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Spatial and temporal variability of microplastic abundance in estuarine intertidal sediments: Implications for sampling frequency

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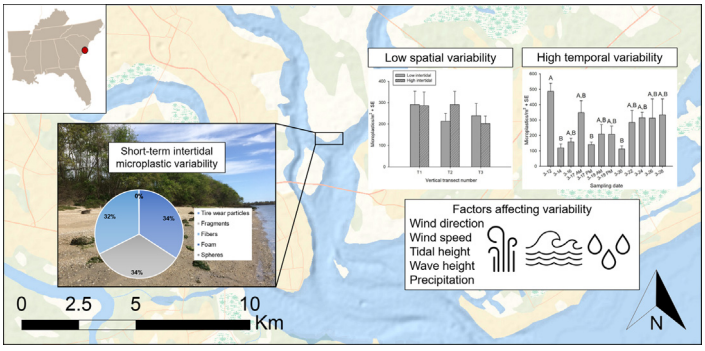
HIGHLIGHTS

- Collected intertidal microplastics from a coastal beach every 1 to 2 days for 17 days.
- Microplastic abundance varied over 20-fold during the 17-day sampling period.
- Nearly 2.5-fold difference in microplastic abundance observed over 24 h.
- Wind direction had greatest effect on microplastic abundance and variability.
- First study of short-term spatiotemporal microplastic variability in US estuary.

GRAPHICAL ABSTRACT



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ABSTRACT

Microplastics (<5 mm) are well documented across shorelines worldwide; however, high variability in microplastic abundance is often observed within and among field studies. The majority of microplastic surveys to date consist of single sampling events that do not consider spatiotemporal variability as a potential confounding factor in the interpretation of their results. Therefore, these surveys may not accurately capture or reflect levels of microplastic contamination in the environment. Here, we provide the first investigation of small-scale spatial and temporal variability of microplastic abundance, distribution, and composition in the intertidal zone of an urbanized US estuary to better understand the short-term, daily spatiotemporal variability of microplastics in dynamic coastal environments. Intertidal sediment was collected from both the low and high intertidal zones of a sandy estuarine beach located in South Carolina, southeastern US every 1 to 2 days at low tide over 17 days (12 sampling events; total n = 72). Study-wide, microplastic abundance ranged from 44 to 912 microplastics/m^2 and consisted primarily of polyethylene, nylon, polyester, and tire (or tyre) wear particles. High temporal variability was observed, with microplastic abundance differing significantly among sampling events ($p = 0.00025$), as well as among some consecutive tidal cycles occurring within 12 h of each other ($p = 0.007$). By contrast, low spatial variability was observed throughout the study with no significant differences in microplastic abundance detected between the low and high intertidal zones ($p = 0.76$). Of the environmental factors investigated, wind direction on the day of sampling had the greatest effect on temporal microplastic variability. Our results demonstrate that there can be significant temporal variability of microplastic abundance in estuarine intertidal sediments and are important for informing the methods and interpretation of future microplastic surveys in dynamic coastal environments worldwide.

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1. Introduction

In coastal communities, the mismanagement of plastic waste results in an estimated annual input of 4.8 to 12.7 million metric tons of plastic to the world's oceans (Jambeck et al., 2015). Currently, plastic waste is the most abundant ocean contaminant, constituting 80 to 85 % of marine debris (Auta et al., 2017). This plastic waste can be classified by size and includes macroplastic (>20 mm), mesoplastic (5–20 mm), and microplastic (<5 mm) (Barnes et al., 2009). The majority of plastic particles (92.4 %) in the ocean are microplastics (Eriksen et al., 2014). Microplastics are considered contaminants of emerging concern (CECs) because they occupy the same size fraction as the plankton and sediments in aquatic habitats, making them bioavailable to a wide range of organisms (Wright et al., 2013a).

Microplastics can be of primary or secondary origin and can enter the environment through both point and non-point sources. Primary microplastics are manufactured to be microscopic and are used for a variety of industrial and domestic applications including personal care products, pharmaceutical vectors, air-blasting media, and plastic pre-production pellets (Auta et al., 2017). Primary microplastics enter the environment directly through industrial outfall, municipal wastewater treatment plant effluent, and accidental spillage. In contrast, secondary microplastics result from the fragmentation of larger plastic items as they degrade (Weinstein et al., 2020). This degradation is a consequence of prolonged exposure to ultraviolet (UV) light, physical abrasion from sediment and wave action, and biological degradation (Browne et al., 2007). In this way, secondary microplastics can enter the environment indirectly as plastic litter degrades over time and constitute the majority of coastal microplastic debris (Jambeck et al., 2015; Weinstein et al., 2019). Because of these various sources of input, microplastics are a heterogeneous mixture of synthetic particles that vary in size (<5 mm), shape, color, density, and polymer composition.

These numerous sources of input have also facilitated the exponential accumulation and widespread distribution of microplastics in the coastal environment. Globally, microplastic abundance in coastal surface waters ranges from 2.8×10^{-5} microplastics/L in the Tamar Estuary, UK (Sadri and Thompson, 2014) to 30.8 ± 12.1 microplastics/L (mean ± SD) in Winyah Bay, South Carolina, US (Gray et al., 2018). In coastal intertidal sediments, abundance ranges from 13.2 ± 2.96 microplastics/m^2 (mean ± SE) in Mobile Bay, Alabama, US (Wessel et al., 2016) to 5595 ± 27,417 microplastics/m^2 (mean ± SD) in the Pearl River Estuary, Hong Kong (Fok and Cheung, 2015). Once in the environment, microplastics may be consumed intentionally (Graham and Thompson, 2009) or accidentally (de Sá et al., 2015) by aquatic organisms, leading to adverse effects such as gut blockage, internal abrasion, inflammation, and mortality (Browne et al., 2008; Détrée and Gallardo-Escárate, 2017; Gray and Weinstein, 2017; Leads et al., 2019; Von Moos et al., 2012; Lei et al., 2018; Wright et al., 2013a; Wright et al., 2013b). In addition, these particles may accumulate within marine food webs due to trophic transfer and biomagnification (Au et al., 2017). The bioavailability of microplastics to various aquatic organisms is influenced by their size, density, color, and abundance in the environment (Wright et al., 2013a). For this reason, it is important to accurately quantify and monitor the level of microplastic contamination in the environment to properly assess exposure and risk.

While microplastics have been quantified in coastal surface waters and sediments worldwide, high variability in abundance is observed within and among studies (reviewed by Akdogan and Guven, 2019 and Underwood et al., 2017). In addition, the majority of microplastic surveys to date consist of single sampling events and do not quantify spatial and temporal variability as potential confounding factors in the interpretation of their results (e.g., Kaliszewicz et al., 2020; Pan et al., 2019; Pojar et al., 2021; Xiong et al., 2019). On a large scale, some of this variability can be influenced by geographic differences in urbanization and land use, and seasonal changes in precipitation and hydrology (de Carvalho et al., 2021; McEachern et al., 2019; Quesadas-Rojas et al., 2021; Rasta et al., 2021; Tsang et al., 2020). However, few studies have investigated microplastic variability on smaller spatial and temporal scales, such as daily changes in microplastic abundance within a single sample site. Transient environmental conditions such as wave height, wind direction, tidal height, and surface currents may influence microplastic abundance, contributing to daily variability (Forsberg et al., 2020; Moreira et al., 2016). For example, in a study on Vavvaru Island in the Maldives archipelago, Imhof et al. (2017) reported a 40-fold difference in daily microplastic abundance in intertidal sediments over 7 consecutive days. Similarly, Carvalho et al. (2021) reported highly variable microplastic concentrations ranging from 0.6 ± 2.5 particles/m^2 to 1059.3 ± 1385.6 particles/m^2 over a 13-day sampling period on the island of Fernando de Noronha off the coast of Brazil. Together, these studies suggest that microplastic surveys comprised of single sampling events may not accurately represent microplastic abundance, distribution, and composition. To better understand the magnitude and drivers of microplastic variability in coastal environments, the present study investigated the small-scale spatial and temporal microplastic variability within intertidal sediments of the Charleston Harbor estuary, located in South Carolina along the southeastern coast of the US. The present study is the first to investigate this short-term, daily variability in microplastic abundance, distribution, and composition in the intertidal zone of an urbanized US estuary.

