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Seasonal and Cross-Shore Assessment of Large and Small Microplastics Collected on the Ferrara Coast (Italy)

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Abstract

Microplastic (MP) contamination along coastlines is a globally recognized environmental concern. This paper investigates the seasonal and cross-shore distribution of large and small microplastics (LMPs and SMPs) at four sites along the Ferrara coast in the northern Adriatic Sea (Italy). A combination of sampling and analytical approaches was employed to characterize the typology, morphology, and size of MPs. A subsample of LMPs was analyzed by Raman spectroscopy to determine polymers' composition. The mean abundances recorded were 5.66 ± 13.15 LMPs/m^2 and 2402.19 ± 1169.85 SMPs/m^2. Among the LMPs, pellets and fragments, essentially cream and white in color, were dominant. The samples were predominantly composed of polyethylene, polypropylene, and polyethylene terephthalate. SMPs primarily consisted of black fibers. LMPs and SMPs displayed their lowest abundances in winter and cross-shore patterns indicated preferential accumulation at dune foot and crest. Since the study sites are located downstream of the Po and Reno river mouths, urban and riverine discharges, as well as emissions from plastic-processing industries, are likely major contributors to the observed MPs.

Keywords: environmental monitoring; coastal sediments; stereomicroscope; Raman; Po Delta

1. Introduction

Coastal environments are highly dynamic systems that are subject to multiple natural and anthropogenic stressors [1,2]. One of the greatest global concerns regarding anthropogenic pressures is the increasing presence of plastic litter [3–7].

Global demand for plastic products has sharply risen over recent decades, largely due to the versatility of key polymers such as polyamide (PA), polyethylene (PE), polyethylene terephthalate (PET), polypropylene (PP), polystyrene (PS), and polyvinyl chloride (PVC) [8]. Depending on their intended applications, these polymers may incorporate fillers and additives, such as plasticizers, pigments, reinforcers, flame retardants, and stabilizers [3,9,10]. Once released into the environment, plastic items undergo degradation through abrasion, hydrolysis, biodegradation, and photo-thermal-oxidative processes [8,9,11–13]. Plastic residues can increase their ecotoxicological risk by absorbing and releasing additional substances and pollutants, such as insecticides, persistent organic pollutants, polycyclic aromatic hydrocarbons, polyfluoroalkyl substances, and potentially toxic elements [14–23].

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Degradation processes alter the chemical and physical properties of plastic items, resulting in the formation of increasingly smaller fragments [8,24]. These changes influence their environmental fate, transport pathways, and accumulation mechanisms [10,25–28].

Microplastics (MPs) are generally defined as heterogeneous, solid, synthetic polymer particles that are less than 5 mm in size, with regular or irregular shapes, insoluble in water and persistent in the environment [8,10,29–31]. MPs can be classified based on their physical characteristics (shape, color, size) or their chemical properties (polymer composition and/or additives) [17,32]. Due to their persistence, MPs are considered a major threat to ecosystems' health, having been detected in all environmental compartments and in a wide range of organisms [33–37]. MPs include primary items (e.g., virgin resin pellets, pre-production pellets, microplastics used in cosmetics, or abrasive compounds or sandblasting) and secondary items (fragments and fibers derived from the degradation of larger objects) [9,38–41]. MPs can reach coastal environments through surface runoff or in situ fragmentation of macro/meso litter [8,41–43]. MPs are found in both water bodies and coastal sediments, which act as the ultimate long-term sink [44–47]. Daily temperature excursions, high UV exposure, wave action, and abrasion by sediment particles particularly favor fragmentation in coastal sedimentary environments [12,48–50]. The presence of MPs can modify the porosity, permeability, and thermal properties of coastal sediments [51–54]. A recent study suggests that global plastic production will increase to approximately 753 Mt by 2060, with total MP emissions reaching around 298 Mt [55].

The dynamics of the coastal zone, variations in marine and weather conditions, and seasonality strongly influence the deposition and distribution of MPs, which may affect the reproducibility of monitoring studies [7,56,57]. Depending on the purpose of the study, sampling areas along beaches may vary considerably, with some investigations focusing on cross-shore transects and others on specific beach zones [53,58–61]. Therefore, selecting an appropriate sampling unit is crucial for accurately assessing MP contamination in coastal environments [62].

The proper monitoring of MP abundance and characteristics is essential for improving the understanding of coastal contamination. This paper investigates the seasonal and cross-shore distribution of MPs along the Ferrara coast and identifies seasonal distribution and cross-shore zones of higher accumulation and potential redistribution.

2. Study Area and Sampling Sites

The study area and the four sampling sites, i.e., Goro, Volano, Estensi, and Spina (Figure 1), are located along the Italian coast of the Northern Adriatic Sea, in the Province of Ferrara, between the mouths of the Po and Reno rivers [63]. Seasonal variations in sediment texture and geochemistry have been characterized according to different geomorphological features [64]. The sites were selected based on their proximity to ports, river mouths, urban coastal areas and their recreational value [63,65].

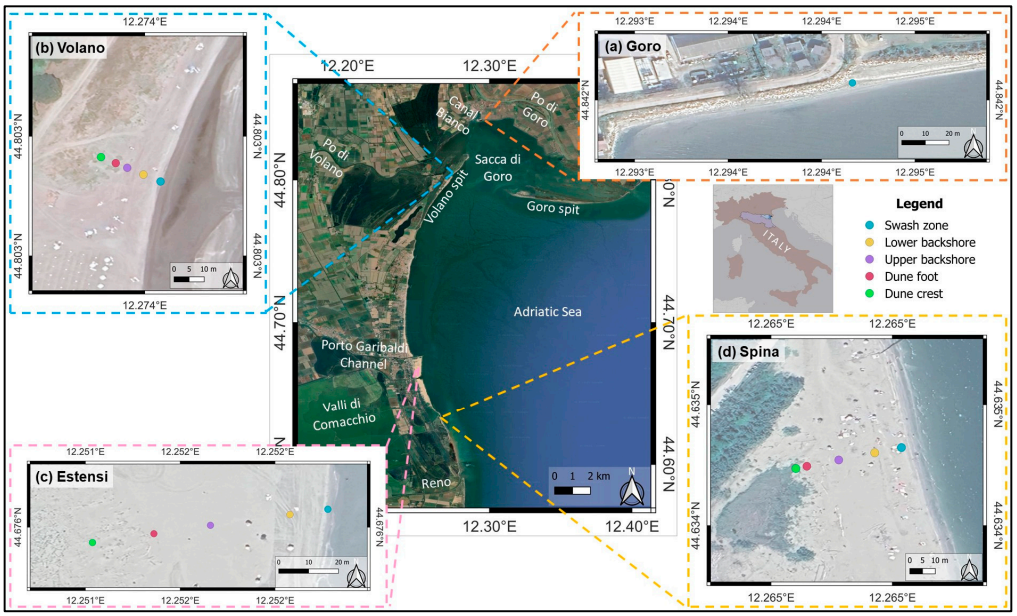

Figure 1. Overview of the study area (central map) and detailed views of the four sampling sites in the insets: (**a**) Goro; (**b**) Volano; (**c**) Estensi; and (**d**) Spina [64].

3. Materials and Methods

In this paper, different sampling and analytical methodologies were applied to extract two classes of microplastics (MPs) from coastal sediments [5,60,66]: large microplastics (LMPs, 1–5 mm in size), and small microplastics (SMPs, 1 µm–1 mm in size). These methodologies follow the recommendations of the Marine Strategy Framework Directive (MSFD) technical group [53,66], and include some modifications described below.

