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Salt marsh sediments act as sinks for microplastics and reveal effects of current and historical land use changes

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ABSTRACT

Microplastic particles are widespread in marine sediments and the abundance of the different types of particles vary widely. In this paper we demonstrate that salt marshes effectively capture microplastics in their sediments, and that microplastic accumulations increase with the level of urbanization of the land surrounding estuarine areas. We extracted microplastics from sediment cores in salt marshes of SE New England estuaries at different degrees of urbanization and land use intensity. Microplastics were present everywhere, but their abundances increased markedly with the degree of urbanization of the land. Microplastic fragment counts were linked to nearby urbanization and their abundances seemed to be linked to more local, within-watershed inputs. The number of fibers was similar across all sites suggesting that fiber accumulation in these sediments is likely influenced by effective long-distance transport from large-scale areas. The sedimentary record confirmed that microplastics have been accumulating in these estuaries since the early 1950s, and their abundances have increased greatly in more recent years in response to the progressive urbanization of the watersheds and intensification of land uses. Our results highlight the role of salt marsh sediments as sinks for microplastics in the marine environment.

Introduction

Since the 1950s, mass production and use of plastics has grown exponentially. In 2018, global plastic production reached nearly 360 million metric tons, and production has been increasing by more than 3% each year (Plastics Europe, 2019). Decades of increasing plastic production and use have resulted in large quantities of plastic waste entering the environment. Of the 275 million metric tons of plastic waste globally generated in 2010 alone, about 5 to 13 million metric tons entered the ocean (Jambeck et al., 2015). Plastic pollution has now become a global concern as plastic debris have reached all of the world's oceans with adverse effects on marine organisms and biodiversity as well as on human livelihoods and economy (Cózar et al., 2014; Thevenon et al., 2014).

Among the various types of plastic wastes, microplastics (particles <5 mm, GESAMP, 2016), generally resulting from the fragmentation of larger plastic debris, are highly persistent in the environment. Microplastics are widely distributed across the oceans and are accumulating at increasing rates in the marine biosphere (Andrady, 2011; Fisner et al., 2017; Hu et al., 2018; Jambeck et al., 2015; Law et al., 2010; Martí et al., 2017; Suaria et al., 2016; Yu et al., 2016). Microplastic particles accumulate in sediments in aquatic systems, and they have been found in locations that range from the deep sea (Van Cauwenberghe et al., 2013; Woodall et al., 2014; Zhang et al., 2020) to high-altitude lakes in Tibet (Zhang et al., 2019), and from Arctic (Bergmann et al., 2017; Kanhai et al., 2019) and Antarctic (Reed et al., 2018; Waller et al., 2017) regions to tropical inhabited coral islands (Patti et al., 2020) and mangroves (Martin et al., 2020; Mohamed Nor and Obbard, 2014; Zhou et al., 2020). Although the potential damage to ecosystems posed by microplastics has yet to be adequately quantified and modelled, evidence of impacts in marine food webs is accumulating (Andrady, 2011; Khalid et al., 2021).

Highly depositional estuarine habitats, such as salt marshes, are potentially more sensitive to microplastic contamination and its potential impacts. Despite the ecological relevance of estuarine habitats and the threat that microplastics pose to the provision of important ecological services, our current knowledge of microplastic accumulations in estuaries is limited, and salt marshes and other estuarine habitats remain

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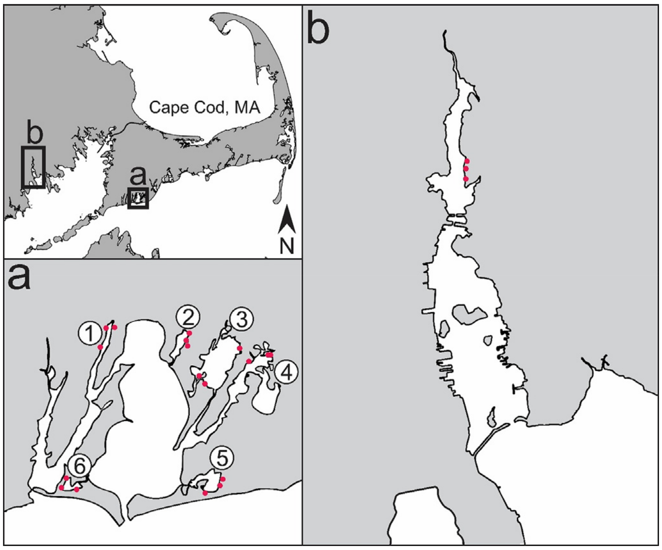