Specifically, we collected sediments from the low intertidal zone and high intertidal zone of a small sandy beach in Charleston Harbor every 1 to 2 days over a 17-day sampling period. Over this short time scale, we quantified the abundance of microplastics in sediments and analyzed changes in microplastic abundance, spatial distribution, and particle composition. We also investigated the environmental factors and physical drivers that may be influencing microplastic variability within the estuary such as tidal height, precipitation, wind speed, wind direction, and wave height. Quantifying short term microplastic variability and determining the physical drivers impacting variability is important for accurately assessing the level of microplastic contamination at a given site. This information is crucial for designing comprehensive sampling surveys in coastal environments worldwide, optimizing sampling methods, and for properly evaluating risk to coastal ecosystem health.

2. Materials and methods

2.1. Study site and sampling design

Charleston Harbor is a partially mixed coastal plain estuary located on the southeastern coast of the United States (US). This ebb-dominant estuary is an inlet of the Atlantic Ocean and experiences semidiurnal tides with a tidal range of 1.4 m (Dame et al., 2000). The estuary receives freshwater input from 3 large rivers (Ashley River, Cooper River, Wando River) and is dominated by salt marsh, mudflat, and sandy beach habitats. As a large urban estuary and the deepest port in the southeastern US, Charleston Harbor has a surrounding population of 411,406 (Charleston County; US Census Bureau, 2019) and supports numerous industries including shipping, commercial manufacturing, tourism, and recreational and commercial fishing.

Daniel Island (Fig. 1) is located in Charleston Harbor at the confluence of the Cooper and Wando rivers and has a land area of approximately 16 km^2. The island contains primarily residential properties, and the southern portion of the island (including the area sampled in the present study) was developed from dredge spoils. The Daniel Island study site consisted of a 175 m stretch of isolated sandy beach (32° 49′ 08.9″ N, 79° 54′ 58.1″ W; Fig. 1) and was accessed by boat. Sampling occurred every 1 to 2 days between 12 March (full moon) to 28 March (new moon) 2017 (12 total sampling events; Table 1). The sampling schedule was determined by the semidiurnal tidal cycle in Charleston Harbor which experiences 2 high tides and 2 low tides per day. Samples were collected at low tide ±1 h to allow for adequate sampling time while intertidal sediments were exposed, and samples were only collected during the low tides occurring within daylight hours. On days where 2 low tides occurred during daylight hours (17 March and 19 March), the beach was sampled twice (approximately 12 h apart). During each sampling event, a 100 m horizontal transect was established on the shoreline using a transect tape. Three equidistant vertical transects were then established from the water line to the most recent high tide line. To distinguish the low intertidal zone from the high intertidal zone,

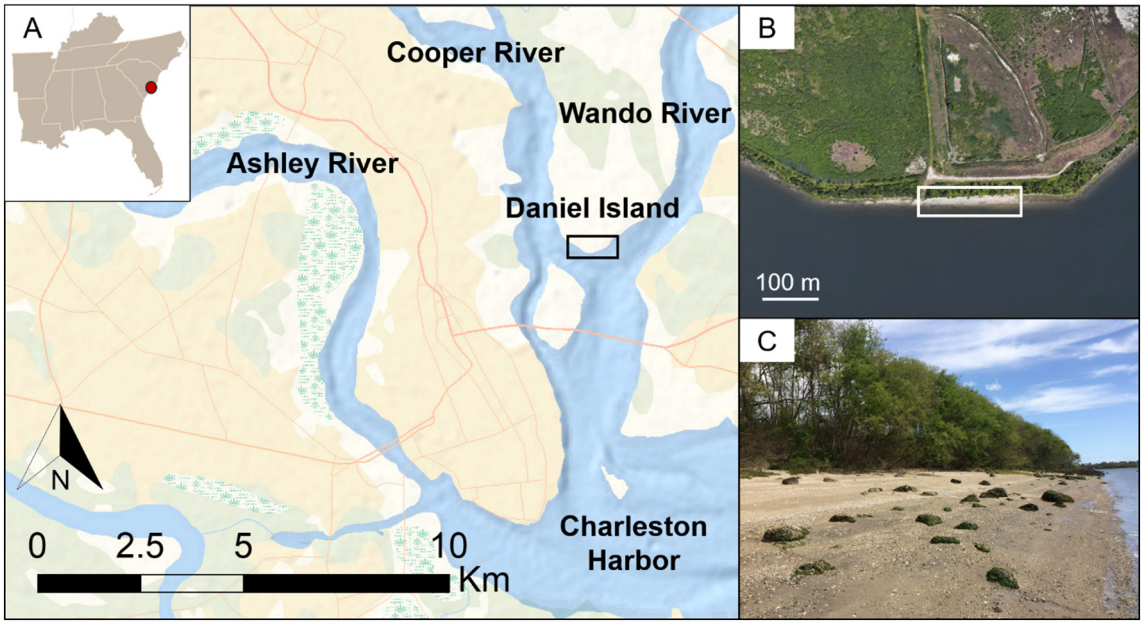

Fig. 1. A) The Daniel Island study site (32° 49′ 08.9″ N, 79° 54′ 58.1″ W; black box on map) located within the Charleston Harbor estuary (Charleston, South Carolina; inset red circle) on the southeastern coast of the United States. B) Satellite image of the study site which consisted of a 175 m stretch of isolated sandy beach (white box). C) Ground-level image of the Daniel Island study site.

the distance from the water line to the most recent high tide line was measured and divided in half. Quadrats (0.25 m × 0.25 m) were placed along the length of each vertical transect within the low intertidal zone and high intertidal zone using a random number generator. From each of the 3 vertical transects, intertidal sediment was collected from 1 quadrat within the low intertidal zone (total n = 3) and 1 quadrat within the high intertidal zone (total n = 3). Therefore, a total of 6 intertidal sediment samples were collected during each of the 12 sampling events, resulting in a study total of 72 samples collected over a 17-day period.

The top 2 cm of sediment was removed from each quadrat using a stainless-steel trowel. The sediment was then placed into a stainless-steel bucket and weighed using a portable digital scale. The wet weight (ww) of collected sediment ranged from 2.783 to 8.193 kg with a mean ± SD of 4.60 ± 1.16 kg. Microplastics were extracted from intertidal sediments on site using a sodium chloride density separation (1.16 ± 0.01 g/mL) as previously described (Leads and Weinstein, 2019). Given the density of this hypersaline solution, low-density polymers such as polyethylene, polystyrene, and polypropylene were most likely to be extracted while higher density polymers such as polyvinyl chloride and polyethylene terephthalate may have been excluded (Leads and Weinstein, 2019). Following the density separation procedure, the extraction solution was poured through stainless-steel nested sieves with standard mesh sizes of 500, 150, and 63 μm. The surface of each sieve was then rinsed into 250 mL wide-mouth amber glass jars. After transporting back to the laboratory, samples were treated with 10 mL of 30 % H_2O_2 (Fisher Chemical) and incubated at room temperature for 1 week to digest natural organic material (Nuelle et al., 2014). Prior to microplastic identification, each sample was rinsed onto a 38 μm stainless-steel sieve and poured into a clear glass crystallizing dish for examination.

2.2. Microplastic identification

Suspected microplastics were first visually identified and enumerated using a stereomicroscope according to the criteria developed by Lusher et al. (2020a). These particles were categorized by size (63–149, 150–499, ≥500 μm), particle type (fragments, fibers, foam, spheres, tire

Table 1
March 2017 sampling dates. Twelve sampling events occurred at low tide ±1 h. The time listed is the time of the low tide during which samples were collected. Samples were collected twice on days where two low tides occurred during daylight hours (17 March and 19 March) and data for those days are separated by a slash. Tidal height at low tide on the day of sampling is represented by mean lower low water (MLLW). Tidal height at the high tide prior to sampling is represented by mean higher high water (MHHW). Precipitation represents the amount of precipitation received on the day prior to sampling. Wind direction, average wind speed, and recorded high wave height are provided for the day of sampling. Wind direction is reported as the direction from which the wind is coming in degrees clockwise from north. Environmental data were retrieved from the NOAA Tides and Currents database for station 8665530 (tidesandcurrents.noaa.gov); the National Weather Service local climate and data plots for Downtown Charleston, SC (weather.gov/chs/climate); the NOAA National Data Buoy Center for station CHTS1 (ndbc.noaa.gov); and from the forecast archive SailFlow for Charleston-Edisto Buoy 41004 (sailflow.com).