Within each study site, geomorphological zones (GZs) were identified based on beach width and cross-shore slope gradients [63,67] (Figure 1): swash zone (SZ), lower backshore (LB), upper backshore (UB), dune foot (DF), and dune crest (DC). At Goro, only the SZ was present, whereas all five GZs were identified at Volano, Estensi, and Spina. Sampling points were recorded using a Leica GS16 dGPS (Leica Geosystems SpA, Lodi, Italy) and georeferenced in WGS 84 UTM 32N to allow precise repetition of sampling during each season from summer 2023 to summer 2024 [64]. During low tide, one sediment sample was collected per site, season, and GZ for LMP extraction and another one for SMP extraction.

To minimize plastic contamination during field sampling and laboratory procedures, the following standard quality assurance and quality control (QA/QC) protocols were adopted [28,66,68–71]: (i) SMP samples were collected before LMP samples; (ii) cotton laboratory clothing was worn; (iii) filtered solutions and glass, metal, or paper tools were used; (iv) all tools and surfaces were rinsed with distilled water and ethanol before use; (v) samples and instruments were kept covered with glass lids or aluminum/paper sheets; (vi) laboratory work was conducted in a confined environment; and (vii) air movement was minimized by keeping doors and windows closed and restricting access.

3.1. Large Microplastics (LMP)

For LMP collection (1–5 mm), a 1 × 1 m quadrat was marked with a wooden folding ruler by placing one vertex at the GPS coordinates (Figure 1). Within each quadrat, 3 L of surface sediment were collected using a metal shovel. If the sediments were dry, the samples were sieved in situ through stacked metal sieves (5 mm and 1 mm; Giuliani Tecnologie srl, Turin, Italy). For wet sediments, the samples were collected and labeled in field, air-dried for 2–3 days in a controlled environment to avoid contamination, and subsequently sieved. The material retained between the sieves (LMPs mixed with shells, sediment grains, and

vegetable debris) was stored in labeled paper bags and manually sorted to isolate the LMPs. The following characteristics of each LMP item were recorded in an Excel sheet (Office 365, Microsoft Corporation, Redmond, WA, USA) [60,66,72]: (i) typology: fiber/filament, film, foam, fragment, or pellet/granule/sphere; (ii) roundness: angular, sub-angular, rounded, sub-rounded, very angular, or well-rounded; (iii) dimensions: length, width, and height; (iv) color: black, blue, brown, colorless, cream, green, gray, orange, pink, red, tan, white, or yellow; and (v) transparency: opaque or transparent. The LMP dimensions were measured using an electronic digital vernier caliper.

3.2. Small Microplastics (SMP)

For SMP sampling (1 µm–1 mm), a 0.4 × 0.4 m quadrat was placed in the least-disturbed surface adjacent to the LMP sampling area, with one vertex positioned at the GPS coordinates (Figure 1). Using a metal spoon, 500 mL of surface sediment were collected and placed in labeled glass jars. The jars were then transported to the laboratory. Each sample was quartered to obtain three 50 mL aliquots. For each analytical set (three aliquots), a procedural blank consisting of 200 mL of high-density sodium chloride (NaCl) solution was processed with the sediment samples [73].

Each sediment aliquot was placed in a glass beaker and mixed with 200 mL of saturated NaCl solution [28,60,66]. A magnetic stirrer (Atlas, Milan, Italy) was used at 600 rpm for 2 min. In the Volano samples, the minerals containing Potentially Toxic Elements (PTEs) included ferromagnetic minerals [64]. When magnetic stirring was not possible, vigorous manual stirring with a glass stirrer was used instead. The beakers were covered with a glass lid, and the aliquots were allowed to settle for 40 min. This duration was determined through preliminary tests based on the sediment grain size. The 2 min settling time recommended by the MSFD protocol [66] was insufficient due to the presence of fine to very fine sands.

The supernatant was vacuum-filtered through borosilicate glass microfiber filters (Whatman® GF/D, GE Healthcare Life Sciences Ltd., Little Chalfont, UK) using a 6-place SpeedFlow Multivac filtration unit (CRAMI Group srl, Milan, Italy) with an oil-free Rocker 900 pump (Rocker Scientific, New Taipei City, Taiwan). The filters were rinsed three times with 200 mL of distilled water. Then, they were placed in borosilicate Petri dishes (Anumbra®, FGH plus s.r.o., Šumperk, Czech Republic), covered, and dried at room temperature with silica gel (self-indicating without cobalt, granular size 2.5–6 mm, Carlo Erba Reagents srl, Val-de-Reuil, France). The filters were examined under a Nikon SMZ745T stereomicroscope (Nikon Instruments SpA, Florence, Italy), equipped with an 80× magnification, DFK 23UP031 camera (The Imaging Source Europe GmbH, Bremen, Germany), and IC Measure software (v2.0.0.245, The Imaging Source Europe GmbH, Bremen, Germany). Each filter was examined exhaustively in an "S-pattern" from top-right to bottom-left to ensure full coverage. The following characteristics of each SMP item were recorded in an Excel sheet: (i) typology: fiber/filament, film, foam, fragment, or pellet/granule/sphere; (ii) dimensions: length and width; (iii) color: black, blue, brown, colorless, cream, green, gray, orange, pink, red, tan, white, or yellow; and (iv) transparency: opaque or transparent. The SMP dimensions were measured using the integrated tool in IC Measure software.

Visual identification of SMPs is challenging and time-consuming, and requires experience to avoid misclassification [17,28,74–81]. When uncertainty arose, published guides were consulted ([10,78,82], https://flseagrant.ifas.ufl.edu/media/flseagrantifasufledu/sea-grant/pdf-files/microplastics/MERI_Guide-to-Microplastic-Identification.pdf, accessed on 30 August 2025).

3.3. Polymer Identification

The MSFD protocol recommends identifying the polymer of at least 10% of the collected items [53,66]. In this study, 15% of LMPs were randomly selected for polymer identification via Raman spectroscopy, to maintain proportional representation of all observed typologies.

A LabRAM HR800 μRaman spectrometer (Horiba Scientific, Turin, Italy) was employed. It was equipped with an air-cooled CCD detector (−70 °C), an Olympus BXFM microscope, a 600-groove/mm grating, and a 632.8 nm He-Ne laser (17 mW maximum power). Spectra (100–4000 cm^{-1}) were acquired with an optimized exposure time, beam power, and accumulations for each sample, and were calibrated using a silicon standard. The data were processed using LabSpec 6 software and then were compared with the BioRad library (KnowItAll, Informatic System 2021, John Wiley & Sons Inc., Hoboken, NJ, USA) and published spectral references.

Despite multiple preliminary attempts to identify polymers of SMPs via Raman spectroscopy, the analysis was not feasible due to (i) the extremely small size of SMPs, preventing isolation from the filters; (ii) luminescence from the borosilicate filters and residual salt (despite triple rinsing); and (iii) limitations associated with the available μRaman configuration.

3.4. Data Analyses

Following Hanke et al. [66] MPs were categorized by appearance and size. Abundances were normalized per unit area (items/m^2) and per volume (items/L), but not per weight (items/kg). Elevated concentrations of PTE-bearing minerals at Volano increased sediment bulk density [64], which prevented meaningful weight-based comparisons across sites.

To evaluate procedural contamination in SMP samples [10,68,70,71,83–85], the LOD/LOQ method was applied to determine the limits of detection and quantification of SMPs [86]. SMPs were considered detected when their counts exceeded the LOD (3.3 × SD of blanks) and quantifiable when they exceeded the LOQ (10 × SD of blanks). Data were excluded from further analyses when SMP counts were ≤LOD. Non-normalized raw data were used for these calculations.

To assess the environmental impact of MPs on sediments, the following indices were calculated [35,87]:

1. The Microplastic Pollution Index (MPPI), which represents MP abundance per standardized sampling unit (items/m^2);
2. The Coefficient of Microplastic Impact (CMPI), which evaluates the relationship between MP typologies and total MP abundance per standardized sampling unit (items/m^2).

