Rising Seas and Roadway Debris: Microplastic and Low-Density Tire Wear Particles in Street-Associated Tidal Floodwater

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Abstract:
Tidal flooding is increasingly common in low-lying coastal regions as sea levels rise. This type of flooding can occur on sunny days with no rainfall and may transport street-associated debris, such as microplastics (MPs) including tire wear particles (TWPs), to coastal systems. This research aimed to quantify MP abundance in tidal floodwater and investigate their fate. Three locations around Charleston, SC (USA) were sampled during 12 tidal floods, and their adjacent tidal creeks were sampled before and after 5 floods. Floodwater contained an average of 342± 60 MP/L. Most MPs in floodwater were low-density TWP (86.5%). MP abundance in tidal creek surface water following flooding did not change, suggesting that MPs were not immediately transferred to coastal waterways but deposited in adjacent marsh sediment. Elucidating transport routes of MPs in coastal environments is critical to understanding and preventing this type of contamination in the face of a changing climate.

Introduction

Flooding is one of the most frequent and widespread manifestations of climate change in coastal environments. Over the past several decades, increased flooding frequency along the coast has resulted in high tides encroaching upon streets in coastal cities globally, even on days with no rainfall (Sweet et al., 2014). This type of tidal flooding, often referred to as "sunny day" or "nuisance" flooding, is exacerbated by global mean sea level rise (SLR) and is anticipated to become more frequent with future climate change predictions (Vitousek et al., 2017; Moftakhari et al., 2015; Thompson et al., 2021). Globally, it is estimated that 0.5-0.7% of the world's land area will experience episodic coastal flooding by 2100 if defensive measures (sea walls, dykes, etc.) are not adapted; this is a 48% increase compared to present day coastal flooding (Kirezci et al., 2020). Based on these predictions, ~570 coastal cities and 800 million people living in coastal regions are at risk to experience regular tidal flooding (UCCRN Technical Report, 2018). Tidal flooding is more than just a nuisance, resulting in economic losses related to damaged infrastructure, overwhelmed stormwater systems, disrupted harbor operations, closed roadways, and disrupted economic activity (Sweet et al., 2019, Hino et al., 2020).

Like many cities along the east coast of the United States (USA), Charleston, South Carolina experiences frequent tidal flooding. Charleston ranks as one of the most vulnerable cities in the USA with 61,221 residents (40% of total population) and more than $11 million at risk due to frequent flooding by 2050 given current SLR and local land subsidence projections (Climate Central, 2016). Over the past 70 years, tidal flooding in Charleston has become increasingly more common (Fig. 1; Holloway, 2021). For example, in the two decades between 1951 and 1971, there was only 1 major flood, 5 moderate floods, and 56 minor floods. In 2020 alone, there were 7 major floods, 15 moderate floods, and 51 minor floods – resulting in 73 tidal floods occurring over 57 days (Coastal Flood Event Database). Beginning in 2025, Charleston Harbor is predicted to experience an estimated 28 additional days of annual flooding based on the National Oceanic and Atmospheric Administration's (NOAA) Intermediate-Low SLR scenario of 0.5m global mean SLR by 2100 (Thompson et al., 2021).

Across several locations in downtown Charleston, tidal flooding typically occurs during predictable high tide events which cause low-lying streets to be inundated as brackish seawater overflows street-adjacent tidal creeks (Fig. S1). Flooding from tidal inundation differs from flooding caused by precipitation, which typically flows through stormwater infrastructure (e.g.

sewer pipes, detention ponds, WWTP) before entering natural receiving waters. Ebbing tidal floodwater typically bypasses this stormwater infrastructure, resulting in a direct pathway for street-associated contaminants to enter coastal environments such as tidal creeks and salt marshes. Contaminant loads of tidal floodwater and their fate in the environment remains largely understudied; this represents a critical gap in our knowledge concerning the risks that tidal floodwaters may pose to the coastal environment.

Based on previous studies in Charleston Harbor (Weinstein, 2023), tidal floodwater is expected to contain microplastics. Microplastic particles (MPs) are small pieces of plastic <5mm which have been documented to be ubiquitous in aquatic and terrestrial ecosystems (Andrady, 2011; Law and Thompson, 2014; Hale et al., 2020). It has been estimated that in the USA alone, around 100 tons of MPs enter coastal waters annually from land-based sources (Gouin et al., 2011). MPs can be transported from the land into the aquatic environment via various pathways, including WWTP effluent, illegal dumping, or stormwater runoff (Horton et al., 2017). To date, no studies have investigated the MP load in tidal floodwater or its potential to be transported to coastal waters via this pathway. Previous studies in the Charleston Harbor have documented the presence of MPs in surface water, sediment, zooplankton, fish, and mammals (Gray et al., 2018; Leads and Weinstein, 2019; Payton et al., 2020; Parker et al., 2020; Battaglia et al., 2020).