Sampling date	Sampling time	Tide MLLW at sampling (m)	Tide prior MHHW (m)	Precipitation day prior (cm)	Wind direction (degT)	Average wind speed (m/s)	Wave height high (m)
12 March	14:56	−0.15	0.01	0.0	11	5.3	2.9
14 March	16:13	−0.07	−0.08	0.8382	294	3.9	2.4
16 March	17:24	0.07	−0.23	0.0	307	2.4	1.2
17 March	06:01/18:00	0.10/0.14	−0.15/−0.31	0.0	297/322	2.8	0.7
19 March	07:31/19:24	0.24/0.24	−0.25/−0.43	0.0	219/341	4.1	2.3
20 March	08:24	0.28	−0.29	0.0	338	2.0	1.7
22 March	10:19	0.24	−0.28	0.0254	202	4.5	2.0
24 March	12:04	0.07	−0.15	0.0	38	2.7	1.8
26 March	13:38	−0.13	−0.01	0.0	125	2.4	1.4
28 March	15:07	−0.27	0.05	0.0	222	3.4	1.9

wear particles), and color (black, blue, colorless, gray, green, red, white, other) (Hidalgo-Ruz et al., 2012). Tire wear particles (TWP) were included in this categorization because previous studies in the Charleston Harbor estuary have identified high proportions of these particles among the microplastic litter (Gray et al., 2018; Leads and Weinstein, 2019). Identification of TWPs was determined using the criteria reported by Leads and Weinstein (2019). Suspected microplastics were also individually evaluated to be plastic particles using the hot needle test as previously described (Barrows et al., 2017; De Witte et al., 2014; Leads and Weinstein, 2019). These particles were removed from the samples and archived in 20 mL clear glass vials for further polymer analysis.

2.3. Particle polymer identification

The polymeric composition of a subsample of isolated particles (n = 73; 1.6 % of all particles) was identified using either attenuated total reflection Fourier transform infrared spectroscopy (ATR-FTIR) using a Nicolet iS20 FTIR with OMNIC software (version 9.12) (Thermo Scientific) or Raman spectroscopy using a Xplora Plus micro-Raman spectrometer (Horiba Scientific) with LabSpec 6 software (version 6.5). Particle size and shape determined whether ATR-FTIR or micro-Raman spectroscopy was used. Particles >500 μm (excluding fibers) were analyzed using ATR-FTIR (n = 41); all others (including all fibers) were analyzed with micro-Raman. Particles for spectroscopic analysis were chosen using the following criteria: all particles >500 μm were analyzed using FTIR (n = 41) using a diamond ATR, whereas an assortment of particles <500 μm from randomly selected samples were analyzed using micro-Raman spectroscopy. Because of carbon black interference, tire wear particles were not analyzed using these methods (Leads and Weinstein, 2019).

FTIR analyses were conducted at a mid-IR range of 400–4000 cm^{-1} and a resolution of 4 cm^{-1} at a rate of 16 scans using a Diamond Smart iTX ATR accessory. FTIR spectra were compared to the spectra in the OMNICS software library set (Hummel polymer library, HR Nicolet Sampler library). Micro-Raman analyses were conducted using a 785 nm (range 50-3000 cm^{-1}) or 532 nm (range 50–3000 cm^{-1}) laser. Identification used gratings of 600 or 1200 grooves/mm, 1 or up to 10 s for acquisition time, 2, 4, 6, 8, or 10 number of accumulations. Spectra were obtained with a confocal slit width of 100 μm slit and a confocal hole diameter of 100 μm or 300 μm using a 50× LWD objective with filters ranging from 0.1 to 100 %. Particle identification from the Raman spectra were compared matched to spectral library databases (Know it All and ID Expert) (Munno et al., 2020). For both FTIR and Raman, spectral matches fell between 60 and 99 %.

Particles were classified based upon criteria previously described in Hamilton et al. (2021) and Klasios et al. (2021). Classifications included plastic (for particles that a specific plastic polymer type was identified, e.g., polypropylene, polyester, nylon), natural (for particles identified as hair, minerals, cellulose), anthropogenic unknown (refers to particles that produced a spectra that matched with a dye, but no underlying polymer was identified), anthropogenic synthetic (refers to the detection of a dye, pigment, or additive used in plastic manufacturing, but no underlying polymer was identified), and anthropogenic cellulosic (for particles where both an anthropogenic dye or pigment and cellulose were detected). Particles were classified as unknown if the spectra did not match any in the libraries.

2.4. Quality assurance/quality control

To reduce the potential for plastic contamination in samples, stainless-steel and glass equipment was used whenever possible in the field and in the laboratory. White cotton laboratory coats and nitrile gloves were also worn to process samples in the laboratory. Procedural blanks (n = 50) collected from our laboratory during the time of the current study previously determined background microplastic levels to be 0.76 ± 0.15 microplastics/blank (Leads and Weinstein, 2019). The extraction efficiency of the sodium chloride density separation used in the present study was also previously determined to be 87 % (Gray et al., 2018). The data reported herein were not blank corrected.

2.5. Statistical analyses

Microplastic abundance in intertidal sediments is reported as microplastics/m^2 and microplastics/kg wet weight (ww). Reported values represent mean ± standard error (SE) unless otherwise noted. Statistical analyses of microplastic abundance per unit area (microplastics/m^2) reported herein were corroborated by microplastic abundance normalized by weight (microplastics/kg ww). For each analysis, data were $\log_{10}$- or $\log_{10}$ (x + 1)-transformed to meet the assumptions of normality and homogeneity of variance. To assess microplastic composition, differences in the total number of microplastics collected in each particle type and size fraction were analyzed using one-way analyses of variance (ANOVA). Tukey's Honest Significant Difference (HSD) post-hoc tests were conducted for pairwise comparisons. In addition, differences in microplastic particle and size composition among the 12 sampling events were determined using two-way repeated measures ANOVA. To assess spatial distribution, differences in total microplastic abundance between the low and high intertidal zones and among transects were analyzed using a two-way ANOVA. In addition, differences in microplastic abundance between the low and high intertidal zones among sampling events were determined using a two-way repeated measures ANOVA. To assess temporal variability, differences in total microplastic abundance among sampling events was analyzed using a one-way repeated measures ANOVA. Total microplastic abundance among consecutive low tide sampling events on 16 to 17 March and 19 to 20 March were further analyzed using one-way repeated measures ANOVA. Pairwise differences were assessed using least square mean comparisons with Bonferroni adjustment.

To determine the environmental factors potentially contributing to microplastic variability, we analyzed the effect of tidal height, precipitation, wind speed, prevailing wind direction, and wave height on microplastic abundance among sampling events using linear regression analyses (Table 1 and Table S1). Tide prediction data for each sampling date was retrieved from the National Oceanic and Atmospheric Administration (NOAA) Tides and Currents database (tidesandcurrents.noaa.gov; National Oceanic and Atmospheric Administration, 2017a) for station 8665530. Tidal height is reported for mean lower low water (MLLW) at the time of sampling and for mean higher high water (MHHW) at the high tide preceding sampling (Table 1). Precipitation data were retrieved from the National Weather Service local climate and data plots for downtown Charleston, SC (weather.gov/chs/climate; National Weather Service, 2017) (Table 1 and Table S1). Precipitation levels on the day prior to sampling are listed in Table 1 and precipitation levels on the day of sampling are listed in Table S1. Wind direction, wind speed, and wave height were retrieved from NOAA's National Data Buoy Center for station CHTS1 (ndbc.noaa.gov; National Oceanic and Atmospheric Administration, 2017b) and from the forecast archive SailFlow (sailflow.com; SailFlow, 2017) for Charleston-Edisto Buoy 41004. Wind direction (degT) is reported as the direction from which the wind is coming in degrees (0–360°) clockwise from north, with 0° indicating calm wind conditions and 360° indicating north (toward the Daniel Island study site). Wind direction, average wind speed, recorded high wind speed, recorded low wind speed, wind gust, recorded high wave height, and recorded low wave height are reported for the day of sampling and are provided in Table 1 and Table S1. For these analyses, microplastic abundance was $\log_{10}$-transformed to meet the assumptions of normality and homogeneity of variance. The association of each environmental variable with microplastic abundance across sampling events was analyzed using linear regression. In addition, the combined effect of environmental variables most associated with microplastic abundance was assessed using multiple linear regression analysis and Akaike Information Criterion (AIC). All statistical analyses were conducted in R (RStudio, 1.4.1717).