Statistical analyses were performed using RStudio (2025.05.0+496) [88] and Jamovi 2.6.26 [89]. Pearson's chi-square goodness-of-fit tests (χ^2) were used to test hypotheses on categorical data. Independent-sample contingency tables with χ^2 statistics were used to evaluate the associations between categorical variables. Degrees of freedom and significance thresholds ($p < 0.05$) are reported for each χ^2 test. For each site, season, and GZ, the descriptive statistics (mean, standard deviation, median, minimum, maximum) were calculated for the MP dimensions.

4. Results

A total of 128 sediment samples were analyzed, including 64 samples for LMP extraction and 64 for SMP extraction. On average, 5.66 ± 13.15 LMPs/m^2 and 2402.19 ± 1169.85 SMPs/m^2 were recovered. MPs were found across all sites, seasons, and GZs, with some exceptions described below.

4.1. Results—Large Microplastics (LMPs)

Overall, 362 LMPs were collected (Table 1). No LMPs were detected at the Goro site. Of the sites where LMPs were present, Volano had the highest abundance (132 items; 6.60 ± 17.98 items/m^2), followed by Estensi (124 items; 6.20 ± 11.24 items/m^2) and Spina (106 items; 5.30 ± 10.74 items/m^2). A chi-square goodness-of-fit test revealed significant differences in LMP abundance among the sites ($\chi^2(2) = 2.939$, $p > 0.05$), suggesting that the likelihood of detecting an equivalent number of LMPs varied among the sites. According to the MPPI (Table 1), Goro is generally characterized by the absence of LMPs, while Volano, Estensi, and Spina display moderate LMP abundance, with seasonal and spatial variations.

Table 1. LMP data collected at each site, season, and GZ, expressed as items/m^2 and items/L. A color scale was applied to facilitate interpretation of MPPI classes: light blue = "<2, very low abundance/absence"; green = "2–5, low abundance"; yellow = "5–15, moderate abundance"; orange = "15–25, high abundance"; red = "≥25, very high abundance".

Site	Sampling Season	GZ	Items/m^2	Items/L
Goro	Summer 2023	SZ	0	0.00
	Autumn 2023	SZ	0	0.00
	Winter 2024	SZ	0	0.00
	Spring 2024	SZ	0	0.00
Volano	Summer 2023	SZ	1	0.33
		LB	1	0.33
		UB	0	0.00
		DF	1	0.33
		DC	81	26.67
	Autumn 2023	SZ	2	0.67
		LB	0	0.00
		UB	4	1.33
		DF	0	0.00
		DC	5	1.67
	Winter 2024	SZ	2	0.67
		LB	0	0.00
		UB	3	1.00
		DF	3	1.00
		DC	0	0.00
	Spring 2024	SZ	0	0.00
		LB	6	2
		UB	18	6.00
		DF	5	1.67
		DC	0	0.00

Table 1. *Cont.*

Site	Sampling Season	GZ	Items/m^2	Items/L
Estensi	Summer 2023	SZ	1	0.33
		LB	0	0.00
		UB	0	0.00
		DF	0	0.00
		DC	3	1.00
	Autumn 2023	SZ	0	0.00
		LB	23	7.67
		UB	12	4.00
		DF	1	0.33
		DC	1	0.33
	Winter 2024	SZ	0	0.00
		LB	2	0.67
		UB	0	0.00
		DF	3	1.00
		DC	0	0.00
	Spring 2024	SZ	44	14.67
		LB	13	4.33
		UB	19	6.33
		DF	0	0.00
		DC	2	0.67
Spina	Autumn 2023	SZ	0	0.00
		LB	0	0.00
		UB	8	3.00
		DF	48	16.00
		DC	0	0.00
	Winter 2024	SZ	0	0.00
		LB	6	2.00
		UB	8	2.67
		DF	0	0.00
		DC	0	0.00
	Spring 2024	SZ	6	2.00
		LB	4	1.33
		UB	2	0.67
		DF	9	3.00
		DC	0	0.00
	Summer 2024	SZ	1	0.33
		LB	2	0.67
		UB	0	0.00
		DF	12	4.00
		DC	0	0.00

Figure 2 illustrates the different types of LMPs collected in this study. LMPs consisted primarily of pellets (57%; 208 items), followed by fragments (30%; 107 items), fibers (8%; 30 items), and films (5%; 17 items). A chi-square test confirmed significant differences in the distribution of types ($\chi^2(3) = 255.7$, $p < 0.001$).

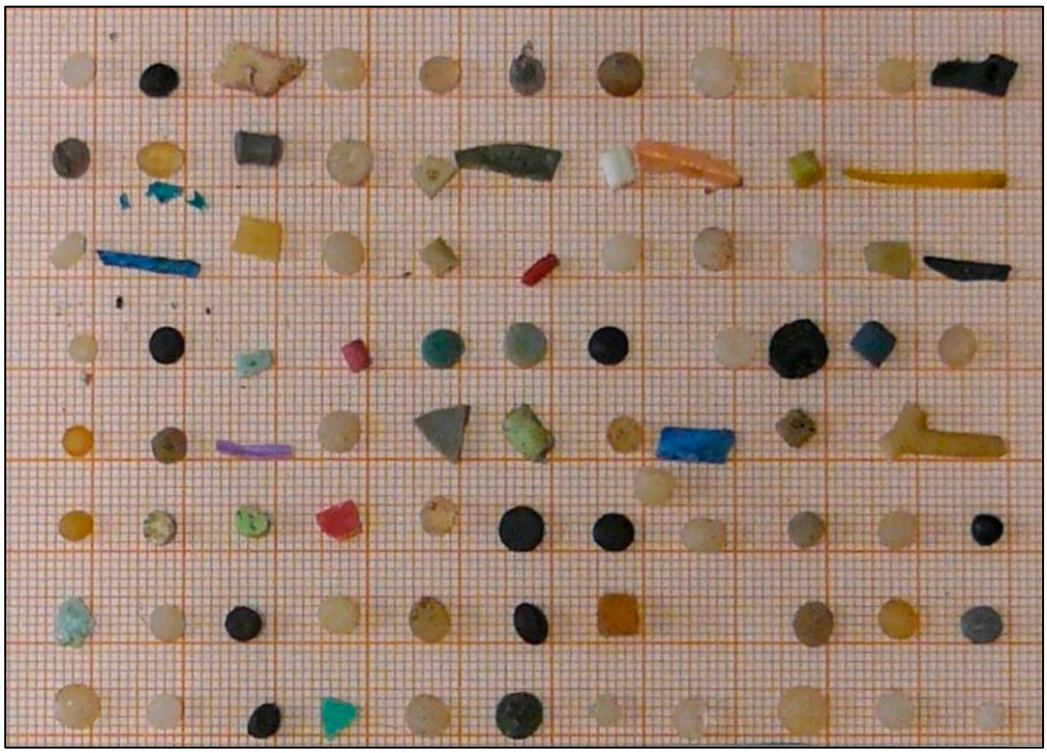

Figure 2. Examples of the different LMP typologies collected in this study.

The CMPI was maximum for pellets (0.57), average for fragments (0.30), and minimum for fibers and films (0.08 and 0.05). At Estensi, the CMPI was average for pellets (0.27), maximum for fragments (0.52), and minimum for fibers and films (0.10 each one). Regarding roundness, LMPs were mostly rounded (30%), followed by very angular (29%), well-rounded (25%), angular (13%), sub-rounded (2%), and sub-angular (1%).

The mean dimensions of the LMPs were as follows: (i) length: 5.23 ± 4.56 mm (median 4.20 mm; 1.00–30.10 mm); (ii) width: 3.08 ± 1.11 mm (median 3.30 mm; 0.10–5.10 mm); and (iii) height: 1.60 ± 1.02 mm (median 1.60 mm; 0.00–4.90 mm). The mean dimensions by type were: (i) pellets: 3.92 × 3.49 × 2.24 mm; (ii) fragments: 4.98 × 2.82 × 0.93 mm; (iii) fibers: 17.34 × 0.85 × 0.45 mm; and (iv) films: 7.98 × 3.57 × 0.04 mm. The pellets were generally spherical, discoidal, or cylindrical, while fragments and films were elongated and thin. The fibers were the longest items, with minimal width and height.