Microscopic tire particles, commonly referred to as tire wear particles (TWPs) or sometimes tire-road-wear particles, are produced by the mechanical abrasion of tire tread on street surfaces and have gained increasing attention as a type of MP in recent years (Halle et al., 2020). TWPs are characterized by an elongated cigar shape and contain material from tire tread, including synthetic rubbers, carbon black, zinc, and mineral encrustations from the street surface (Sommer et al., 2018; Kreider et al., 2010). Tire-derived chemical leachates such as 6PPD-quinone, hexamethoxymethylmelamine (HMMM), benzothiazole, and zinc have been documented as toxic to aquatic organisms (Peter et al., 2018; Capolupo et al., 2020; Tian et al., 2020; Yang et al., 2022). Thus, the presence of TWPs in coastal waters warrants further research.

Modeling and chemical marker studies suggest that TWPs may be substantial contributors to MP pollution, including the Charleston Harbor, where they have been documented in surface waters and sediment (Kole et al., 2017; Hartmann et al., 2019, Leads and Weinstein, 2019; Sommer et al., 2018). Globally, it is estimated that 6,000,000 tons of TWPs are emitted annually (Kole et al.,2017). Most TWPs (90-99.9%) become airborne particulate or are deposited near the street surface (Panko et al., 2013; Sommer et al., 2018), but it has been suggested that a portion of these particles, especially those with lower densities (<1.2 g/cm^3) are capable of transport into estuarine environments (Leads and Weinstein, 2019; Unice et al., 2019). The environmental fate of these TWPs is largely dependent on their size, shape, and density, which can vary based on the original tire tread material composition and interactions with the street surface (Sommer et al., 2018; Halle et al., 2020).

The objective of this study was to assess the abundance and physical attributes of MPs, including TWPs, collected from street floodwater and adjacent salt marshes around downtown Charleston, SC. To accomplish this, we investigated levels of MPs in (1) floodwater during tidal flooding, (2) tidal creek surface waters before and after flooding, and (3) intertidal salt-marsh sediment near frequently flooded streets and their associated tidal creeks. We hypothesized that street-

associated MPs, including TWPs, would be present within street floodwater, would increase in abundance in tidal creeks following tidal flooding, and would be in high abundance in salt marsh sediments near streets that experience frequent tidal flooding. Tidal floodwater ebbing from the street surface drains across the salt marsh, which could act as a sink for anthropogenic particles before floodwater reaches the tidal creeks. Identifying these particles in tidal floodwater and street-adjacent salt marshes fills a critical gap in our understanding of how climate-related flooding contributes to pollution in the coastal environment.

Methods

Study locations

Three sites around the downtown Charleston peninsula that experience frequent tidal flooding were selected for this study: Cedar Street, Hagood Avenue, and North Market Street (Fig. 2). All three sites have adjacent tidal creeks which serve as the proximal source of tidal floodwater. The Cedar Street site is in a residential area near two highway bridges and experiences frequent flooding from its adjacent tidal creek, Newmarket Creek, which flows into the Cooper River (Fig. S2). The Hagood Avenue site is in a commercial and low-income residential area and experiences frequent and intense flooding from its adjacent tidal creek, Gadsden Creek, which flows into the Ashley River (Fig. S3). North Market Street is in the downtown Charleston Historic District, a commercial area home to the Charleston City Market and a popular tourist destination (Fig. S4). North Market Street was constructed on an old tidal creek bed (Major Daniel's Creek) about 200 years ago when the sea level was 0.6 meters lower. It is now connected to the remnants of this creek through a historic brick-lined conveyance system, which flows directly into Charleston Harbor (charleston-sc.gov).