3. Results

3.1. Microplastic composition

Over the course of the study, a total of 4515 particles were identified and collected from intertidal sediments. Microplastics were present in

every sample collected with concentrations ranging from 44 to 912 microplastics/m^2 (2.2 to 65.0 microplastics/kg ww). Study-wide, the number of microplastics collected within each particle type category differed significantly (one-way ANOVA, df = 4, F = 213.6, p < 0.00001; Fig. 2A). The most abundant type of microplastic collected was TWPs which constituted 34.2 % of total microplastics. Fragments constituted 33.4 % of total microplastics collected. These fragments were a variety of colors, including blue, green, gray, red, black, and white. Fibers constituted 32.3 % of collected microplastics, the majority of which were blue, black, and colorless. Foam constituted 0.1 % of collected microplastics, the majority of which were white. No microplastic spheres were collected in the present study. Among the 12 sampling events, the number of microplastics in each particle type category varied significantly (repeated measures ANOVA; effect of particle type category: df = 4, F = 220.1, p < 0.00001; effect of sampling event: df = 11, F = 4.0, p < 0.00001; Fig. 2B).

In addition, the majority (50.3 %) of microplastics were collected in the 63 to 149 μm size fraction. The 150 to 499 μm and ≥500 μm size fractions constituted 34.7 % and 15.0 %, respectively, of total microplastics collected. Accordingly, significantly more microplastics were collected in the 63 to 149 μm and 150 to 499 μm size fractions than in the ≥500 μm size fraction (one-way ANOVA, df = 2, F = 39.9, p < 0.00001; Fig. 2C). While these smaller size fractions were more abundant study-wide, the number of microplastics within each size fraction differed significantly across the 12 sampling events (repeated measures ANOVA; effect of size fraction: df = 2, F = 52.6, p < 0.00001; effect of sampling event: df = 11, F = 5.3, p < 0.00001; Fig. 2D). Together, these data show that microplastic shape and size composition significantly varied among sampling events over the 17-day sampling period.

3.2. Particle polymer identification

Polymeric analysis indicated anthropogenic particles (including plastics) were successfully identified 80.8 % across all types excluding one particle (1.4 %) which we were not able to chemically identify (classified as unknown). Plastic particles represented 74.0 % of the identified particles, while anthropogenic synthetic represented 5.5 % of the particles. A small percentage of particles were classified as anthropogenic unknown (2.7 %), while no particles were classified as anthropogenic cellulosic. Commonly identified plastic polymers included high density polyethylene (44.4 %), nylon and nylon copolymers (20.4 %), polyethylene (16.7 %), and polyester (5.6 %). Commonly identified anthropogenic dyes and plastic additives included pyrocatechol violet, Clearstrength, and Fumetrol. Natural particles comprised 16.4 % of our total analyzed particles, of which 9 of the 12 particles (75.0 %) were white fragments composed of sodium

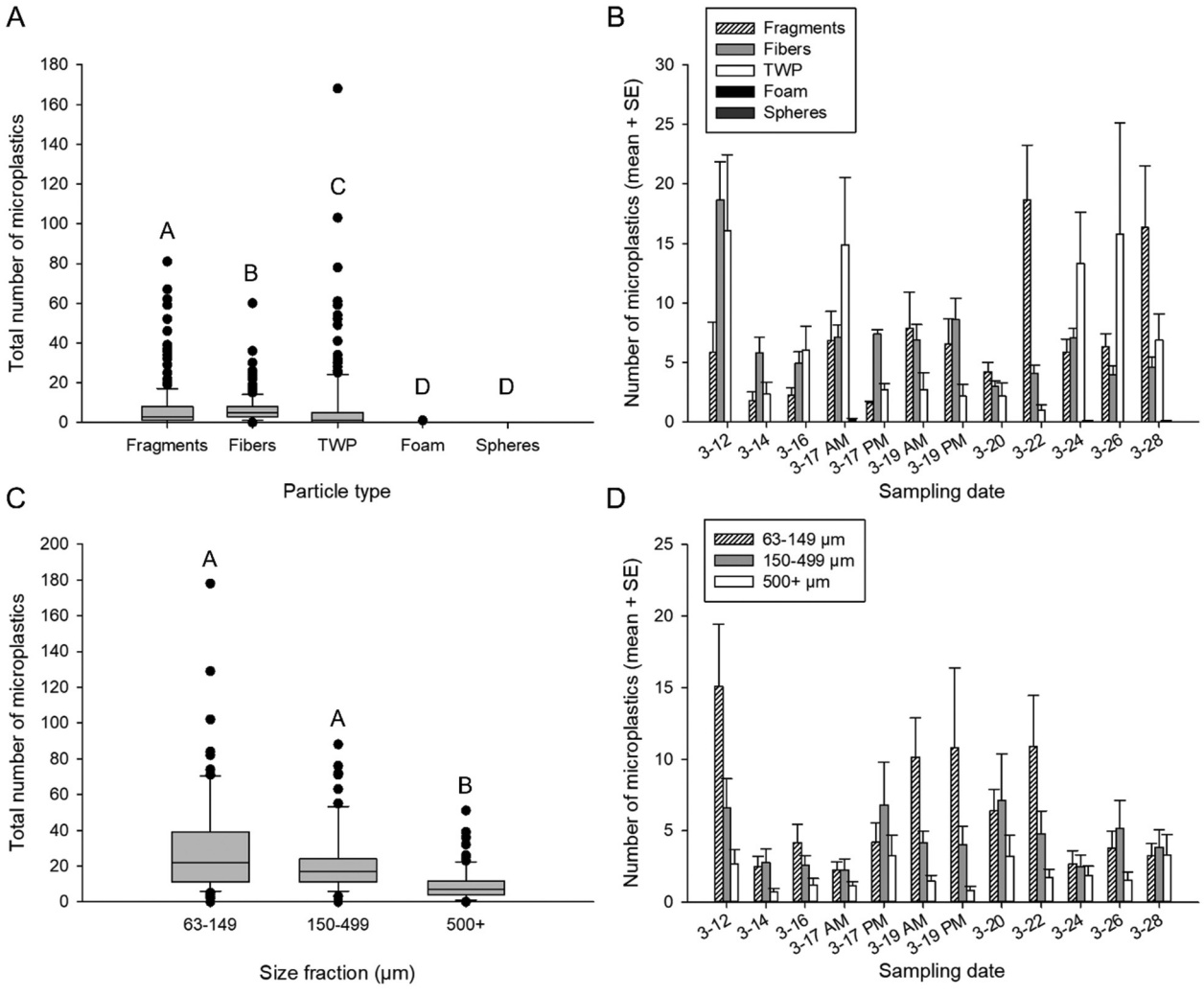

Fig. 2. Distribution of microplastic particle types (A–B) and size fractions (C–D) collected study-wide (A and C) and across sampling events (B and D). The number of microplastics collected within each particle type category differed significantly study-wide (A) (one-way ANOVA, p < 0.00001) and across sampling events (B) (repeated measures ANOVA, p < 0.00001). The number of microplastics collected within each size fraction differed significantly study-wide (C) (one-way ANOVA, p < 0.00001) and across sampling events (D) (repeated measures ANOVA, p < 0.00001). In panels A and C, different letters indicate significant differences. Pairwise comparisons are not indicated on panels B and D.

chloride or carbonate composite material. None of the fibers were classified as anthropogenic cellulosic and only one fiber as natural (wool), underscoring the value of the hot needle test in initially distinguishing fibers composed of synthetic materials from those composed of cellulose.