Fig. 1. A map of the selected estuaries in a) Waquoit Bay, including the Childs River (1), Quashnet River (2), Hamblin Pond (3), Jehu Pond (4), Sage Lot Pond (5), and Timm's Pond (6) estuaries, and in b) the New Bedford Harbor. Red dots indicate the approximate location of the salt marsh sediment core samples collected in each estuary.

relatively understudied compared to coastal and open marine environments. Only a relatively small number of studies have reported the presence of microplastics in salt marsh sediments (Khan and Prezant, 2018; Li et al., 2020; Willis et al., 2017). The marsh sediments offer a unique opportunity to evaluate microplastic contamination because the dense mat of grass roots and rhizomes confers substantial stability to salt marsh sediment columns, a feature that may diminish potential bioturbation and hydrodynamic sediment disturbances that commonly take place in bare sediments (Näkki et al., 2017).

In this paper we take advantage of the ability of salt marsh sediments to sequester microplastics to test the hypothesis that, as has been suggested in other coastal environments (Browne et al., 2011; Jang et al., 2020; Yao et al., 2019), increased presence and activity of people on the nearshore upland from estuaries results in larger abundances of microplastic particles of different types. To test that question we collected sediment cores from a set of SE New England estuaries that differ in intensity and history of watershed land use and population density (Fig. 1), and counted and identified microplastic particles.

Material and methods

Sediment sample collection

We collected sediment cores from salt marshes from seven SE New England estuaries subject to different degrees of urbanization (Table 1, Fig. 1). Data on the degree of watershed urban development and population density was extracted from publicly available datasets from the state of Massachusetts' GIS program (MassGIS www.mass.gov). Field work took place during summer and fall, 2019.

First, to see whether there were current differences in microplastic particle abundance and types among the salt marsh sediments adjoining the different estuaries, we collected surface (2 cm depth) sediment samples from low salt marsh habitats within the estuaries. In all locations, the dominant vegetation was the salt marsh cordgrass *Spartina alterniflora*, the most common low marsh plant species in the region. Samples were collected by inserting 9 cm diameter core liner pipes into the sediments. Three to four cores were obtained from each of the estuaries to obtain some measure of within estuary variation. All samples were collected on stable marsh platform sediments at approximately 2-3 m from the marsh edge. Samples were stored in sealed containers and refrigerated upright until analysis.

Table 1
Level of development and population density in the watersheds of the selected estuaries in SE New England. Data from 2016 obtained from MassGIS (Bureau of Geographic Information).

Estuary	Level of watershed development (%)	Population density (people km^{-2})
Waquoit Bay estuaries		
Timm's Pond	0	0
Sage Lot Pond	8	32
Jehu Pond	42	674
Quashnet River	53	300
Hamblin Pond	60	1018
Childs River	71	873
New Bedford Harbor	81	1340

Second, to test whether we could document and reconstruct decadal history of microplastic abundance sequestered in salt marsh sediments, three deeper (30 cm depth, 9 cm diameter) sediment core samples were collected from low salt marsh *S. alterniflora* areas within each of two estuaries, Childs River and Timm's Pond (Fig 1a). These two sites were selected because the adjoining land areas contrasted in the degree of land use (Bowen and Valiela, 2001; Valiela et al., 2016), with a high degree of urbanization of the land surrounding Childs River that was lacking in Timm's Pond (Table 1). Each 30 cm core was sliced into 2 cm sections and stored in sealed containers. Samples were kept refrigerated until analysis. In the laboratory, sections were dried and weighed prior to microplastic extraction.

To verify the decadal age of each section of the cores, we used existing ^{210}Pb and ^{137}Cs sediment dating and estimates of marsh accretion rates obtained in the same marshes used in this study (Gonneea et al., 2019; Kinney, 2010; Orson and Howes, 1992). Sediment accretion rates in the salt marshes of the Waquoit Bay area have been relatively spatially homogeneous, averaging 2.82 mm yr^{-1} (±0.11 mm yr^{-1} standard error). The relatively limited spatial and temporal variability of sediment dating measurements found in the cited studies allowed us to estimate, with certain level of confidence, the relative date of our core sections, at least at the decadal scale. The cores we collected provided vertical sediment profiles deep enough to capture material deposited from around the 1940 horizon, a time before the widespread use of plastics, to the present time.