Most LMPs were opaque (94%). Regarding the color distribution, 32.32% of items were cream-colored and 23.76% were white. The other colors were minimally represented (Supplementary Materials, Figure S1).

A subsample of 54 LMPs analyzed via Raman spectroscopy revealed the following polymer composition (Table A1, Figure A1): PE (48%) > PP (26%) > PET (17%) > PA (6%) > PS (4%). Pellets mainly consisted of PE (55%), while fragments were predominantly of PET (38%) and PE/PP (31% each). Fibers were primarily composed of PP (75%), and films were entirely composed by PE.

4.1.1. LMPs—Seasonal Variations

A total of 103 LMPs were collected in summer (6.87 ± 14.10 items/m^2), 104 in autumn (6.93 ± 11.05 items/m^2), 27 in winter (1.80 ± 2.28 items/m^2), and 128 in spring (8.53 ± 9.52 items/m^2). A chi-square test revealed significant differences among the seasons ($\chi^2(3) = 63.83$, $p < 0.001$), with the lowest LMP abundance observed in winter. According to the MPPI (Table 2), summer, autumn, and spring exhibited moderate LMP abundance, whereas winter showed very low abundance.

Table 2. Abundances of LMPs (items/m^2) detected at Volano, Estensi, and Spina across the different seasons. A color scale was applied to facilitate interpretation of MPPI classes: light blue = "<2, very low abundance/absence"; green = "2–5, low abundance"; yellow = "5–15, moderate abundance"; orange = "15–25, high abundance".

	Items/m^2			
	Summer	**Autumn**	**Winter**	**Spring**
Volano	16.80	2.20	1.60	5.80
Estensi	0.80	7.40	1.00	15.60
Spina	3.00	11.20	2.80	4.20
Overall	6.87	6.93	1.80	8.53

Pellets were most abundant in summer, autumn, and winter, while fragments were most prevalent in spring (41.41%), particularly at Estensi (58.97%). A significant association was found between LMP typologies and seasons ($\chi^2(9) = 41.62$, $p < 0.001$).

CMPI patterns varied seasonally:

- Summer–autumn: the CMPI was maximum for pellets (0.75 and 0.64), average for fragments (0.18 and 0.28), and minimum for fibers/films (0.03–0.05).
- Winter: the CMPI was also average for fibers (0.15).
- Spring: the CMPI was average for pellets, fragments, and fibers (0.38, 0.41, and 0.15), and remained minimum for films (0.06).

Roundness also varied seasonally. LMPs were mostly rounded, well-rounded, and very angular in summer, autumn, and winter, whereas very angular particles dominated in spring (44.53%). The results were significant ($\chi^2(15) = 48.48$, $p < 0.001$).

LMPs were generally longer in winter and spring (6.42 ± 5.62 mm), and wider and thicker in summer and autumn (width: 3.33 ± 0.94 mm; height: 1.77 ± 0.91 mm). Cream and white remained the predominant colors (Supplementary Materials, Figure S2), and more than 90% of the LMPs were always opaque across seasons.

4.1.2. LMPs—Cross-Shore Variations

The LMP distribution across GZs was as follows: (i) SZ: 57 items (4.75 ± 12.48 items/m^2, 15.75%); (ii) LB: 57 items (4.75 ± 6.90 items/m^2, 15.75%); (iii) UB: 74 items (6.17 ± 6.94 items/m^2, 20.72%); (iv) DF: 82 items (6.83 ± 13.53 items/m^2, 22.65%); and (v) DC: 92 items (7.67 ± 25.15 items/m^2, 25.14%). A chi-square test revealed significant variation among the GZs ($\chi^2(4) = 13.17$, $p < 0.05$).

According to the MPPI (Table 3), the SZ and LB are characterized by low LMP abundance, while the UB, DF, and DC are characterized by moderate LMP abundance. Notably, high LMP levels were found at Volano (DC) and Spina (DF), although LMPs were absent at Spina's DC.

Table 3. Abundances of LMPs (items/m^2) detected at Volano, Estensi, and Spina across the different GZs. A color scale was applied to facilitate interpretation of MPPI classes: light blue = "<2, very low abundance/absence"; green = "2–5, low abundance"; yellow = "5–15, moderate abundance"; orange = "15–25, high abundance". Abbreviations: SZ—swash zone; LB—lower backshore; UB—upper backshore; DF—dune foot; DC—dune crest.

	Items/m^2				
	SZ	**LB**	**UB**	**DF**	**DC**
Volano	1.25	1.75	6.25	2.25	21.50
Estensi	11.25	9.50	7.75	1.00	1.50
Spina	1.75	3.00	4.50	17.25	0.00
Overall	4.75	4.75	6.17	6.83	7.67

Fragments dominated the SZ (61%) and LB (40%), while pellets dominated the UB (58%), DF (87%), and DC (72%). Fibers and films accounted for less than 16% in all zones, and the films were absent at the DF. The typology distribution differed significantly among GZs ($\chi^2(12) = 107.55$, $p < 0.001$).

According to the CMPI classification, the pellets' impact was average in the SZ and LB (0.14 and 0.35), maximum in the UB and DC (0.58 and 0.72), and extreme in the DF (0.87). The impact coefficient for fragments reached its maximum in the SZ (0.61), was average in the LB, UB, and DC (0.40, 0.28, and 0.22), and was minimum in the DF (0.10). The impact coefficient for fibers was minimum in the SZ, DF, and DC (0.09, 0.04, and 0.05), whereas it was average in the LB and UB (0.14 and 0.12). For films, the impact coefficient was average in the SZ and LB (0.16 and 0.11), while minimum values (0.01) were recorded in the UB and DC.

Roundness patterns matched typologies. In the SZ and LB (fragment-dominated) most of LMPs were very angular (50.88%) and angular (22.81%). At the UB, DF, and DC (pellet-dominated), most of the LMPs were rounded (37.50%) and well-rounded (31.85%). A chi-square test revealed a significant association ($\chi^2(20) = 102.74$, $p < 0.001$). LMPs were longer at the SZ, LB, and UB (6.21 $\pm$ 5.62 mm) than at the DF and DC (4.74 $\pm$ 2.74 mm). Cream and white remained the predominant colors, though the SZ essentially showed blue (21%) and green (16%) items (Supplementary Materials, Figure S3). LMPs were opaque ($\geq$82%) across all GZs.

4.2. Results—Small Microplastics (SMPs)

A total of 7687 SMPs were identified (mean: 2402.19 $\pm$ 1160.67 SMPs/m^2). The procedural blanks contained an average of 23.84 $\pm$ 12.55 SMPs, yielding LOD = 41.41 items and LOQ = 125.50 items. Table 4 displays the normalized data to standard surface and volume, indicating the detection and quantification limits for each sample. Three samples fell below the LOD and were excluded, while 61 samples exceeded the LOD, and 26 samples exceeded the LOQ and were accurately quantified.

The most contaminated site was Spina (2944.00 items/m^2), followed by Volano (2402.11 items/m^2), Estensi (2185.56 items/m^2), and Goro (2150.00 items/m^2). A chi-square test revealed significant differences among sites ($\chi^2(3) = 1784.66$, $p < 0.001$). The MPPI classified all sites as exhibiting very high SMP abundance. Figure 3 illustrates the different types of SMPs collected in this study. The Supplementary Materials contain enlarged versions of the following image (Figures S7–S36).

SMPs consisted of fibers (81.54%), fragments (13.54%), pellets (4.53%), and films (0.39%). The typology distribution differed significantly among sites ($\chi^2(3) = 13{,}273.55$, $p < 0.001$). CMPI values were extreme for fibers (0.82), average for fragments (0.14), and minimum for pellets and films ($\leq$0.10).