Water sampling and processing

Street locations were opportunistically sampled for tidal floodwater from September 2019 to May 2021, though not all sites flooded during each sampling event. Tidal flooding was anticipated by the use of tide charts and the careful monitoring of water levels and wind patterns, both of which update every 6 minutes on the National Ocean Service (NOS) Tides and Currents Tidal Gauge for Charleston, Cooper River Entrance, SC- NOS site 8665530 (NOAA Tides and Currents Station Homepage). According to thresholds defined by the National Weather Service (NWS) for Charleston, minor flooding occurs when water level exceeds 2.13 m Mean Lower Low Water (MLLW), moderate flooding occurs with water levels between 2.286 m and 2.438 m MLLW, and major flooding occurs with water levels ≥2.438 m MLLW (Holloway, 2021). Street flooding begins during minor flooding; therefore, tidal flooding in this study was defined as water levels ≥2.13 m MLLW with no rainfall. Wind speed and direction were also monitored because it was noted that persistent winds from the northeast increase water levels in Charleston Harbor and lead to street flooding even when predicted tides were <2.13 m MLLW. If multiple days of nuisance flooding were expected, we collected floodwater when possible during the first flood event. Street flooding due to extreme rainfall alone was not sampled for this study, but precipitation during the previous seven days was noted for each sampling event obtained from the NWS Weather Observation History for Downtown Charleston (https://www.weather.gov/chs/climate) (Table 1).

Grab samples of floodwater (n=2 at each site) were collected in amber glass bottles (4L, n=62 total samples) during 12 tidal floods (Fig. S5). Samples were collected from the surface of the

floodwater by either submerging the glass bottle or scooping with a tin-coated steel can. Floodwater salinity (ppt) was recorded using an Oakton Portable Conductivity 450 handheld meter. Floodwater samples were transported back to the laboratory where contents were poured through nested stainless-steel sieves (63 µm and 500 µm). Particles retained on sieves were rinsed into 250 mL amber glass jars using deionized (DI) water, and any biological material such as algae or bacteria was digested by adding 5mL of 30% H_2O_2 (Fisher Scientific) to the samples at room temperature for 7 days (Nuelle et al., 2014; Leads and Weinstein, 2019).

Tidal creek surface water was collected from Cedar Street and Hagood Avenue's adjacent tidal creeks, as the creeks at these two sites are accessible and currently undeveloped. The tidal creek at North Market Street connects directly to the Charleston Harbor via an underground conveyance system and therefore was not sampled in this study. Cedar Street and Hagood Avenue's tidal creeks were opportunistically sampled before and after 5 predictable flood events (4L, n=2 at each site). Samples were collected during the flood tide within one day prior to anticipated street flooding (n=24) and during the ebb tide within one day following street flooding (n=24). Surface water samples were collected from the tidal creeks using a stainless-steel bucket and transported in amber glass bottles to the laboratory where they were processed using the same methodology as floodwater samples.

Sediment sampling and processing

Sediment was collected from Cedar Street and Hagood Avenue's adjacent salt marshes, as these two sites have well-established fringing salt marshes. All sediment samples were collected at low tide on October 1, 2021. Sediment replicates were collected in 250mL amber glass jars using a stainless-steel spoon from the surface (2cm depth) of the salt marsh. Samples were collected directly adjacent to the street as well as directly adjacent to the creek at each location (Fig. S2, S3). Sediment samples (n=8 total) were transported to the laboratory in amber glass jars and homogenized before 100g of each sample was removed for processing using a NaCl density-separation procedure adapted from Leads and Weinstein (2019). Briefly, the 100g sediment sample was added to a 1L glass jar with 400mL of filtered NaCl (≥99.0%, Fisher Chemical) solution of density 1.15-1.17 g/mL and shook for 2 minutes. Following a 24-hour settling period, the liquid supernatant was poured over nested stainless-steel sieves (63 µm and 500 µm), and retained particles were rinsed with DI water into 250mL amber glass jars. The remaining sediment underwent four density separations to remove all MPs ≤1.15 g/mL. This method extracts low-density MPs and TWP; this fraction is mobile and can be transported within aquatic ecosystems (Leads and Weinstein, 2019). Biological material was digested by adding 5mL of 30% H_2O_2 (Fisher Scientific) to the samples at room temperature for 7 days prior to analysis (Leads and Weinstein, 2019).