Sample processing and laboratory analyses

Samples were processed using a stepwise approach that included sieving, organic material digestion, and density separation to isolate microplastics from the bulk marsh sediments and other buried marsh plant remains [modified from Masura et al., (2015)]. This methodology has been successfully used to extract microplastics in sediments of similar characteristics (Esiukova et al., 2020; Firdaus et al., 2020; Zobkov and Esiukova, 2017).

Since microplastics are present in every environment, including indoor air, proper precautions and strict contamination control measures were adopted to prevent contamination of samples, both in the field and in the laboratory (Prata et al., 2021; Wesch et al., 2017). In the field, glass, metal, wood and cardboard equipment was used whenever possible (Brander et al., 2020). All laboratory work was conducted under a vacuum hood, and any exposed samples and equipment were covered with foil to prevent contamination from airborne microplastics. All liquid reagents were passed through a 0.7 μm GF/F Whatman filter. Only natural fiber clothing and laboratory coats were worn throughout the analysis to reduce microplastic contamination from synthetic clothing (Hermsen et al., 2018; Zhao et al., 2017). Daily controls to monitor any possible contamination by airborne particles in the laboratory were made by placing a glass microfiber filter in a labelled open petri-dish.

Filters were visually checked for any deposited microplastics at the end of each day. The only items recorded in these filters were a few non-plastic clothing fibers that did not appear in our samples.

Whole dried samples (≃24 gr) were first placed in a beaker and rinsed with filtered deionized water and agitated with a metal spatula to dis-associate large clumps of sediment. The contents of the beaker were then poured through stacked sieves of 5 mm and 250 μm (Van Cauwenberghe et al., 2015). Only microplastics of sizes 5 mm–250 μm were extracted, identified and counted according to protocols developed by Lusher, et al. (2020) for studies where infrared spectroscopy or other methods to infer plastic polymer structures are limited or not readily available (Abidli et al., 2017; Chubarenko et al., 2018; Lusher et al., 2014; Martin et al., 2017; Vermaire et al., 2017; Willis et al., 2017). The size range of microplastics in this study is not subject to the technological limitations of laboratory processing and uncertainty of smaller particles (Frias et al., 2018; Frias and Nash, 2019), and is more likely to be correctly identified (visually) as plastics (Lusher et al., 2020; Primpke et al., 2020).

The contents of the sieves were rinsed with deionized water, collected in a beaker and dried in an oven at 60 °C, temperature that ensures integrity of plastic particles (Munno et al., 2018). To separate microplastic particles, organic matter, and other lighter fractions from the heavier sediments, 300 mL of zinc chloride solution (density 1.50–1.65 g mL^{-1}) was added to the dried sediments in the beaker. The density of the solution was sufficient for the recovery of the most common types of microplastics, which densities range from 0.28 g mL^{-1} for some polystyrenes to 1.47 g mL^{-1} for some PVCs (Driedger et al., 2015; Van Cauwenberghe et al., 2015). Sediments were stirred for 20 min and allowed to settle for 1h (or until the supernatant was clear of sediment). All floating solids were carefully decanted to a 250 μm sieve and then transferred to another beaker with deionized water. This process was repeated on the remaining sediments for a second time and then the decanted fraction was rinsed and dried at 60 °C.

Each dried decanted sample was placed in the beaker with Fenton's reagent (20 mL of 30% hydrogen peroxide, and 20 mL of 0.05 M iron (II) solution), and a magnetic stir bar to help digestion and removal of the natural organic matter. The catalyst solution was adjusted to pH 3.0 using concentrated sulfuric acid. The sediment solution was left at room temperature for 5 min, then placed on a bath heated up to 60 °C for 30 min. Additional 20 mL of hydrogen peroxide was added every 15 min, and stirring/heating continued until all visible organic material was digested. Fenton's reagent is an optimum protocol for extracting microplastics from complex, organic-rich, environmental matrices like estuarine sediments (Hurley et al., 2018). Remaining solids after organic matter digestion were drained through a 250 μm sieve, which was then rinsed with deionized water and transferred to a sealed glass petri-dish for later microscope analysis.