The mean dimensions of the SMPs were as follows: (i) length: 256.99 $\pm$ 243.78 µm (median 199.80 µm; 9.83–999.89 µm); and (ii) width: 56.31 $\pm$ 54.94 µm (median 42.12 µm; 10.47–815.60 µm). Due to their morphology, most fibers lacked a measurable width. Pellets were mostly spherical, and diameter was reported as length. The mean dimensions by type were: (i) fibers: 333.62 $\times$ 33.17 µm; (ii) fragments: 76.29 $\times$ 55.39 µm; (iii) pellets: 74.25 $\times$ 97.09 µm; and (iv) films: 320.21 $\times$ 88.04 µm.

SMPs were primarily black (40.18%) and colorless (20.92%) (Supplementary Materials, Figure S4), and 71% were opaque.

Table 4. SMP data collected at each site, season, and GZ (items/m^2 and items/L).

Site	Sampling Season	GZ	Items/m^2	Items/L
Goro	Summer 2023	SZ	1200 [2]	400.00 [2]
	Autumn 2023	SZ	1860 [2]	620.00 [2]
	Winter 2024	SZ	1440 [2]	480.00 [2]
	Spring 2024	SZ	4100 [3]	1366.67 [3]
Volano	Summer 2023	SZ	3820 [3]	1273.33 [3]
		LB	3560 [3]	1186.67 [3]
		UB	2800 [3]	933.33 [3]
		DF	2920 [3]	973.33 [3]
		DC	3700 [3]	1233.33 [3]
	Autumn 2023	SZ	1540 [2]	513.33 [2]
		LB	1700 [2]	566.67 [2]
		UB	4360 [3]	1453.33 [3]
		DF	1340 [2]	446.67 [2]
		DC	2060 [2]	686.67 [2]
	Winter 2024	SZ	520 [1]	173.33 [1]
		LB	1420 [2]	473.33 [2]
		UB	2340 [2]	780.00 [2]
		DF	980 [2]	326.67 [2]
		DC	1660 [2]	553.33 [2]
	Spring 2024	SZ	1580 [2]	526.67 [2]
		LB	2560 [3]	853.33 [3]
		UB	2220 [2]	740.00 [2]
		DF	2460 [2]	820.00 [2]
		DC	2620 [3]	873.33 [3]
Estensi	Summer 2023	SZ	2040 [2]	680.00 [2]
		LB	2360 [2]	786.67 [2]
		UB	2540 [3]	846.67 [3]
		DF	2520 [3]	840.00 [3]
		DC	3360 [3]	1120.00 [3]
	Autumn 2023	SZ	1740 [2]	580.00 [2]
		LB	3320 [3]	1106.67 [3]
		UB	2440 [2]	813.33 [2]
		DF	2760 [3]	920.00 [3]
		DC	4260 [3]	1420.00 [3]
	Winter 2024	SZ	260 [1]	86.67 [1]
		LB	1320 [2]	440.00 [2]
		UB	880 [2]	293.33 [2]
		DF	500 [1]	166.67 [1]
		DC	1140 [2]	380.00 [2]
	Spring 2024	SZ	1800 [2]	600.00 [2]
		LB	1820 [2]	606.67 [2]
		UB	1620 [2]	540.00 [2]
		DF	1600 [2]	533.33 [2]
		DC	1820 [2]	606.67 [2]

Table 4. *Cont.*

Site	Sampling Season	GZ	Items/m^2	Items/L
Spina	Autumn 2023	SZ	3220 [3]	1073.33 [3]
		LB	3600 [3]	1200.00 [3]
		UB	2440 [2]	813.33 [2]
		DF	3000 [3]	1000.00 [3]
		DC	2520 [3]	840.00 [3]
	Winter 2024	SZ	1400 [2]	466.67 [2]
		LB	3220 [3]	1073.33 [3]
		UB	2260 [2]	753.33 [2]
		DF	6920 [3]	2306.67 [3]
		DC	1700 [2]	566.67 [2]
	Spring 2024	SZ	1780 [2]	593.33 [2]
		LB	1640 [2]	546.67 [2]
		UB	2280 [2]	760.00 [2]
		DF	4440 [3]	1480.00 [3]
		DC	2940 [3]	980.00 [3]
	Summer 2024	SZ	2180 [2]	726.67 [2]
		LB	2780 [3]	926.67 [3]
		UB	1920 [2]	640.00 [2]
		DF	5100 [3]	1700.00 [3]
		DC	3540 [3]	1180.00 [3]

[1] Value $\leq$ LOD; [2] Value > LOD; [3] Value > LOQ.

Figure 3. Examples of the different SMP typologies collected in this study.

4.2.1. SMPs—Seasonal Variations

The distribution of SMPs varied by season (Table 5): 30.39% were recorded in summer (2896.25 items/m^2), 27.65% in autumn (2635.00 items/m^2), 24.45% in spring (2052.31 items/m^2), and 17.50% in winter (2330.00 items/m^2). A chi-square test revealed significant seasonal differences ($\chi^2(3) = 282.66$, $p < 0.001$). According to the MPPI, SMP abundance was very high across all seasons.

Table 5. Abundances of SMPs (items/m^2) detected at Goro, Volano, Estensi, and Spina across the different seasons.

	Items/m^2			
	Summer	**Autumn**	**Winter**	**Spring**
Goro	1200.00	1860.00	1440.00	4100.00
Volano	3360.00	2200.00	1600.00	2288.00
Estensi	2564.00	2904.00	1113.33	1732.00
Spina	3104.00	2956.00	3100.00	2616.00
Overall	2896.25	2635.00	2052.31	2330.00

Fibers dominated across all seasons. Film particles were absent in winter and spring, and largely absent in summer at Goro, and in autumn at Goro and Spina. At Goro, fragments exceeded fibers in summer and autumn (67.32% vs. 27.45%). The typology distribution differed significantly across seasons ($\chi^2(9) = 247.90$, $p < 0.001$).

CMPI patterns varied seasonally:

- Summer–spring: The CMPI was extreme for fibers (0.79 and 0.76), average for fragments (0.15 and 0.18), and minimum for pellets and films (0.00–0.06).
- Autumn: The CMPI was extreme for fibers (0.81), average for fragments (0.14), and minimum for pellets and films (0.05 and 0.00).
- Winter: The CMPI was extreme for fibers (0.94), and minimum for other categories (0.00–0.04).

In summer and autumn, SMPs were longer (310.73 ± 255.01 µm; median 223.57 µm; 19.32–999.69 µm) than in winter and spring (253.05 ± 218.84 µm; median 170.69 µm; 16.15–985.47 µm). Black remained the predominant color (Supplementary Materials, Figure S5). SMPs were ≥54% opaque, except for some transparent peaks at Volano and Estensi in summer, and at Estensi in autumn.

4.2.2. SMPs—Cross-Shore Variations

The SMP distribution across GZs was as follows (Table 6): (i) DF: 3094.55 items/m^2 (22.33%); (ii) DC: 2610.00 items/m^2 (20.54%); (iii) SZ: 2121.43 items/m^2 (19.48%); (iv) LB: 2441.67 items/m^2 (19.22%); and (v) UB: 2341.67 items/m^2 (18.43%). A chi-square test revealed significant differences among the GZs ($\chi^2(4) = 34.51$, $p < 0.01$). According to the MPPI, all zones displayed very high SMP abundance.

Fibers dominated across all GZs. However, the SZ had slightly fewer fibers (71.85%) and more fragments (23.97%). There were significant typology differences across GZs ($\chi^2(12) = 231.43$, $p < 0.001$). According to the CMPI classification, the impact coefficient was maximum for fibers at the SZ (0.72), and extreme at all other GZs (0.81–0.86). For fragments, the impact coefficient was average at the SZ, LB, and UB (0.14–0.24), and minimum at the DF and DC (0.08 and 0.09). The impact coefficient was minimum for pellets and films across all zones (≤0.05).