3.3. Spatial distribution

Study-wide, pooled microplastic abundance did not differ significantly between the low intertidal zone and the high intertidal zone, nor among transects across the length of the beach (two-way ANOVA; effect of intertidal height: df = 1, F = 0.097, p = 0.76; effect of transect: df = 2, F = 1.00, p = 0.37; Fig. 3A). This uniformity in microplastic distribution was also observed when analyzed over time. Over the 17-day period, microplastic abundance did not differ significantly between the low and high intertidal zones but did differ significantly among sampling events (repeated measures ANOVA; effect of intertidal height: df = 1, F = 0.38, p = 0.55; effect of sampling event: df = 11, F = 3.58, p = 0.006; Fig. 3B). These results indicate that within each sampling event, microplastics were uniformly distributed throughout the intertidal zone and along the length of the beach; however, the total abundance of microplastics varied significantly among these sampling events.

3.4. Temporal variability

Microplastic abundance did not vary significantly with spatial distribution. Therefore, data within intertidal zones and transects were pooled for further temporal analyses. Total microplastic abundance differed significantly among sampling events (repeated measured ANOVA, df = 11, F = 4.03, p = 0.0002; Fig. 4A). Among the 12 sampling events, average microplastic abundance ranged from 112.7 ± 19.9 microplastics/m^2 on 20 March to 486.7 ± 53.6 microplastics/m^2 on 12 March (6.0 ± 0.8 to 23.5 ± 7.4 microplastics/kg ww). These data show that microplastic abundance varied by an over 20-fold difference during the 17-day sampling period. In addition, we observed over a 4-fold difference in microplastic abundance between the first 2 sampling events on 12 March and 14 March (Fig. 4A).

Similarly, microplastic abundance varied significantly among some consecutive low tides (~12 h apart; Table 1 and Fig. 4B). Between 16 and 17 March, average microplastic abundance differed significantly across three consecutive low tide sampling events (repeated measures ANOVA, df = 2, F = 7.2, p = 0.008; Fig. 4B). During this ~24 h period, average microplastic abundance ranged from a high of 348.7 ± 77.3 microplastics/m^2 (19.6 ± 5.1 microplastics/kg ww) on the morning (AM) of 17 March to a low of 140.8 ± 17.7 microplastics/m^2 (7.3 ± 1.2 microplastics/kg ww) on the evening (PM) of 17 March (Fig. 4B). This represents a nearly 2.5-fold difference in microplastic abundance between these consecutive low tides. By contrast, microplastic abundance did not significantly differ across the three consecutive low tide sampling events occurring between 19 and 20 March (repeated measures ANOVA, df = 2, F = 1.4, p = 0.29; Fig. 4B). Within this time period, microplastic abundance ranged from a high of 210.0 ± 61.3 microplastics/m^2 (12.4 ± 2.8 microplastics/kg ww) on the morning (AM) of 19 March to a low of 112.7 ± 19.9 microplastics/m^2 (6.0 ± 0.8 microplastics/kg ww) on the morning (AM) of 20 March. While not statistically significant in this instance, this range represents a 1.9-fold difference in microplastic abundance among consecutive tidal cycles. Together, these data show the high potential for variation in microplastic concentration over very short timescales (~12 h).

3.5. Environmental factors

To better understand how environmental factors contribute to microplastic variability, we analyzed the effect of tidal height, precipitation, wind speed, wind direction, and wave height on microplastic abundance among sampling events (Table 1 and Table S1). In addition, we analyzed the effect of these environmental factors both on the day of sampling and the day before sampling to determine the timeframe in which weather conditions may most influence microplastic abundance (Table 1 and Table S1). Throughout the study, tidal height ranged from −0.27 to 0.28 m mean lower low water (MLLW) and − 0.43 to 0.05 mean higher high water (MHHW) (Table 1). Precipitation was minimal for the local area over the 17-day period with 6 rain events ranging from 0.025 to 1.27 cm, with an average of 0.14 cm (Table 1 and Table S1). Wind direction varied throughout the study, and average wind speed ranged from 2.0 to 5.3 m/s (Table 1). Wave height ranged from 0.4 to 2.9 m (Table 1 and Table S1).

Wind direction on the day of sampling had the greatest individual effect on microplastic abundance across the 12 sampling events (F = 21.8, p < 0.0001, adjusted r^2 = 0.23; Fig. 5). Specifically, lower microplastic abundance was generally observed on sampling dates where the prevailing

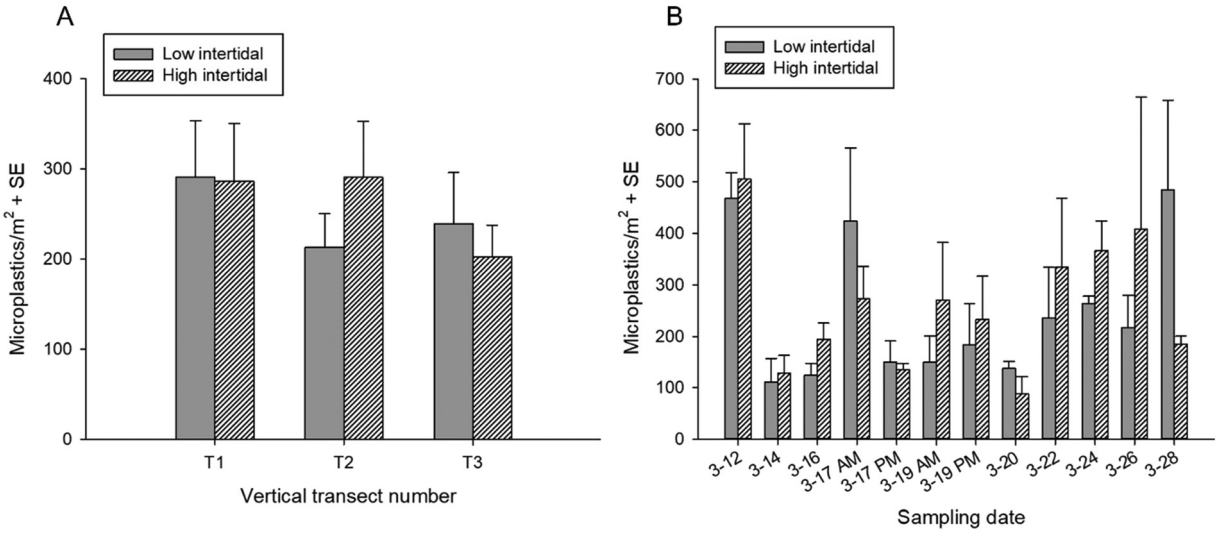

Fig. 3. Distribution of microplastics/m^2 (mean + SE) collected from the low intertidal zone (gray bars) and high intertidal zone (striped bars) across transects (A) and among sampling events (B). A) Microplastic abundance did not differ significantly between the low intertidal zone and the high intertidal zone, nor among transects (two-way ANOVA, p > 0.05). T1 to T3 represent transects 1 through 3 distributed across the length of the beach. B) Microplastic abundance did not differ significantly between the low intertidal zone and the high intertidal zone but did differ significantly among sampling events (repeated measures ANOVA, effect of intertidal height: p = 0.55, effect of sampling event: p = 0.006).