Extracted particles were placed under stereomicroscope magnification of 10X to 40X directly on the glass petri-dish. For a robust visual identification of microplastic particles, morphology (size, shape, and texture), optical (color, reflectivity) and physical properties (flexibility, density) were used as descriptive categories (Masura et al., 2015; Zhao et al., 2017; Lusher et al., 2020). Particles that did not have uniform coloration, were matt, or had cellular or organic structures were rejected. The relatively large size fraction of microplastics considered in this study (>250 μm), the protocols for organic matter digestion and density separation performed on our samples, and the consideration of the above mentioned morphological, optical and physical criteria to aid our identifications greatly reduced any possible bias associated with visual classifications (Lusher et al., 2020; Primpke et al. 2020). Each microplastic particle was classified as to type and color, and then counted and photographed. The typology of extracted microplastics was quite variable, and the particles found differed greatly in size, shape and color (Fig. S1).

In this study we reported two major types of microplastic particles: fibers and fragments. Fibers have their origin in the breakdown (shedding) of plastic-based textiles and garments, and also from abrasive action on synthetic fishing gear and marine ropes. Fragments are a type that comprises many different items that in general resulted from the fragmentation of larger plastic items including irregularly shaped plastic particles, paint chips, sheet-like plastic films, microbeads and foam particles.

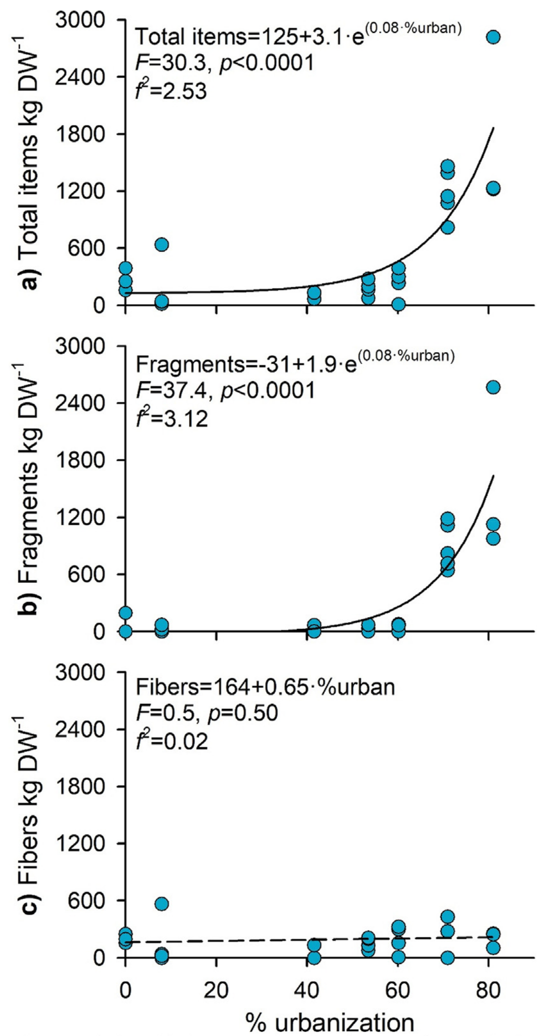

Fig. 2. Relationship between the abundances of a) total microplastic, b) fragments, and c) fibers, expressed as number of particles per kg of dry weight (DW) and degree of urbanization of the land surrounding the sampled estuaries. Dashed lines indicate non-significant regression curves. Significance of the regression curves was assessed by the use of traditional statistics and calculations of effect size (Smith, 2020).

Results and discussion

Microplastic abundances across a gradient of urban development

The abundances of total microplastic particles in surface salt marsh sediment samples increased as the degree of urban development on adjoining land increased from nil in Timm's Pond to 81% in the New Bedford Harbor (Fig. 2a), with the rise becoming evident at about 50% of urban land cover. Our results on microplastic abundances across the gradient of urbanization of the land surrounding the selected estuaries clearly confirmed the increase of microplastic contamination of estuarine sediments by the intensification of human uses on coastal watersheds, consistent with findings from other studies (Frère et al., 2017;