Table 6. Abundances of SMPs (items/m^2) detected at each site across the different GZs. Abbreviations: SZ—swash zone; LB—lower backshore; UB—upper backshore; DF—dune foot; DC—dune crest.

	Items/m^2				
	SZ	**LB**	**UB**	**DF**	**DC**
Goro	2150.00	-	-	-	-
Volano	2313.33	2310.00	2930.00	1925.00	2510.00
Estensi	1860.00	2205.00	1870.00	2293.33	2645.00
Spina	2145.00	2810.00	2225.00	4865.00	2675.00
Overall	2121.43	2441.67	2341.67	3094.55	2610.00

SMP lengths increased from the SZ (265.50 ± 248.07 µm) to the DC (315.68 ± 234.36 µm). Widths were larger at the SZ, DF, and DC, and smaller at the LB and UB. Color distributions varied (Supplementary Materials, Figure S6), but black, colorless, and white remained predominant. SMPs were ≥62% opaque across GZs. However, transparent SMPs dominated in the SZ at Volano and Estensi, as well as in the DF at Estensi.

5. Discussion

5.1. Large Microplastics (LMPs)

In this paper, an average of 5.66 ± 13.15 LMPs/m^2 was recorded. The absence of LMPs at the Goro site is likely due to local geomorphological conditions, specifically the limited beach width and the presence of a rip-rap revetment [63]. Under these conditions, LMPs cannot accumulate and form stable deposits because they are quickly removed by incident and reflected waves and tidal flow. This pattern was also documented in other studies [90,91]. Although local peaks reached up to 81 LMPs/m^2 (Volano, summer, DC), these values are lower than those reported in heavily impacted coastal systems, such as beaches in South Korea where densities reached 976 items/m^2 in 2013 [92] and 880.4 items/m^2 in 2015 [93].

According to the MPPI, Volano, Estensi, and Spina exhibit moderate LMP abundance. Similar results were reported along the coast of Rio de Janeiro (Brazil) [94], whereas substantially higher LMP abundances were found in the Bay of Bengal (Bangladesh) [95].

At Volano, Estensi, and Spina, LMPs were mostly pellets (57%) and fragments (30%), followed by fibers (8%) and films (5%). Accordingly, CMPI values were highest for pellets, average for fragments, and minimum for fibers and films. Previous research conducted along the Po Delta coast found that LMPs were dominated by fragments (85%) and pellets (10%) [72]. Similarly, on the Vulcano island (Sicily), LMPs were dominated by fragments (56%) and pellets (33%), with foams, films, and filaments accounting for less than 5% of each [96]. Along the urban beaches of São Paulo (Brazil), foam particles were the most abundant (49%), followed by fragments (28%), pellets (23%), and fibers (0.4%) [97]. Secondary MPs formed from the fragmentation of larger items are typically more abundant in areas subjected to strong wave action and sediment abrasion [98]. In this paper, the fragments were generally longer than wide, and relatively thin (4.98 × 2.82 × 0.93 mm). These characteristics increase their fragility and susceptibility to further fragmentation.

Pellets were the predominant LMP typology. This pattern is plausibly linked to the presence of numerous industrial hubs in northern Italy and manufacturing facilities along the Po and Reno rivers. These facilities function as highly effective transport pathways to the sea. Similar findings have been reported in earlier studies, where the occurrence of pellets was attributed to nearby industrial sources [99,100]. Typical industrial pellets (spherical, cylindrical, or oval particles 1–5 mm in size) are the primary type of microplastic used as raw material for multiple plastic products [10,40,90,101–103]. Falih et al. [104] estimated

that between 3 and 36 million pellets are released annually per production facility. Both land-based and ocean-based sources contribute to pellet discharge, including losses during manufacturing, transportation, loading, and unloading operations [105]. At Costa Nova (Portugal), elevated concentrations of microspheres were linked to port activities and the role of local hydrodynamics [106]. Pellets have also been found on urban beaches far from potential emission sources, indicating long-distance marine transport [96,107]. Following the Toconao ship accident in the Northwest Atlantic Ocean in December 2023, 25 tons of pellets were lost off the Portuguese and Spanish coasts [108]. After the X-Press Pearl container ship incident off the coast of Sri Lanka in 2021, plastic pellets became the primary source of microplastic pollution, reaching a mean density of 147.14 items/m^2 [109,110]. In May 2025, the sinking of the MSC ELSA 3 container ship off the southern coast of India caused an unprecedented release of plastic granules (with peaks exceeding 2000 items/m^2) [111].

In addition to their widely distribution, pellets have characteristics that make them well-suited as indicators of chemical pollution [61]. Their small size, long persistence, and high surface-to-volume ratio allow them to absorb hydrophobic contaminants, such as polycyclic aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs), and dichlorodiphenyltrichloroethane (DDT) [112]. In Japan, PCB concentrations on PP pellets were reported to be up to one million times higher than in the surrounding seawater [112]. The degree of surface weathering also influences adsorption efficiency: eroded pellets with irregular surfaces and higher polarity accumulate contaminants more effectively than pristine pellets [113]. Aged, discolored, or blackened pellets exhibit a greater diversity of adsorbed pollutants. Discoloration (yellowing) reflects prolonged exposure to seawater, which increases the likelihood of polymer oxidation [114,115]. Due to their high surface-to-volume ratio, MPs degrade more rapidly than meso- and macrolitter because degradation processes typically begin on the polymer surface. Early indicators include color changes and the formation of microcracks [116]. Color alterations are mainly associated with UV radiation exposure. The emergence of fracture lines exposes internal layers, promoting secondary degradation and ultimately leading to increased brittleness and polymer disintegration [113,117]. In some coastal sediment studies, the pellets were predominantly white or transparent [103,118]. However, in the present paper, opaque pellets of various colors were observed, with cream, white, and black being most common. These findings are consistent with other studies [82,90,119,120], suggesting multiple potential sources [121]. The predominance of cream-colored pellets indicates prolonged environmental exposure and weathering. Investigating the color variations in items has a fundamental ecological implication because some organisms can select MPs of specific colors [10].

Raman spectroscopy revealed that LMPs were primarily composed of PE, PP, and PET, with smaller contributions of PA and PS (<6%). These results are consistent with those reported for sediments in the Venice Lagoon [122] and along the Po Delta coast [72], where polyolefins (PE and PP) dominated. Similar polymer distributions have been documented along the coasts of Tuscany (Italy) [123], Tunisia [90], Morocco [124], and the Bay of Brest (Brittany, France) [125], and in the North Atlantic subtropical gyre [126]. This pattern reflects global production scales [127]. These polymers originate from common products, such as bags, packaging films, containers, caps, and numerous industrial applications [128]. The most intensively produced polymers at the global scale include PA, PE, PET, PP, PS, PU, and PVC [129,130]. Low-density plastics, such as PE, PP, and PS, are particularly prevalent in surface waters and beach sediments due to their buoyancy and potential for long-range dispersion [9,129,131]. Depending on environmental conditions, polymers such as PE, PET, PP, PS, PU, and PVC undergo photo-thermal-oxidative degradation, as well as hydrolysis and biodegradation [116]. These processes often occur concurrently, leading to the progressive deterioration of mechanical properties accompanied by chromatic

alterations, cracking, and the formation of fracture lines [113,116,117]. PP is particularly susceptible to thermal degradation even at ambient temperatures, unless UV stabilizers and antioxidants are added [113,117].