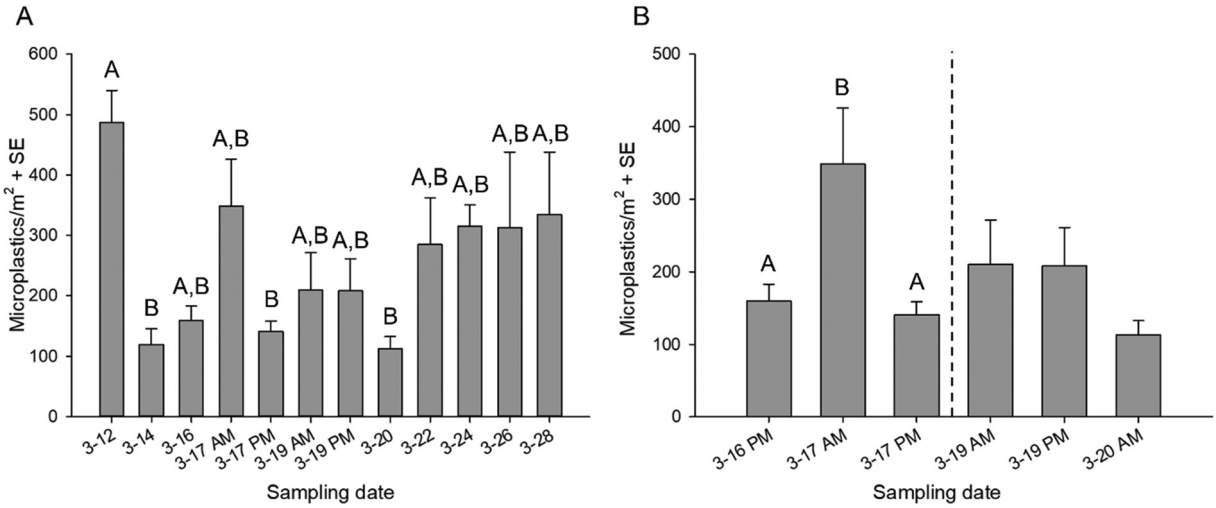

Fig. 4. Microplastic abundance (mean + SE) across all sampling events (A) and among consecutive low tides (B). A) Microplastic abundance differed significantly among sampling events (repeated measured ANOVA, $p = 0.0002$). B) Microplastic abundance differed significantly across three consecutive low tide sampling events between 16 and 17 March (repeated measures ANOVA, $p = 0.008$). Microplastic abundance did not differ significantly across the three consecutive low tide sampling events between 19 and 20 March (repeated measures ANOVA, $p = 0.29$). Consecutive tidal cycles are separated by the dashed line.

wind was oriented south away from Daniel Island (Fig. 5 and Table 1). Accordingly, wind direction accounted for 23 % of the variation in microplastic abundance. Using multiple linear regression and AIC, the combination of environmental factors most contributing to microplastic variation included wind direction on the day of sampling, MHHW levels at the high tide preceding sampling, precipitation levels the day before sampling, average wind speed on the day of sampling, and high wave height on the day of sampling ($F = 7.2$, $p < 0.0001$, adjusted $r^2 = 0.31$; Table 1). Together, these environmental factors accounted for 31 % of the variation in microplastic abundance among sampling events.

When analyzed individually, environmental factors associated with microplastic abundance included tide levels (MLLW at sampling: $F = 5.5$, $p = 0.02$, adjusted $r^2 = 0.06$; MHHW at high tide preceding sampling: $F = 7.7$, $p = 0.007$, adjusted $r^2 = 0.09$); precipitation levels the day before sampling ($F = 6.5$, $p = 0.01$, adjusted $r^2 = 0.07$); and average wind speed on the day of sampling ($F = 4.1$, $p = 0.04$, adjusted $r^2 = 0.04$) (Table 1). However, the low r^2 values of these associations indicate that these environmental factors individually only accounted for 4 to 9 % of variation in microplastic abundance. The following environmental conditions on the day of sampling were not associated with microplastic abundance: precipitation on the day of sampling ($F = 0.18$, $p = 0.67$, adjusted $r^2 = -0.01$); high and low wind speeds (high: $F = 0.9$, $p = 0.34$, adjusted $r^2 = -0.001$; low: $F = 2.2$, $p = 0.14$, adjusted $r^2 = 0.02$); wind gust ($F = 0.73$, $p = 0.4$, adjusted $r^2 = -0.004$); and wave height (high height: $F = 0.67$, $p = 0.41$, adjusted $r^2 = -0.005$; low height: $F = 0.67$, $p = 0.41$, adjusted $r^2 = -0.004$) (Table 1 and Table S1). Together, these results suggest that wind direction was the greatest contributing factor to microplastic abundance among sampling events, and that up to 31 % of variation in microplastic abundance was influenced by the combination of wind direction, MHHW levels at the high tide preceding sampling, precipitation levels the day before sampling, average wind speed on the day of sampling, and high wave height on the day of sampling.

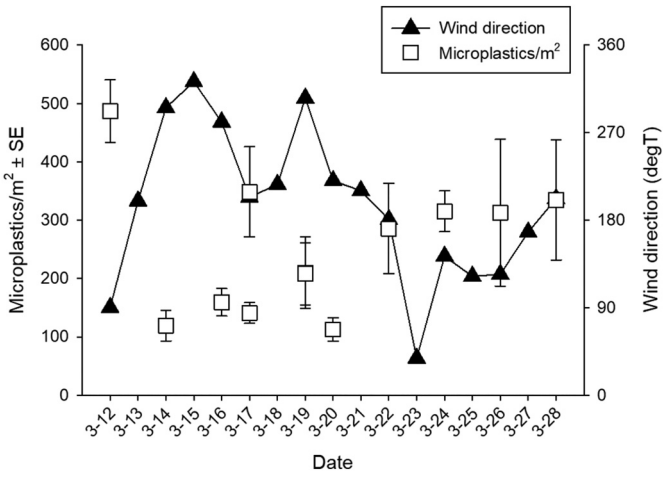

Fig. 5. Microplastic abundance (mean ± SE) (left Y axis; white squares) collected from each sampling event and prevailing wind direction (right Y axis; black triangles) over the 17-day sampling period. Wind direction (degT) is reported as the direction from which the wind is coming in degrees (0–360°) clockwise from north, with 0° indicating calm wind conditions and 360° indicating north (toward the Daniel Island study site). Wind direction on the day of sampling had the greatest individual effect on microplastic abundance across the 12 sampling events (linear regression, $F = 21.8$, $p < 0.0001$, adjusted $r^2 = 0.23$) with generally lower microplastic abundance when the wind was oriented away from the study site.

4. Discussion

The present study investigated the short-term spatial and temporal variability in microplastic abundance and composition within intertidal sediments of an urban estuary in the southeastern US (Charleston Harbor, SC) to better understand the magnitude and drivers of microplastic variability in coastal environments. Here, we show that microplastic abundance in intertidal sediments was highly variable within very short timescales ranging from 12 to 24 h (Fig. 4). In addition, microplastic composition varied among sampling events with daily changes in predominant particle types and size fractions (Fig. 2). Within each sampling event, however, spatial variability was low and microplastics were similarly distributed within the low and high intertidal zones and along the length of the beach (Fig. 3). Wind direction was the greatest individual contributor of microplastic variability, accounting for 23 % of variation in microplastic abundance (Fig. 5). These results suggest that single sampling events may not accurately reflect the level of microplastic contamination in the environment or the composition of predominant microplastics, and that environmental conditions can significantly impact survey results and interpretation.

The value of conducting long-term temporal studies with repeated sampling efforts have been demonstrated in previous studies. For example, in a year-long study of microplastics in subsurface water and intertidal sediment of the Río de la Plata estuary, Argentina, Pazos et al. (2021) determined that

microplastic abundance ranged from 23 to 613 microplastics/m^2 in intertidal sediments, with fibers being the dominant microplastic particle type throughout the year. Similarly, our previous work in the Charleston Harbor estuary spanning over four years shows how repeated sampling provides insight into the temporal variability of microplastic contamination. For example, Gray et al. (2018) documented an average concentration of 1195.7 ± 193.9 microplastics/m^2 at Daniel Island with more particles in the 150 to 499 μm size fraction. Fragments (76.2 %) and foam (18.9 %) were the dominant particles (Gray et al., 2018). In a follow-up study of the same site two years later, Leads and Weinstein (2019) reported an average of 226.3 ± 45.24 microplastics/m^2 with more particles in the 63 to 149 μm range. Fibers (55.0 %) were the dominant particle (Leads and Weinstein, 2019). In the present study, microplastic concentrations at Daniel Island ranged from 112.7 ± 19.9 to 486.7 ± 53.6 microplastics/m^2 over the 17-day sampling period. In addition, we observed a >4-fold decrease in microplastic abundance between the first two sampling events on 12 and 14 March. This decrease was not due to the removal of microplastics from the top layer of sediment on the first day of sample collection because samples were collected from different randomly-selected quadrats each day. These results suggest that microplastic abundance and composition can vary widely over short timescales, and long-term monitoring is necessary to provide a broader understanding of microplastic contamination at a given location.