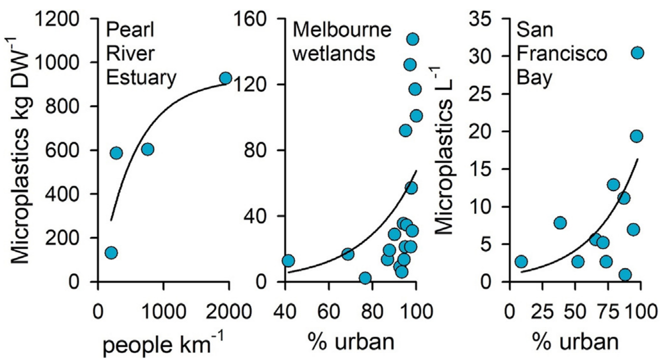

Fig. 3. Some examples of responses of microplastic abundances to variables linked to the degree of urbanization of the land, including data from the Pearl River estuary (Fan et al., 2019), estuarine wetlands in Melbourne (Townsend et al., 2019), and the San Francisco Bay area (Sutton et al., 2019).

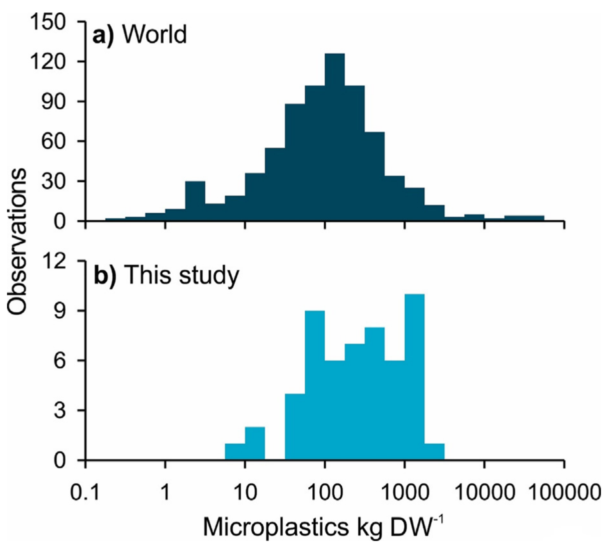

Fig. 4. Frequency distribution of the number of microplastic particles per kg^{-1} of surface dry marine sediments collected from different areas a) around the world, and b) in this study. Data in a) compiled from references included in the supplementary materials.

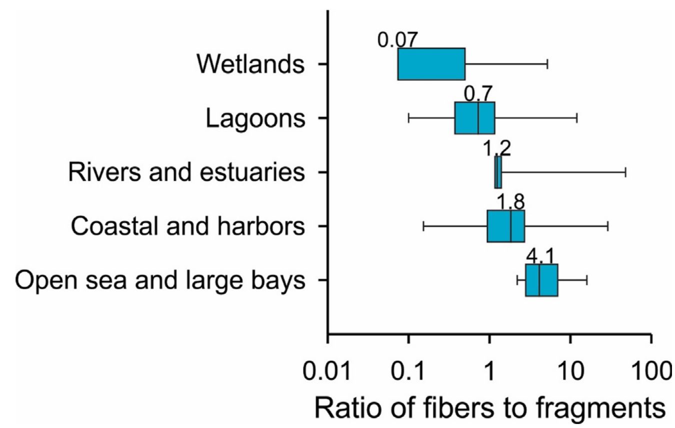

Fig. 5. Ratio of abundance of fibers to fragments in a set of marine environments that can be taken as a proxy for approximate distances away from human populations and activities. Data from (Abidli et al., 2018; Claessens et al., 2011; Fischer et al., 2015; Kane and Clare, 2019; Lozoya et al., 2016; Martin et al., 2017; Naji et al., 2017; Sathish et al., 2019; Simon-Sánchez et al., 2019; Townsend et al., 2019; Tsang et al., 2017; Van Cauwenberghe et al., 2013; Vianello et al., 2013; Wen et al., 2018; Willis et al., 2017; Woodall et al., 2014; Yona et al., 2019; Zheng et al., 2019; Zobkov and Esiukova, 2017). The median for beaches was 8.08, a larger value than for other environments. Data for beaches were not included in this figure since we could not identify beaches close or far from human centers.

Huang et al., 2020; Naidoo et al., 2015; Tsang et al., 2017; Vianello et al., 2013).

The exponential responses of microplastic abundance to urbanization we observed in our data seems to be the norm in other estuarine areas around the world (Fig. 3). The specific shape and the magnitude of the responses differ across different sites (Fig. 3), probably as a result of contrasting transport processes, depositional and sedimentological factors, as well as differences in plastic availability, use patterns, effectiveness of disposal, re-use, and recycling.