Particle identification

Suspected synthetic particles were visually identified using a stereomicroscope (Leica EZ4, magnification 8x – 35x) and were characterized based on particle shape (fiber, fragment, film, or sphere), color, texture, and other morphological features (Hidalgo-Ruz et al., 2012; Miller et al., 2021). A subset of particles (excluding TWP) was stored on double-sided Scotch® tape and analyzed for polymer composition using a XploRA PLUS micro-Raman Spectrometer (HORIBA) and processed using LabSpec software (version 6.5). Raman spectra were obtained using a 785 nm or 532 nm laser using gratings of 600 or 1200 grooves/µm, 1 – 10s acquisition

time, and 2, 4, 6, 8, or 10 number of accumulations. Spectra were obtained with a confocal slit width of 100 µm slit and a hole diameter of 300 µm using a 50-100x objective with filters ranging from 0.1 to 100%. Raman spectra were processed using Wiley's KnowItAll Raman Spectral Library and a spectral match of >70% to either a polymer or known plastic additive determined if the particle was anthropogenic in origin.

TWPs were identified based upon their distinctive shape and surface characteristics described in Kreider et al. (2010) and Sommer et al. (2018) using the criteria set forth in Leads and Weinstein (2019). Specifically, particles were classified as TWP based upon the following physical attributes: dark black in color, elongated and cylindrical in shape, surface texture rough with encrustations of road and/or brake dust, irregular morphology, and overall rubbery consistency when probed with forceps (Leads and Weinstein, 2019). Additionally, a composite sample of suspected TWP (n=150) pulled from eight floodwater samples was analyzed using a Nicolet iS20 FTIR Spectrometer equipped with a Smart iTX Attenuated Total Reflectance ATR Accessory (Thermo Fisher Scientific Inc.). FTIR spectrum was obtained using a germanium crystal plate with 16 scans at a resolution of 4 and gain of 1 (Table S1), according to Thermo Scientific Application Note AN53141 (2019). The obtained spectrum was collected, processed, and identified using OMNIC Specta software (version 9.12.1019; Thermo Fisher Scientific Inc.) Spectral peaks between 3300-600 cm^{-1} were analyzed using a multi-component search using the Thermo Fisher Scientific Hummel Polymer and Additives FT-IR Spectral Library in OMNIC Specta.

Quality assurance/quality control

Laboratory protocols were in place to minimize contamination of samples. All materials used throughout the collecting, processing, storage, and analysis of samples were rinsed 3x with DI water and covered with aluminum foil when not in immediate use. Plastic materials were avoided at all steps, apart from DI wash bottles (100% LDPE, Fisher Scientific) used to rinse sieves. White cotton lab coats (100% cotton, Fisher Scientific) and nitrile gloves (Fisher Scientific) were worn in the laboratory, and the color of clothing during sample collection was noted. Contamination was quantified using procedural blanks, which were processed using DI water alongside samples for each sampling date. Laboratory procedural blanks (n=16) contained 6.0 ± 0.6 particles/blank (mean ± SE) for the 63 µm size fraction and 3.7 ± 0.5 particles/blank (mean ± SE) for the 500 µm size fraction. These values were subtracted from samples of each respective size fraction. In addition, positive controls (n=5) were processed using polyethylene beads, crumb rubber, and nylon fibers; we had an 87% MP extraction efficiency.

Statistical analyses

Particles retained in the 63 µm and 500 µm size fractions were independently analyzed and both blank-corrected size fractions were summed for each water and sediment replicate. Replicates were then averaged for statistical analysis. All water data is reported as MP/L and was tested for normality (Shapiro-Wilk, $p = 0.076$); all sediment data is reported as MP/kg wet weight (ww) and was tested for normality (Shapiro-Wilk, $p = 0.957$). Unless otherwise noted, all values represent mean ± SE. Differences in floodwater MP abundance were tested for significance using one-way analysis of variance (ANOVA) tests with the following factors: site, sampling date, season, floodwater salinity, and recent rainfall. Differences in overall MP abundance in tidal creeks was tested for significance using a paired t-test comparing before and after flooding.

Differences in tidal creek and sediment MP abundance by site were determined using ANOVAs and Tukey's Honest Significant Difference (HSD) post-hoc test indicated pairwise groupings by site. Statistics were conducted in R Studio (version 1.4).

Results

Street floodwater

MPs and TWPs were found in all street floodwater samples collected in this study. Across all sites and dates (n=62), floodwater samples contained an average of 341.8 ± 59.6 MP/L (Fig. 3). Cedar Street floodwater contained an average of 265.2 ± 95.5 MP/L (range 4-1916 MP/L). Hagood Street floodwater contained an average of 447.9 ± 119.3 MP/L (range 1-1708 MP/L). North Market Street floodwater contained an average of 284.4 ± 100.6 MP/L (range 3-1135 MP/L). There were no significant differences in MP abundance in tidal floodwater when the following factors were considered: site, sampling date, season, floodwater salinity, and recent rainfall (all ANOVA, $p>0.1$).

