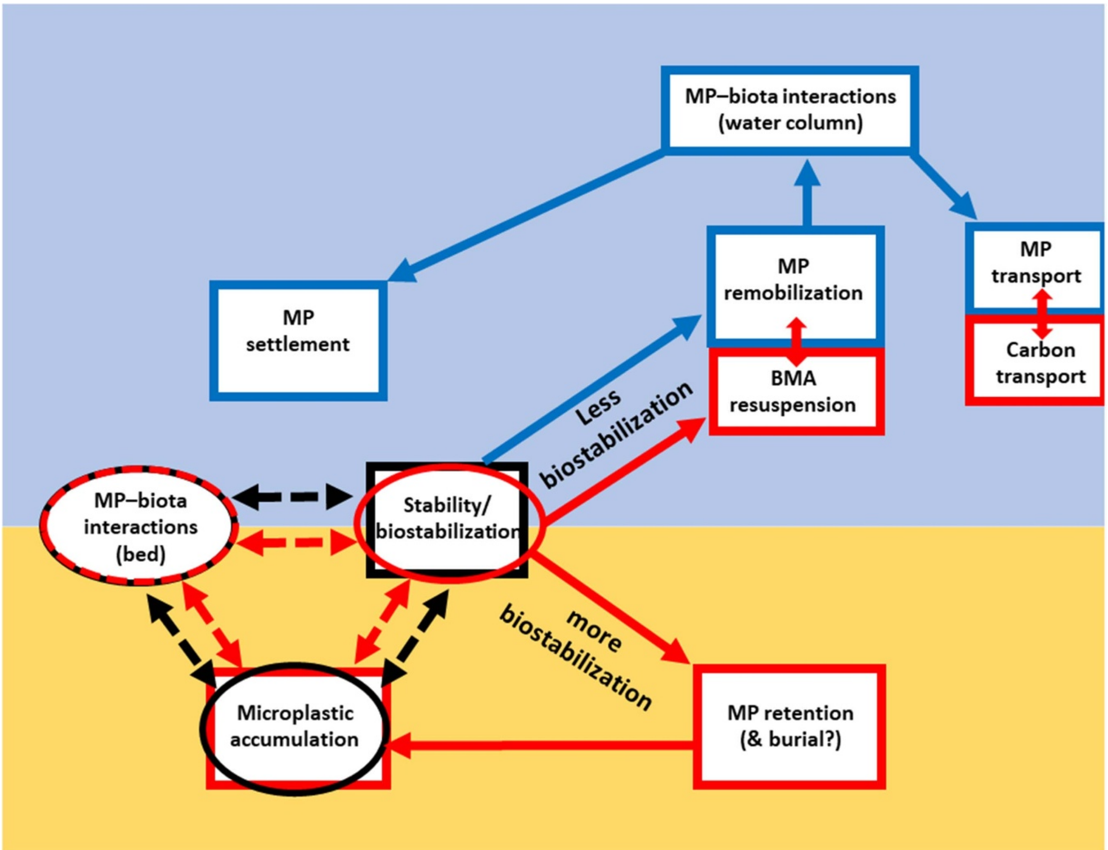

Figure 3. A conceptual diagram of the proposed interactions and feedbacks occurring in intertidal sediments. Processes and connections in blue (pelagic) were not evaluated as part of this study but closely linked to what happens on the benthos. Processes and links in red start with microplastic accumulation (square box, bottom left) with the variance partitioning analysis represented with the dashed arrows. Processes and links in black, start with the biostabilization (black square, center), which determine the MP-biota interactions and the accumulation of MPs.

from the model, but no reduction when environmental properties were removed (29% variance shared between MPs and environmental matrices), suggests MP accumulation on the bed may also mediate the influence of other environmental properties on sediment stability. This may be due to MPs modifying key biophysical properties, such as BMA biomass, faunal presence/activity and even grain size (Carson *et al* 2011, Green *et al* 2017, Hope *et al* 2020a). As these properties play key roles in sediment stability (Joensuu *et al* 2018, Paterson *et al* 2018, Hope *et al* 2020c) the multiple feedback processes surrounding MP-biota interactions (figure 3) require further quantification. Moreover, the effects of MPs on the complex interactions between benthic algae, fauna and sediment properties and their influence on nutrient dynamics and nitrogen removal processes (Schenone and Thrush 2020, Hope *et al* 2020d, Vieillard and Thrush 2021) have not been fully considered.