Previous studies have determined that the bioavailability and toxicity of microplastics to aquatic organisms is a function of concentration, particle size, particle shape, and polymer type (Gray and Weinstein, 2017; Lozano-Hernández et al., 2021; Qiao et al., 2019). For example, in the estuarine daggerblade grass shrimp (*Palaemon pugio*), microplastic fibers were significantly more toxic than spheres or fragments, and particles measuring 93 μm exerted greater toxicity than 11 other size fractions tested (Gray and Weinstein, 2017). Similarly, ingested fibers have been shown to be more toxic and induce greater inflammation than fragments or spheres in adult zebrafish (*Danio rerio*) (Qiao et al., 2019). In addition, Zimmermann et al. (2020) showed that chronic exposure to polyvinyl chloride significantly impaired *Daphnia magna* reproduction compared to polyurethane or polylactic acid. For these reasons, having an accurate understanding of the concentration and composition of microplastics in the environment at a given location is important for properly assessing exposure, bioavailability, toxicity, and risk. Over the course of the present study, predominant particle types were collected in relatively similar proportions. Tire wear particles (TWPs) constituted 34.2 % of total microplastics, fragments constituted 33.4 %, and fibers constituted 32.3 %. However, the proportion of microplastics in each particle type varied significantly among individual sampling events (Fig. 2A and B). Similarly, while microplastics within the 63 to 149 and 150 to 500 μm size fractions were more abundant study-wide, the number of microplastics within each size fraction differed significantly across the 12 sampling events (Fig. 2C and D). These results are similar to those of Imhof et al. (2017) who reported daily changes in the concentration and proportion of larger plastic size fractions (1–5 mm and >5 mm) collected from intertidal beach sediments on Vavvaru Island in the Maldives archipelago. To our knowledge, however, the present study is the first to quantify the short-term variability of microplastic shapes and sizes at such a fine temporal scale. Together, these results show that in addition to daily changes in microplastic concentration, dynamic coastal environments can also experience daily changes in predominant microplastic particle types and size fractions. Accordingly, one-time sampling events may not provide an accurate representation of the level or type of microplastic contamination and associated environmental risk.

In the present study, the predominant polymer types identified were tire wear, high density polyethylene, nylon and nylon copolymers, polyethylene, and polyester. Similarly, other microplastic studies conducted in Charleston Harbor have reported polymers consisting of nylon, polyester, polyethylene, polypropylene, polystyrene, polyvinyl alcohol, and tire wear (Gray et al., 2018; Leads and Weinstein, 2019; Payton et al., 2020). These previous studies in Charleston Harbor have addressed the sources and emissions of predominant polymers and other microplastics to the estuary. For example, Conley et al. (2019) determined that wastewater treatment plants (WWTPs) emit approximately 500 million to 1 billion microplastics per day to the Charleston Harbor estuary, the majority (>75 %) of which are fibers. While this point source effluent is a contributor of microplastics to the harbor, the majority of microplastics in Charleston Harbor are secondary microplastics from non-point sources released primarily through the degradation of larger plastic debris (Conley et al., 2019; Gray et al., 2018; Kell, 2020; Leads and Weinstein, 2019; Wertz, 2015). For example, Wertz (2015) determined that over 7.5 tons of macroplastic litter are present on the shorelines of Charleston Harbor, consisting mostly of polyethylene terephthalate, high density polyethylene, polypropylene, and polystyrene. This debris is primarily associated with recreational land and water use of the harbor, and previous research in our laboratory determined that these items can begin degrading and emitting microplastic fragments into the environment in as little as 4 to 8 weeks (Weinstein et al., 2016; Weinstein et al., 2020). In the present study, we included TWPs in our analysis because previous research in Charleston Harbor has identified relatively high proportions of these particles in sediments, surface waters, and biota throughout the estuary (Battaglia et al., 2020; Blosser, 2022; Gray et al., 2018; Kell, 2020; Leads and Weinstein, 2019; Parker et al., 2020). These particles are generated from tire abrasion on roads and have been identified as an often overlooked, but potentially more toxic component of microplastic litter (Knight et al., 2020). Previous research in our laboratory determined that rivers, tidal creeks, and stormwater ponds are substantial contributors of TWPs and other land-based plastic debris to the harbor (Kell, 2020; Leads and Weinstein, 2019). Based on this previous research, the microplastic particle and polymer types collected from intertidal sediments in the present study likely originated from synthetic fibers released from WWTPs, commercial and recreational fishing equipment, the degradation of larger plastic litter, and tire abrasion (Conley et al., 2019; Gray et al., 2018; Kell, 2020; Leads and Weinstein, 2019; Wertz, 2015).

Sediment contamination tends to have a patchy distribution for a variety of chemicals (Burton and Johnston, 2010; Mecray and ten Brink, 2000). This patchy distribution has previously been documented for organic contaminants throughout estuarine sediments in coastal South Carolina (Sanger et al., 1999). By contrast, in the present study, we observed a relatively uniform distribution of microplastic contamination within the intertidal zone throughout the Daniel Island sampling area (Fig. 3). Similarly, Gray et al. (2018) reported uniform microplastic distribution within the low intertidal, high intertidal, high tide, and supralittoral zones of Daniel Island as well as several other sample locations within Charleston Harbor. The sediment grain size distribution on Daniel Island is relatively similar between the low and high intertidal zones. These sediments can be classified as medium to course sands, with an average grain size of 641 ± 252 μm in the low intertidal zone and 418 ± 39.8 μm in the high intertidal zone (Wertz, 2015). In a study analyzing the association between microplastic abundance and sediment grain size, Mendes et al. (2021) determined that fine grain sediments such as mud accumulated higher microplastic concentrations than courser sediments. However, within each of the courser sediment grain sizes (fine, medium, and course sand), microplastics were accumulated at similar concentrations (Mendes et al., 2021). When applied to the present study, these results suggest that the relatively similar grain size distribution throughout the Daniel Island intertidal zone may be one factor contributing to the low spatial microplastic variability observed at this site.

The level of spatial variability in microplastic abundance varies widely among studies. Consistent with the present study, Dekiff et al. (2014) reported a homogenous distribution of microplastics throughout beach sediments of the island of Norderney, Germany. Other studies, however, have reported significant spatial variability of microplastics in coastal sediments. For example, Díaz-Jaramillo et al. (2021) observed significantly greater microplastic accumulation in the upper intertidal zone of estuarine sediments in the Pampean and north Patagonian regions of Argentina. In addition to differences in grain size distribution, hydrodynamic processes may contribute to differences in microplastic spatiotemporal variability within and among studies. In Charleston Harbor, tides are the primary hydrodynamic force impacting the estuary, punctuated by daily differences in

wave, wind, and precipitation conditions (Yassuda et al., 2000). Other hydrodynamic processes impacting the Daniel Island study site include fluvial and marine inputs from the Cooper River, Wando River, and Atlantic Ocean (Patterson, 1987) (Fig. 1). In the present study, significant temporal variability in microplastic composition and abundance was observed (Figs. 2 and 4), and short-term differences in hydrodynamic and environmental processes (e.g., wind conditions) contributed to these differences. Within individual sampling events, however, daily hydrodynamic conditions were likely applied evenly across the 175 m sample site, potentially contributing to the low spatial variability observed in the present study. These results suggest that despite daily differences in hydrodynamic processes and environmental conditions, microplastics were deposited uniformly throughout the intertidal zone as tides receded (Stead et al., 2020). Similarly, Dekiff et al. (2014) determined that the environmental factors affecting microplastic accumulation such as wind and water currents were applied evenly across a 500 m stretch of beach in Norderney Island, Germany, leading to a homogenous spatial distribution of microplastics. Because the spatial distribution of microplastics in the present study and previous studies (Gray et al., 2018) was generally homogenous, future sampling efforts in Charleston Harbor should likely be directed at collecting a greater number of temporal replicates rather than spatial replicates. Together, these results highlight the necessity of conducting preliminary analyses to gain a basic understanding of the spatial and temporal variability at a study site prior to undertaking a more comprehensive sampling regime.