Of the 86,409 particles identified from street floodwater, the most common particle types were TWPs (86.5% of all MP) and fibers (13.0% of all MP) (Fig. 4, Table S2). Most fibers were transparent fibers (87.6% of all fibers) followed by blue fibers (4.3% of all fibers). Fragments, microbeads, and films of various colors constituted 0.3%, 0.1% and 0.1% of all MP in floodwater, respectively.

Tidal creek surface water

MPs and TWPs were identified in all tidal creek surface water samples, though there were no clearly defined trends in MP abundance before and after tidal flooding of street surfaces (Fig. 5). Across all sites and dates, tidal creek samples contained an average of 11.7 ± 1.9 MP/L before flooding and 10.2 ± 1.8 MP/L after flooding, however there was not a statistically significant increase in MP abundance after flooding as we had hypothesized (paired t test, $p =0.51$). When analyzed by site, the only significant difference in MP abundance was at Hagood Avenue during June 2020 when there was a significant decrease in MPs after flooding (1.2 MP/L) compared to before flooding (9.9 MP/L) (ANOVA, $p = 0.03$).

Of the 2,578 particles identified from tidal creek surface water, the most common particle types were fibers (51.7% of all MP) and TWPs (43.4% of all MP) (Table S1). Fragments, microbeads, and films of various colors constituted 3.4%, 1.0%, and 0.6% of all MP in tidal creeks, respectively. One preproduction pellet, or "nurdle," was present in a tidal creek sample after tidal flooding at N. Market Street on Sep 24, 2020.

Salt marsh sediment

MPs and TWPs were identified in all salt marsh sediment samples collected adjacent to the street and adjacent to the tidal creek (Fig. 6). At Hagood Avenue, MP concentrations in sediment immediately adjacent to the street averaged 4,270 ± 1,410 MP/kg ww, and at nearby Gadsden Creek concentrations immediately adjacent to the creek averaged 915 ± 105 MP/kg ww. At Cedar Street, MP concentrations in sediment immediately adjacent to the street averaged 3,730 ± 990 MP/kg ww, and at nearby Newmarket Creek concentrations immediately adjacent to the creek averaged 8,735 ± 385 MP/kg ww. MP were found in a significantly higher abundance in

surface sediments along the banks of Newmarket Creek (ANOVA $p = 0.015$, Tukey HSD groups shown in Fig. 6).

Of the 3,530 particles identified from saltmarsh sediment samples, the most common particle types were TWPs (69.2% of all MP) and fibers (25.1% of all MP) (Table S2). Fragments, microbeads, and films constituted 4.3%, 0.08%, and 0.5% of all MP in salt marsh sediment, respectively.

Particle identification

With respect to size, significantly more particles were present in the 63-500 µm size fraction (86.2% of all MP) than in the ≥500 µm size fraction (13.8% of all MP) across all samples (t-test, $p < 0.001$).

Of the subset of particles analyzed with micro-Raman spectroscopy (n=52), 8 (15.4%) were determined to be natural, 5 (9.6%) were inconclusive due to burning or melting of the particles, and 39 (75%) were identified as anthropogenic (Table 2). Anthropogenic particles were classified as such based on either a match to a synthetic polymer or chemical additives. Of the 39 anthropogenic particles, 17 (43.6%) were matched to synthetic polymers (including polystyrene, polyacrylic, polyester and nylon), and 22 (56.4%) were matched to chemical additives or dyes commonly used in plastics (including titanium oxide, superfex 200, drimaren brilliant green, or flexo blau) (Munno et al., 2020).

The composite sample of suspected TWP analyzed with FTIR spectroscopy (n=150) yielded a multi-component match of 72.32 which included tire-related materials (Wagner, 2018). The materials identified were chlorinated polyethylene (20.12% of composite), styrene butadiene (11.45%), silicone polymer (21.21%), and bentonite (47.21%) (Fig. S6). These results suggest that our composite sample did indeed contain TWP with mineral encrustations.