5.2. Small Microplastics (SMPs)

This paper reports an average of 2402.19 ± 1169.85 SMPs/m^2. Spina was the most contaminated site (2944.00 items/m^2), followed by Volano (2402.11 items/m^2), Estensi (2185.56 items/m^2), and Goro (2150.00 items/m^2). The abundance of SMPs was more than two orders of magnitude higher than that of LMPs. This evidence supports the findings of previous studies that aimed to assess the proportional abundance of the two size classes. Those studies indicated that SMPs exceed LMPs by several orders of magnitude [132,133]. Although no LMPs were detected at Goro, SMPs were present, likely due to the ability of medium and fine sands to retain particles smaller than 1 mm within their interstitial spaces. Similar findings were reported in recent studies performed in southwest England [91,134]. According to the MPPI, all sites exhibit very high SMP abundance. However, the worst MPPI class (>25 items/m^2 = "very high abundance" [87]) represents a threshold approximately two orders of magnitude lower than the SMP densities observed in this paper. Thus, while the MPPI is suitable for characterizing LMPs, it is less valuable for representing environmental conditions dominated by extremely high SMP loads.

The predominant SMP typology was fibers (81.54%), followed by fragments (13.54%), pellets (4.53%), and films (0.39%). CMPI values were extreme for fibers, average for fragments, and minimum for pellets and films. These findings are consistent with a wide body of literature in which fibers are the most commonly observed MP typology across different environmental matrices. For instance, fibers dominated MP in four rivers discharging into the northern Adriatic Sea in Italy, followed by fragments [135]. Fortunato et al. [35] also found fibers to be the most common type of SMP along the Portuguese coast in sediment, seawater, and lagoon samples. Fibers were also the predominant type of MPs (77.61%) along the Mediterranean coast of Morocco [136]. Similar patterns have been reported along the Atlantic coast of Argentina, where fibers prevailed in both sand and seawater samples [137]. In wastewater samples, fibers may account for up to 53% of MPs [78]. More than 99% of particles identified in drinking water, beer, and commercial sea salt were fibers [138,139]. While fragments dominate in some sediments [140,141], the international literature consistently demonstrates the widespread occurrence of fibers in various environmental matrices, including water [120,142–147], sediments [90,134,148–153], ice [154,155], groundwater systems [84,156], and marine organisms [157–161]. Globally, the majority of coastal and marine microfibers originate from synthetic textiles [162–164]. Several authors identify washing clothes as a major source of microfibers. Release rates are influenced by fabric type, wash temperature, detergent, and washing machine characteristics [143,165,166]. For instance, Browne et al. [143] demonstrated that a single polyester jacket releases over 1900 fibers per wash. Therefore, microfiber reduction strategies should be implemented throughout the textile life cycle. These strategies include designing more durable fibers, promoting of slow-fashion consumption models, adopting filtration devices in washing machines, and improving end-of-life textile management [167–176]. Wastewater treatment plants (WWTPs) also play a key role in releasing MPs into the environment [177,178]. While WWTPs retain some microfibers, approximately 76% are incorporated into sludge, which is then used for agricultural purposes [179]. Primary and secondary treatments remove many dense MPs [78,180], whereas tertiary treatments are more effective for less dense particles [181]. Recent studies have documented removal efficiencies exceeding 90% [182,183].

In this paper, the SMP color composition was dominated by opaque (71%), black (40.18%), and colorless (20.92%) particles. However, the published papers show high

variability. Black SMPs were predominant on several European beaches [184], in southwest England [134], on the Argentine continental shelf [185], and in mussels from the Goro Lagoon [160]. Transparent, blue, and red SMPs were most common in mangrove ecosystems in Singapore [90,137]. A recent review of 26 studies conducted along the Canary Islands reported that colorless, white, and transparent SMPs predominated [82]. Hidalgo-Ruz et al. [17] also identified white and similar shades as common colors. Browne et al. [143] noted that white and transparent SMPs may reflect long residence times in the marine environment.

The SMPs analyzed in this paper had mean dimensions of 256.99 ± 243.78 µm. The predominance of particles of this size is ecologically significant because small dimensions may facilitate trophic transfer [90]. SMPs of this size can easily be mistaken for natural food items by planktivorous organisms [39]. Numerous studies documented SMP ingestion by marine invertebrates and other taxa [29,31,74,76,160,186–189]. Particles smaller than 130 µm can accumulate in human tissues and can be transported through the bloodstream, lymphatic system, and various organs, via ingestion, inhalation, and dermal absorption [187]. Furthermore, their high surface-to-volume ratio enhances the adsorption of hydrophobic chemicals in aquatic environments. SMPs can act as vectors for hazardous contaminants, including PTEs, PAHs, PCBs, pesticides, and persistent organic pollutants (POPs) [190,191]. Consequently, the potential implications for human health are broad and complex: MPs may release toxic substances, additives, and monomers that can exert carcinogenic effects or induce inflammatory responses [192].

5.3. Considerations on Seasonal and Cross-Shore Variations

The abundance of MPs varied across sites, seasons, and GZs, and these patterns differed between the two MP classes investigated. These findings suggest that MP dispersion is strongly influenced by local hydrodynamic processes, such as tides, coastal morphology, riverine inputs, and natural factors, including waves, winds, and storms [35,193–195]. Sediment particle size also appears to be a key parameter that regulates MP accumulation along coastlines. For instance, Lo et al. [196] found that MP deposition is favored in fine sands, which are characteristic of low-energy environments (see also [197,198]). In contrast, Ling et al. [199] found a positive relationship between MP abundance and wave exposure. Similarly, Fortunato et al. [35] observed that tidal and wave energy can enhance both MP deposition and dispersion in coastal systems. These apparent contradictions reflect the pronounced spatial and temporal variability of coastal environments [188], where MPs may accumulate during depositional phases and be remobilized during erosional events [200,201].

The sediment grain size at the investigated sites was recently characterized and consists of medium and fine sand, which is typical of low-energy coastal settings [64]. In the present paper, LMP and SMP abundances were lowest in winter and peaked in spring and summer, respectively. These patterns can be explained by local coastal dynamics. Material supply along the coast is governed by local hydrodynamics, including the combined action of waves, tides, longshore currents, and river discharge [202,203]. The energetically intense winter conditions, which are characterized by adverse weather, strong wave action, and multiple storm events that affect the study area [204,205], may negatively impact MP abundance, causing a reduction. Conversely, the increased river discharge recorded between March and June of 2024 [205] may have contributed to the enhanced MP inputs along the coast, as observed during the spring and summer. These results align with previous studies demonstrating that local hydrodynamics strongly influence MP distribution along coastlines [195,206–209]. In the Adriatic Sea, longshore circulation and thermohaline fronts can trap particles near the coast [122,210]. Three-dimensional hydrodynamic models applied

to the Po Delta demonstrate that MP dispersion is highly influenced by river discharge, wind patterns, and coastal currents [211]; the same modeling study identified accumulation zones at the river/sea interface and estimated that MPs released by the Po River require approximately ten days to reach the Delta coastline. Similar trends have been observed in other regions where coastal morphology and river mouths are key drivers of MP transport [193,194]. Furthermore, the high proportion of cream- and white-colored LMPs, which were often yellowed or discolored, indicates that these items have persisted in the environment for prolonged times. The seasonal color analysis aimed to determine if variations in color composition reflected recent LMP inputs (i.e., fresh litter). However, the results did not reveal substantial seasonal differences in color distribution. The cross-shore analysis aimed to identify zones characterized by the persistence of weathered items or the presence of items with specific color signatures. In the SZ, white items predominated, followed by blue and green items. In the adjacent zone (LB), colored items (e.g., blue, black, and green) were less abundant than discolored or yellowed items, though they were still present at proportions of at least 10%. In the landward zones (UB, DF, and DC), cream and white items clearly dominated all other colors. These patterns suggest that coastal dynamics that control sediment supply and redistribution (e.g., geomorphology, wave action, tides, wind, and river flooding) also influence the transport, deposition, and long-term persistence of LMPs.