Several studies have investigated how environmental factors influence microplastic variability in coastal environments. In controlled laboratory experiments, Forsberg et al. (2020) showed that microplastic deposition on beaches is influenced by wave height, wind direction, and particle characteristics such as density and shape. In field studies, those environmental factors contributing to microplastic variability in intertidal sediments have varied among locations. For example, Carvalho et al. (2021) determined that the highly variable microplastic concentrations observed over a 13-day sampling period on the island of Fernando de Noronha off the coast of Brazil were primarily due to wind direction and surface currents. By contrast, Imhof et al. (2017) reported that water currents, waves, and tidal cycles likely contributed to high microplastic variability over 7 consecutive days on Vavvaru Island in the Maldives archipelago, while no direct relationship with wind speed or wind direction was observed. Similarly, Moreira et al. (2016) also showed that microplastic concentrations in beach sediments of the Paranaguá estuary in Brazil varied significantly across several consecutive tidal cycles. In the current study, we determined that wind direction on the day of sampling had the greatest individual effect on microplastic abundance across the 12 sampling events, with generally lower microplastic abundance observed when the wind was oriented offshore away from the study site (Fig. 5). These results are consistent with laboratory experiments and modeling studies which show that onshore winds can lead to the aggregation of suspended microplastics near the coastline and subsequent entrainment of microplastics in coastal sediments, whereas offshore winds can quickly move suspended microplastics offshore and prevent deposition (Cohen et al., 2019; Forsberg et al., 2020).

In the present study, multiple linear regression analyses showed that MHHW levels at the high tide preceding sampling, precipitation levels the day before sampling, average wind speed on the day of sampling, and high wave height on the day of sampling could be positively associated with microplastic abundance and also contributed to microplastic variability, but to a lesser extent than wind direction. Like wind direction, wind speed and wave height can influence the mobilization and transport of microplastics (Osinski et al., 2020). In surface waters, increased wind speeds are associated with decreased concentrations of microplastics due to greater vertical mixing (Kukulka et al., 2012). In beach sediments, however, increased wind speeds and wave height have been associated with greater concentrations of microplastics, potentially due to increased turbulence, remobilization, transport, and deposition of suspended microplastics in the intertidal zone (Forsberg et al., 2020; Kumar and Varghese, 2021; Osinski et al., 2020). Tidal processes and precipitation events can also strongly impact coastal microplastic concentrations. For example, Cohen et al. (2019) used field measurements and simulations to show that tidal currents can change surface water microplastic concentrations at a given location by a factor of 1000 within timespans as short as 30 min. In addition, Stead et al. (2020) determined that higher tide levels were associated with greater entrapment of microplastics in estuarine sediments, potentially due to increased water volume or interaction of microplastics with sediments and organic matter. Lastly, a number of studies have documented the association between precipitation levels and coastal microplastic concentrations (Antunes et al., 2018; Eo et al., 2018; Liu et al., 2022; Lorenzi et al., 2021). For example, Antunes et al. (2018) reported increased accumulation of microplastics in beach sediments along the Portuguese coast during seasons with higher precipitation levels, potentially due to increased river transport, stormwater discharge, and/or land-based runoff. In the present study, the combination of wind direction, MHHW levels, precipitation, wind speed, and wave height accounted for, at most, 31 % of the variation in microplastic abundance, meaning that additional abiotic factors may also be significantly contributing to variation and warrant further investigation.

Changes in land use and source emissions can impact microplastic concentration and composition over time (Carvalho et al., 2021; McEachern et al., 2019; Quesadas-Rojas et al., 2021; Rasta et al., 2021; Tsang et al., 2020); however, given our knowledge of microplastic sources in Charleston Harbor, it is likely that the short-term microplastic variability we observed was primarily due to environmental factors acting on microplastics already present within the harbor rather than short-term changes in input or emissions. As previously stated, the microplastics in Charleston Harbor arise primarily from the degradation of larger plastic debris (Conley et al., 2019; Gray et al., 2018; Kell, 2020; Leads and Weinstein, 2019; Wertz, 2015), which we have shown can begin emitting microplastics after 4 weeks of environmental exposure (Weinstein et al., 2016; Weinstein et al., 2020). Therefore, it is unlikely that we would have captured changes in these secondary microplastic emissions during our 17-day sampling period. In addition, precipitation was low (0.025–1.27 cm) throughout the 17-day sampling period and was individually only slightly associated with microplastic abundance, suggesting that new inputs or increased emissions from river flow, stormwater discharge, or land-based runoff was likely minimal. However, additional research on the potential contribution of short-term changes in microplastic emissions (both point source and non-point source) to daily microplastic variability in intertidal sediments is warranted. Several recent reviews have addressed the need and provided recommendations for the harmonization of methods in microplastic field sampling (Adomat and Grischek, 2021; Lusher et al., 2020b; Rochman et al., 2017; Stock et al., 2019). These recommendations depend on the study question being addressed and often identify the need to detect long-term spatial and temporal trends for effective microplastic monitoring programs. However, few reviews discuss the importance of considering short-term or daily microplastic spatiotemporal variability and the influence of environmental conditions when sampling. In one study, Adomat and Grischek (2021) report that 85.1 % of the literature they reviewed collected only single point sediment samples, despite the high potential for spatial and temporal variability. In addition, Michida et al. (2019) recommend that sampling should be conducted on days with similar conditions to limit the influence of environmental sampling conditions on microplastic survey results. Similarly, guidelines provided by the Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection recommend that microplastic sampling should only occur when wave heights are below 0.5 m and wind force is below 3 on the Beaufort Wind Scale (GESAMP, 2019). The present study further underscores the importance of considering these environmental conditions and associated short-term spatiotemporal variability during microplastic surveys. In addition, our results highlight the complexity and interplay that various environmental and hydrodynamic factors have in contributing to the variability observed in microplastic concentrations in coastal environments. Because this variability is likely universal in these dynamic systems, we recommend that future studies should account for similar factors when collecting and interpreting microplastic field data.

5. Conclusion

The present study provides the first analysis of the short-term, daily variability in intertidal microplastic abundance and composition within an urbanized estuary of the United States. Our results show that in dynamic coastal environments, intertidal sediment microplastic concentrations can vary widely (up to a 20-fold difference) within consecutive tidal cycles and on very short timescales between 12 and 24 h. Environmental sampling conditions significantly impacted microplastic abundance and contributed to daily variability. Over the 17-day sampling period, wind conditions had the greatest effect on differences in daily microplastic abundance. Together, these results indicate that single sampling events may not accurately reflect the level of microplastic contamination in the environment or the composition of predominant microplastics, and that environmental conditions can significantly impact survey results and interpretation. For this reason, small-scale spatiotemporal variability in microplastic abundance and composition in dynamic coastal environments worldwide should be considered when designing environmental sampling regimes. These findings are important for informing the harmonization of microplastic sampling methodology and represent important considerations when interpreting the data from one-time sampling studies in these habitats for exposure and risk assessment.

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Disclaimer

The statements, findings, conclusions, and recommendations are those of the authors and do not necessarily reflect the views of the South Carolina Sea Grant Consortium or the National Oceanic and Atmospheric Administration.

CRediT authorship contribution statement

Rachel R. Leads: Methodology, Formal analysis, Investigation, Data curation, Writing – original draft, Writing – review & editing, Visualization. **John E. Weinstein:** Conceptualization, Methodology, Resources, Writing – original draft, Writing – review & editing, Supervision, Funding acquisition, Project administration. **Sarah E. Kell:** Investigation, Data curation, Writing – review & editing. **Johnathan M. Overcash:** Investigation, Writing – review & editing. **Bonnie M. Ertel:** Investigation, Writing – review & editing. **Austin D. Gray:** Methodology, Investigation, Writing – review & editing.

Data availability

Data, associated metadata, and calculation tools are available on figshare, DOI: 10.6084/m9.figshare.20288538

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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