The abundances of microplastic in marine sediments across the world seem to be highly variable, ranging from none to many thousands of particles per kg of sediment (Fig. 4a). In our samples, microplastic abundances also varied widely among the different sampled locations, and their numbers covered 85% of the range of microplastic abundances found in sediments around the world (Fig. 4b). Although variable, the values we report fall within the range of microplastic abundances found in other marsh studies (Khan and Prezant, 2018; Li et al., 2020; Willis et al., 2017), demonstrating that salt marshes efficiently sequester microplastics in their sediments.

With respect to microplastic particle types, the distribution of fragments and fibers across the urbanization gradient differed (Fig. 2b and 2c). Fragments made up the major portion of microplastic particles in near-surface sediments, and their response to urbanization paralleled that of total particles (Fig. 2b). Abundance of fibers did not respond to degree of near-shore land use, remaining relatively constant across the sites (Fig. 2c). These suggest that the accumulation of fragments in salt marsh sediments may have a local origin while fibers might have sources other than the immediate local land surrounding the estuaries.

To test the above conjectures, we compiled published counts of fibers and fragments from a series of marine sediments that arguably tended to be located at different distances from human land sources (wetlands>lagoons>rivers and estuaries>coasts and harbors>open sea and large bays, Fig. 5), and we calculated the ratio of fibers to fragments for each environment. The box plots of the compiled data show that the median ratio from wetlands near urban areas were lower than those from open seas and large bays (Fig. 5). A possible interpretation of this trend is that, although humans generate both types of microplastic particles, fibers travel farther than fragments, and their abundance may be determined by longer-distance transport from larger-scale areas, even involving aeolian mechanisms (Liu et al., 2019; Rezaei et al., 2019; Zhang, 2017), a result that seems to be confirmed in our observations.

Historical accumulation of microplastics in salt marsh sediments

The cores taken from salt marshes in Childs River and in Timm's Pond showed between-site variability and trends in vertical profiles (and decadal trajectory) of microplastic abundance. ANOVA showed that the microplastic abundance values did not differ significantly among the cores from each site, neither in the case of Childs River (F=0.58, p=0.63) nor in Timm's Pond (F=0.57, p=0.58). These results suggested that we can pool the individual core data to then test whether there were differences in the vertical profiles between the Childs River and Timm's Pond marshes.

The vertical profiles of total microplastic particles in salt marsh sediments of the urbanized Childs River and the unpopulated Timm's Pond showed nil to low numbers at about 20 cm deep (Fig. 6a), a depth dated to about 1950 (Gonneea et al., 2019; Kinney, 2010; Orson and Howes, 1992), but differed substantially during more recent decades. In-