4.2. MP retention and remobilization

The large contribution of stability measures to MP characteristics of the sediment suggests biostabilization, promoted by BMA growth, fauna and OM content, may play key roles in MP accumulation and retention. OM at the Waitemata sites appeared to limit subsurface erosion (m_e) and was positively associated with fragment abundance. Clay and OM have previously been used as good predictors of heavy metal (Du Laing *et al* 2009) and potentially MP (Vianello *et al* 2013, Enders *et al* 2019) pollution in sediments. OM tends to coincide with greater BMA and biological cohesivity, reducing sediment erosion. Further exploration of these relationships with controlled laboratory experiments would be beneficial, as these findings suggest MPs have the potential to moderate their own transport dynamics as they influence sediment mobility; by influencing biota and lowering erosion thresholds. Comprehending benthic interactions and MP-effects on key processes is crucial for understanding MP dispersal through estuaries, as MP particles are stored and resuspended from sediments many times for decades before they become completely immobilized (Tramoy *et al* 2020). This is especially important given predictions of increased flood risk, storms and sea level rise. These climate-related changes may remobilize MP particles from riverbeds, sediments, coastal land and even landfill sites (Ockelford *et al* 2020, Roebroek *et al* 2021).

These interactions must be examined across a wider range of sediment sizes including those with higher clay/mud contents (>5%) and biological material as these muddier, sediment systems function differently (Pratt *et al* 2014, McCartain *et al* 2017).

MP aggregation with microbial biofilms in the water column is known to promote deposition (Michels *et al* 2018) and it has recently been shown that MP-algae aggregates do not disassociate when resuspended (Möhlenkamp *et al* 2018). This resuspension study however did not measure the resuspension of MP aggregates from a sediment bed, but a clean erosion chamber floor. The resuspension of MPs bound to a benthic BMA biofilm *in situ*, will therefore behave differently from clean MPs in the water, and MP-algae aggregates resuspended from a chamber floor. This study therefore emphasizes the need to consider not only biofilm growth that occurs in the water column but the resuspension of MPs associated with sediment and benthic biofilms. MP-biofilm-stability relationships on the bed will help us to understand MP transport and fate in relation to their exchange with the seabed. MPs can become part of, and are known to influence the structure of terrestrial soils (de Souza Machado *et al* 2018) with fibers in particular, getting tangled in the soil matrix (Rillig *et al* 2017). The detection of more fibers in 'coarse', mobile subsurface sediments, may be due to greater sediment permeability, with the shape of fibers not only facilitating greater penetration, as they do into glass bead and soil beds (Rillig *et al* 2017, Waldschläger and Schüttrumpf 2020) but their subsequent entanglement in the sediment bed may limit their resuspension. As microbial growth can reduce the permeability of coarse sediments (Caruso *et al* 2017), fauna can rework the sediment, and grain size can alter pore space, biological and physical properties of the bed may modulate both the infiltration of fibers into the bed and their resuspension. The dispersal and influence of MPs as they transition through estuaries will therefore not only depend on the MP size, density and shape, but also the biotic components, biofilm stickiness and the degree of incorporation into the bed/biofilm (depth of penetration). The exchange of MPs in and out of the bed warrants further investigation, and a wider range of sediment types in future studies, is necessary to comprehend the role of benthic organisms and cohesivity.

5. Conclusion

The interactions and feedbacks between MPs, biota and sediment transport dynamics mediate the transport, fate of MPs in coastal systems. This study suggests MP pollution should be considered when assessing coastal sediment stability as it can influence environmental properties of the seabed including BMA biomass and fauna that influence bed stability. MP fibers and fragments exhibited different relationships with erodibility measures and the variance in MPs explained by environmental properties and sediment stability suggests the bed characteristics may mediate the abundance, type and diversity of MPs accumulating on the benthos.

Understanding MP interactions with biota and other environmental stressors across various habitats and under different scenarios is crucial given predicted increases in coastal erosion associated with increased flooding, sea level rise and storm surges and warrants further investigation.

Data availability statement

The datasets generated for this study are available on request to the corresponding author.

The data that support the findings of this study are available upon reasonable request from the authors.

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Author contribution statement

J A H conceived the manuscript with help of G C and S F T. J A H collected, processed and analysed the data, and produced the first draft of the manuscript. All authors contributed to the ideas in the manuscript, further drafts and gave final approval for publication.

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Conflict of interest

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

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