Discussion

The results of this study suggest that the contaminant loads of MP, especially TWPs, are high in street-associated tidal floodwater. The concentrations of MP in tidal floodwater reported here, ranging from 4-1916 MP/L, are higher than that reported for surface water of Charleston Harbor (ranging 3-11 MP/L) (Gray et al., 2018) and its tributaries (ranging 11-146 MP/L) (Leads and Weinstein, 2019). MP abundance was highly variable and not explained by any of the factors investigated in this study. Many other studies have noted that there is great spatial and temporal variability in MP concentrations within coastal ecosystems (Akdogan and Guven, 2019; Balthazar-Silva et al., 2020; Leads et al., 2023;). Therefore, additional research is necessary to better understand the factors driving MP abundance in tidal floodwater.

Tidal floodwater concentrations of MPs reported here are higher than values reported for stormwater runoff associated with precipitation. For example, Werbowski et al. (2021) reported concentrations ranging 1.1-24.6 MP/L in stormwater from the San Francisco Bay area, and Sang et al. (2021) reported concentrations ranging from 2.75-19.04 MP/L in stormwater from Wuhan, China. Several factors may be involved in explaining the order of magnitude differences in MP concentrations between tidal floodwater reported here and stormwater. The higher density of brackish water (≤1.02 g/mL) associated with the tidal floodwater in this study may have resulted

in more MP floating off of the street surface relative to the lower density of rainfall (1.00 g/mL) associated stormwater. Additionally, recent studies have investigated how the transport and settling of MPs in water may be influenced by turbulent dispersion (Cardoso-Mohedano et al., 2023; Shamskhany and Karimpour, 2022). Shamskhany and Karimpour (2022) observed that vertical settling of MPs < 200 µm is dominated by turbulent dispersion more so than gravitational settling. Mild turbulence was observed in the present study as the result of either tidal waters flooding onto the street or vehicles driving through the floodwater (Fig. S5). Under these conditions, small MPs may remain mobile in tidal floodwater regardless of their densities.

Furthermore, the location of sampling can explain some of the differences between MP abundance in tidal floodwater and stormwater. For example, the present study sampled MPs from tidal floodwater directly overlying the street surface, whereas the aforementioned studies sampled MPs from stormwater after it had passed through stormwater infrastructure, including streams (Werbowski et al. 2021) and stormwater pipes (Sang et al. 2021). Such stormwater infrastructure would reduce turbulence and allow for MPs to settle prior to being sampled. Results from Piñon-Colin et al. (2020) support this notion; they sampled MPs from stormwater directly from the street curb in Tijuana, Mexico and reported MP concentrations ranging from 12-2054 MP/L. These MP levels are similar to those reported in the present study. It is important to note that street-associated tidal floodwater bypasses stormwater infrastructure and instead ebbs into adjacent salt marshes and tidal creeks, and these ecosystems may similarly reduce turbulence thus allowing MPs to settle.

Following tidal flood events, we did not observe an increase in MP concentration in the surface water of adjacent tidal creeks, indicating that there was not an immediate transfer of MPs from street floodwater into coastal waters. Rather, it is likely that these street-associated MPs settled into the intertidal sediments of the adjacent, fringing salt marsh before ebbing floodwater reached the tidal creeks. At the two marshes sampled in this study, the vegetation was predominantly salt marsh cordgrass, *Spartina alterniflora*, which could have served as a vegetative buffer to reduce turbulence and therefore prevent many of the MPs from reaching the surface water of tidal creeks. Salt marsh sediment sampled in this study contained relatively high levels of low-density MP, ranging from 915-8735 MP/kg ww. This range is higher than intertidal sediment sampled from other Charleston tributaries (0-652 MP/kg ww; Leads and Weinstein, 2019), supporting the notion that these fringing salt marshes were capturing at least some of the street-associated MPs.

Previous studies have also indicated that salt marsh vegetation can serve as a buffer to capture MPs. For example, Yao et al., (2019) reported that MPs were retained in the sediments at the edge of a *S. alterniflora* salt marsh. Their results suggest that salt marsh vegetation aids in the trapping of MPs as well as macroplastic litter, which can be a source of MPs (Yao et al., 2019). In the salt marsh environment, macroplastic litter degrades in as little as four weeks due to oxidative reactions (Weinstein et al., 2020; Andrady, 2011). By trapping plastic litter, salt marshes are a net sink for MPs in the coastal marine environment. Along the New England coastline (USA), researchers report that salt marsh sediments sequestered MP fibers and fragments in sediment layers dating back to 1950 (Lloret et al., 2021). Certainly, further research is warranted to better characterize the role of adjacent, fringing salt marshes in mitigating MP contamination from street-associated tidal floodwater. We also recommend that future studies

include a reference marsh which experiences tidal flooding but is not near any streets or bridges, to delineate the role of salt marshes in capturing MPs-associated with estuarine surface water and atmospherically-deposited particles.