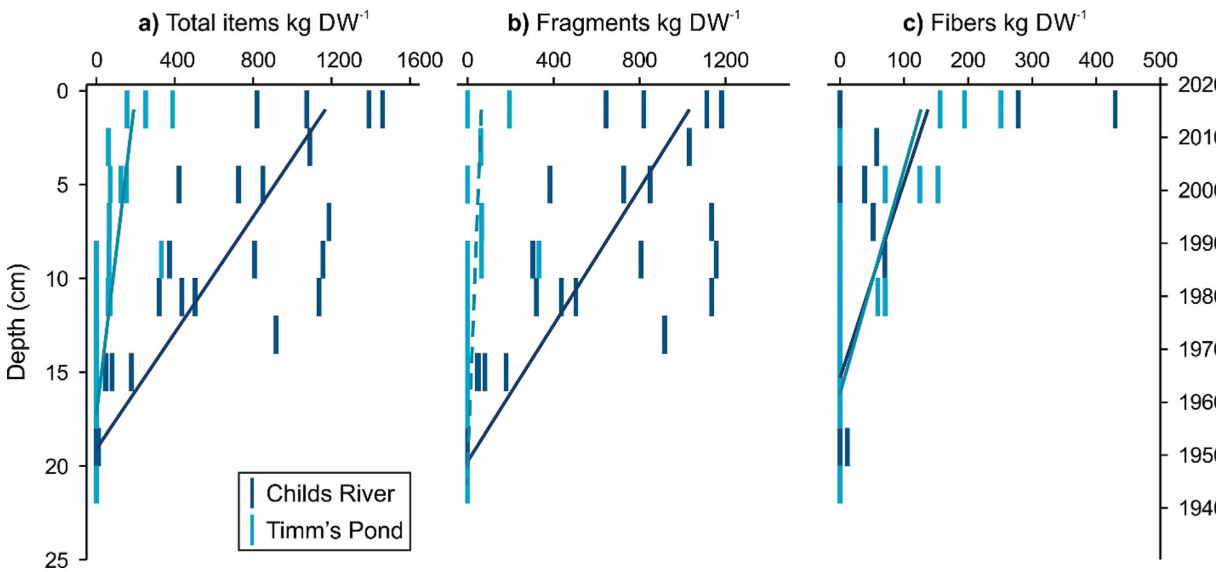

Fig. 6. Results of the microplastic counts for a) total microplastics, b) fragments and c) fibers in sediment cores collected in the Waquoit Bay estuaries of Childs River and Timm's Ponds. Dashed lines indicate non-significant regression lines. Significance of the plotted regressions was assessed by the use of traditional statistics, and calculations of effect size (Smith, 2020).

Table 2
Regression statistics and effect size analyses for data in Fig. 6.

Estuary	Particle type	*N*	Equation [*y*=number of particles, *x*=depth (cm)]	R^2	*F*	*p*	f^2 (effect size class)*
Childs River	Total	28	y=1234-65x	0.69	56.7	<0.0001	2.18 (large)
	Fragments	28	y=1087-55x	0.61	40.7	<0.0001	1.56 (large)
	Fibers	28	y=147-10x	0.34	13.3	0.0012	0.51 (medium)
Timm's Pond	Total	22	y=202-12x	0.45	16.6	0.0006	0.83 (large)
	Fragments	22	y=67-3x	0.07	1.6	0.2222	0.08 (small)
	Fibers	22	y=135-8x	0.49	19.3	0.0003	0.96 (large)

* Effect size classes based on f^2 values: large≥0.8, medium≥0.5, small<0.5 (Cohen, 1988).

creases were evident in both estuaries, but the rate at which microplastic concentrations increased differed greatly, resulting in much larger values in the more urban setting (Fig. 6a, and Table 2).

Microplastic accumulations in the Waquoit Bay estuaries started synchronically around the early 1950s. The predominantly rural landscape and the low level of development of the land in the Waquoit Bay area during the 1930s, 1940s and early 1950s (Bowen and Valiela, 2001), and only nascent plastic production and use during those early years must explain why microplastics were not present below the 1950 horizon. Human presence in much of the Waquoit Bay area increased sevenfold since the 1960s (Bowen and Valiela, 2001; Valiela et al., 2016). Parallel to the increase in population, global use of plastics grew exponentially (Plastics Europe, 2019), factors reflected in the marked increases in microplastic concentrations recorded in the marsh sediments in recent decades (Fig. 6a). Our results are consistent with literature reports of higher microplastic abundances in recent sediment layers, and nil or low abundances in layers that date back to the 1950s and 1960s (Fan et al., 2019; Li et al., 2020; Matsuguma et al., 2017; Willis et al., 2017).

Apart from changes in population and plastic use, there are other factors that may also contribute to differences in the decadal accumulation of microplastics in salt marsh sediments. One of such factors is the possible alteration of marsh sediment accretion rates caused by the recent acceleration of sea level rise (Kirwan and Temmerman, 2009). The potential role that this factor may have played in increasing recent sediment accumulation in the studied salt marshes, and therefore altering our observed microplastic densities, is relatively limited since the sediment-poor, organogenic salt marshes of Southeast New England have shown relatively minor changes in vertical growth in response to recent sea level rise rates, as revealed in recent assessments of the vulnerability of these marshes to current and future sea levels (Valiela et al., 2018; Watson et al., 2017).

The historical trajectory of accumulation of fragments (Fig. 6b) and fibers (Fig. 6c) differed substantially (Table 2). Fragments were scarce and did not increase significantly in Timm's Pond (Fig. 6b), while fragment counts rose by orders of magnitude in Childs River nearer the present (Fig. 6b). Fragment abundance thus was clearly linked to nearby, local density of people and degree of urban development on land near the receiving salt marsh. Abundance of fibers and their decadal trends did not differ between cores taken from Childs River and Timm's Pond (Fig. 6c, and Table 2). Fibers showed increasing accumulations throughout the decades (Fig. 6c, and Table 2), but their abundances were unresponsive to differences in the degree of local human presence or development. The inference from this contrasting result seems to confirm what we observed in the more recent sediments across the various sites. The accumulation of fibers, even in undeveloped areas, and owing to their different physical properties, may likely involve wind transport from larger, regional scales, while the denser, less buoyant microplastic fragments likely derive from local sources.

As observed in other similar estuarine environments, such as mangrove forests (Martin et al., 2020), the current and historical presence of plastic particles in the studied sediment profiles confirmed the role of salt marshes as a sink for microplastics in the coastal zone. Highly depositional estuarine systems efficiently sequester microplastics, and have been doing so for many decades.

Limitations

Our results confirmed the presence of microplastics in the studied salt marsh sediments and highlighted the effects of current and historical urbanization on particle accumulations. These findings are, nevertheless, restricted by some of the limitations imposed by the methodologies used in this study.

First, our data resulted from the analyses of large size (>250 μm) microplastic particles. We did not include data for the smallest microplastic sizes, data that could have potentially included particles as small as 1 μm, commonly defined as the lower size limit for microplastics (Frias and Nash, 2019). Our results are, nevertheless, comparable to other studies including similarly large particle size ranges. Despite current technological limitations of laboratory processing of particles of less than 20-100 μm in size (Frias et al., 2018), some studies confirm that small size microplastics are particularly abundant in environmental samples (Bergmann et al., 2017; Haave et al., 2019; Poulain et al., 2019; Shim and Thompson, 2015). The inclusion of these abundant small particle sizes in future studies of microplastic accumulations in salt marsh sediments will shed more light on the possible links between urbanization, transport mechanisms and differences in particle abundances and types, adding more complete information to the conclusions derived from this study.

Second, we did not perform chemical analyses of the microplastic particles found in our samples. As mentioned above, the consideration of large particles only, our protocols for sample handling and processing, and the use of standardized identification criteria greatly reduced any biases associated with our visual microplastic particle identifications (Lusher et al., 2020; Primpke et al. 2020). However, the use of analytical methods to determine both the presence of plastics and the range of polymers recovered, such as Fourier transform infrared and Raman spectroscopy, or pyrolysis and thermal desorption gas chromatography–mass spectrometry are highly recommended. These methods not only limit the possible biases associated with visual identifications, but also add relevant information on chemical composition that could be useful to assess particle behavior and transport, environmental fate, and interactions with biota (Lusher et al., 2020).

Conclusions

This report is the first to analyze microplastic concentrations in the sediments of SE New England salt marshes. Our data confirmed that microplastic particles can be found in salt marsh sediments of all estuaries we sampled across the region, and highlighted the role of salt marsh sediments as sinks for microplastics in the marine environment.

Microplastic abundances reflected the level of urbanization of the surrounding watersheds. Levels of urban development that cover more than 50% of the land have resulted in a marked increase of over an order of magnitude in sediment microplastic abundances with respect to other less populated estuaries in the area. The sedimentary record confirmed that microplastics have been accumulating in these estuaries since the early 1950s, and their abundances have increased greatly in more recent years in response to the progressive urbanization of the watersheds and intensification of land uses.

The relative abundances of microplastic fragments and fibers revealed important information about the origins and transport mechanisms of the different microplastic particles in estuarine sediments. Fragments have a more local, within-watershed origin while fiber abundances are influenced by effective long-distance transport from large-scale areas. These results have implications for management, as they provide clues on the sources and transport of microplastic particles in the environment that could help design more effective regulatory policies and strategies to reduce plastic pollution in salt marshes and other sensitive environments.

Credit author statement

Javier Lloret and Rut Pedrosa-Pamies: Conceptualization, Methodology, Investigation, Formal analysis, Visualization, Writing - Original Draft. Nicole Vandal, Ruby Rorty, Miriam Ritchie and Claire McGuire: Investigation, Writing - Review & Editing. Kelsey Chenoweth: Investigation, Formal analysis, Visualization, Writing - Review & Editing. Ivan Valiela: Formal analysis, Visualization, Writing - Review & Editing.

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.

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Supplementary materials

Supplementary material associated with this article can be found, in the online version, at doi:10.1016/j.envadv.2021.100060.

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