At our Cedar Street sampling site, sediment concentrations of MPs were highest in the creek-adjacent salt marsh of Newmarket Creek rather than the street-adjacent salt marsh. The proximity of Newmarket Creek to the Arthur Ravenel Jr. Bridge (Fig. 2) is likely why we found higher sediment concentrations in the creek-adjacent salt marsh. This bridge, which is 4 km long and 8-lanes, connects downtown Charleston to Mt. Pleasant, the first and fourth most populous cities in South Carolina, respectively. With an annual average daily traffic volume of 90,900 vehicles/day (SC Department of Transportation, 2022), this bridge is likely a major source of MPs and TWPs into the Newmarket Creek salt marsh through direct drainage or atmospheric transport. Indeed, previous studies from our laboratory have found concentrations as high as 1.3×10^7 TWP/m^2 in the sediment of drainage ditches associated with stormwater runoff from this bridge (Weinstein, 2023).

The multi-component spectral identification of the low-density TWP composite sample from street floodwater indicates that these particles contain synthetic rubber, polymers, and minerals (Fig. S6). Tire materials are complex and proprietary mixtures which differ based on tire application, manufacturer, and region. However, the three polymers identified from this spectrum are involved in tire manufacturing. Styrene butadiene rubber is one of the two main synthetic rubbers used in tire tread, constituting approximately 24% of a passenger car tire tread (U.S. Tire Manufacturing Association, 2020). Additionally, silicone and chlorinated polyethylene are both synthetic polymers used as fillers in tires to improve tread performance and traction (U.S. Tire Manufacturing Association, 2020; Atman, 2023). The remaining component, bentonite, is a water-absorbing clay with widespread applications including concrete stabilization, wastewater treatment, excavation, underground sealant, cat litter, and cosmetics, (Mendez, Ahmad et al., 2022). The detection of bentonite reflects its association as part of the mineral encrustation found on TWP surfaces (Kreider et al., 2010).

This study identified low-density TWP as the most abundant particle type of MPs in the water and sediment surrounding streets which experience frequent tidal flooding. TWP represented 86.5% of all MPs in street floodwater, 69.2% in salt marsh sediment, and 43.4% in tidal creek surface water (Fig. S7). TWPs sampled from street floodwater and tidal creek surface water were floating in brackish water, which may have been aided by turbidity, but it is still likely that the majority of TWP collected from water samples were ≤ 1.2 g/mL in density. TWPs sampled from salt marsh sediment were extracted using saturated NaCl solution with a density of 1.15-1.17 g/mL; this procedure is not affected by turbidity. The density of field-collected TWP has been reported to be up to 1.8 g/mL (Unice et al. 2019), and it is therefore likely that this study only captured a portion of TWP with lower densities. However, these low-density TWPs have greater mobility in turbid coastal waters and are therefore relevant in terms of the overall environmental transport and fate of these otherwise poorly understood MPs. Elucidating the transport and fate of MPs in our coastal environments is critical to understanding and preventing this type of contamination.

Conclusions

The results of this study indicate that street-associated tidal floodwater contains relatively high levels of MPs, especially TWPs. While the environmental fate of these particles on the ebb tide remains unclear, the abundance of these particles in the sediments of adjacent salt marshes suggests that at least some are being captured by the presence of the vegetation. Assessing tidal floodwater as a pathway for MPs, including TWPs, away from the street into the estuarine environment is necessary, especially under anticipated sea level rise scenarios expected to exacerbate coastal flooding in the United States and abroad. Urban cities within the coastal zone will likely experience more frequent tidal flooding in years to come, and, like Charleston, floodwater associated with rising sea levels could be significantly contributing to MP contamination of coastal ecosystems. In the face of a changing climate, it is imperative to better understand how rising sea levels impact marine pollution.

Data Availability

Data is publicly available and can be accessed at:
https://figshare.com/articles/dataset/Microplastics_in_Tidally-Induced_Floodwater_Saltmarshes_and_Tidal_Creeks_/23280365

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