Regarding the cross-shore distribution, the abundance of LMPs increased progressively from the SZ toward the DC, where the highest concentrations of all item typologies were observed, particularly pellets and fragments. Similar patterns have been reported on Vulcano Island (Sicily), where an increase in LMP abundance from the shoreline to the dune was attributed to wind-driven transport and wave action [96]. However, a previous study in the Po Delta revealed a non-uniform cross-shore accumulation pattern, with increasingly heterogeneous LMP distribution in the UB, DF, and DC [163]. Following the X-Press Pearl incident in Sri Lanka, investigations identified the high-tide line as the zone with the greatest pellet accumulation [109]. Other studies have also documented that, due to their low density and spherical shape, pellets tend to accumulate along the high-tide line and in the upper intertidal zone [109,212–214]. Along Australian coasts, smaller particles predominate in the swash zone [215]. However, variability in tidal cycles and shoreline position can generate substantial cross-shore heterogeneity [216]. In the present paper, SMP abundances were highest in the DF and DC, decreasing from the SZ toward the UB. Similar trends were observed in fine sands investigated at the same sites [64]. MPs and coastal sediments are subject to the same forcing factors, geomorphology, hydrodynamics, and aeolian processes [214,217–223]. High-energy incoming waves can transport MPs as far as the ordinary berm or even the storm berm, while weaker backwash flows may carry back smaller MPs. Aeolian transport and storm waves can move MPs into the landward zones [152]. In these areas, morphological features such as dunes, vegetation, and accumulated debris can act as natural barriers or increase the likelihood of MP entrapment [105,224]. Numerous studies have reported an overall trend of increased MP abundance in landward zones [92,105,224–227].

5.4. General Considerations

This paper demonstrated the presence of MPs at all investigated sites, with notable seasonal and cross-shore variation that differed between dimensional classes. The large standard deviations observed for both LMPs and SMPs reflect the high spatial and temporal variability characteristic of MP contamination across different sites, seasons, and GZs. Nevertheless, the results clearly confirm the pervasive presence of MPs in coastal environments, as highlighted by Auta et al. [228]. The site-dependent distribution of MPs

reflects the combined influence of natural dynamics and anthropogenic pressures [188]. It is estimated that global plastic production will rise to 753 Mt by 2060, with total MP emissions reaching nearly 298 Mt over the same period [55]. Using integrated protocols aimed at extracting specific dimensional classes of MPs, allow for accurate and highly detailed estimates of environmental contamination from MPs.

Northern Italy, including the Po Plain and the basins of the Po and Reno rivers, hosts numerous industries involved in plastic production, regeneration, processing, and recycling. While recycling is an important mitigation strategy, many recycling operations (e.g., washing, mechanical shredding, pellet production) involve handling and reprocessing plastic waste and may unintentionally release secondary MPs into the environment [55]. Since the study sites lie downstream of the Po and Reno basins, it is plausible that urban and riverine discharges, along with emissions from plastic processing industries, are likely substantial contributors to the observed MP abundances. Secondary MPs generated in situ from the degradation of larger debris also play a relevant role. A recent paper investigated stranded macrolitter at the same sites using a similar seasonal and cross-shore sampling approach [63]. This approach provides valuable additional insight into the obtained findings. While mechanical or manual cleaning operations can influence the distribution of stranded macrolitter, these practices have minimal effects on MPs, because cleaning technologies cannot retain items smaller than 5 mm [229]. These practices mainly help prevent the formation of secondary MPs [10].

In addition to physical processes, numerous studies reported strong associations between MP abundance and anthropogenic pressure. Higher MP concentrations have been detected in densely populated coastal regions [61,143]. Highly urbanized coastlines around the world exhibit elevated levels of MP contamination, including the Mediterranean [72,122,136,230], Northern Europe [177,226,231], South America [107,232], and Asia [43,82,95,233]. Elevated concentrations of MPs have also been reported in port sediments and riverine zones that receive industrial and urban discharges [61,226,234]. Rivers act as major vectors, transporting MPs from inland sources to the sea and coast, as many particles can bypass wastewater treatment filtration systems [135,225,235,236]. Therefore, it is unsurprising that up to 80% of MPs in coastal sediments originate from wastewater effluents, and that rivers globally discharge millions of tons of plastic into the ocean each year [143,237].

The investigated sites are located within an area of high ecological, tourist, and economic importance, where fisheries, aquaculture, and agriculture are key sectors of the local economy [63,64]. The areas crossed by the Po and Reno rivers are primarily used for agriculture and livestock farming, while the adjacent coastal lagoons are dedicated to extensive shellfish farming activities [238–243]. The interaction of multiple stressors affecting these transitional environments may trigger cascading effects on ecosystem health, human societies, and local economies. From a One Health perspective [244], it is crucial to understand how anthropogenic pressures influence ecosystems, food products (e.g., fisheries and aquaculture resources), and human health.

The presence of MPs in the environment poses significant ecological risks because these particles can be ingested by living organisms and transferred through food web [90,157,245–247]. The ingestion of MPs by marine vertebrates and invertebrates, including commercially important species, is well documented [3,74,76,188,248–253]. Once ingested, MPs can accumulate in organs and tissues, causing adverse effects and, in extreme cases, mortality [29,98,186,189,254–258]. Plastic additives can exacerbate these impacts, as they may be toxic, carcinogenic, mutagenic, or act as endocrine disruptors [176]. Furthermore, environmental exposure promotes the adsorption of additional contaminants onto MP surfaces. Consequently, ingesting MPs may facilitate the bioaccumulation and/or

biomagnification of substances capable of inducing ecotoxicological effects, metabolic alterations, endocrine disruption, and behavioral changes [98,186,259]. This process leads to the transfer of associated contaminants to higher trophic levels, ultimately reaching humans [10,260–264]. Humans are primarily exposed to these complex mixtures by ingesting MP-contaminated water and food, including fish, crustaceans, and mollusks [265]. Due to the extent of exposure, MPs were recently detected in human organs, tissues, and body fluids [266–271]. At the cellular level, MPs may induce oxidative stress, cytokine release, and cellular and DNA damage. They may also promote inflammatory responses and exert neurotoxic effects, thereby increasing the risk of carcinogenesis [272]. Prolonged exposure may also affect the respiratory and pulmonary systems, disrupt endocrine regulation, and impair hepatic and cardiovascular functions, potentially leading to chronic toxicity [273].

6. Conclusions and Future Perspectives

This paper provides an integrated assessment of the seasonal and cross-shore distribution of large and small microplastics (LMPs and SMPs) along the Ferrara coast. By combining specific extraction protocols, polymer identification via Raman spectroscopy, and an analysis of seasonal and geomorphological dynamics, this research provides a detailed and high-resolution framework of MP contamination in a coastal environment that lies between the Po and Reno rivers' mouths.

Due to the proximity of the study area to major plastic-processing industries and densely populated areas, such as those in northern Italy, it is likely that a combination of urban and fluvial discharges, industrial losses, and in situ fragmentation of macrolitter likely contributed to the observed contamination patterns.

The integration of MP surveys with macrolitter monitoring and sediment analyses represents a robust methodological approach for environmental assessment. The long-term datasets derived from these integrated strategies may serve as valuable tools for evaluating whether good environmental status (GES) is maintained or achieved. Strengthening these monitoring efforts is particularly important in dynamic coastal systems, such as the northern Adriatic, where hydrodynamic variability can rapidly alter deposition patterns.

Future perspectives should consider adopting a "weight of evidence" approach that combines in situ surveys, aerial imaging, and land-use mapping. This approach will provide an additional level of detail for environmental assessments, and facilitate the identification of potential release sources near the investigated sites.

Expanding polymer identification to SMPs is essential for assessing risks to benthic communities, trophic transfer, and potential human exposure (One Health vision).

Reducing MP contamination requires minimizing plastic inputs at the source, improving industrial practices and wastewater treatment, and implementing targeted policy interventions and coastal management strategies. Integrating scientific evidence with circular-economy strategies and public awareness campaigns is crucial for mitigating future impacts.

Supplementary Materials: The following supporting information can be downloaded at: https://www.mdpi.com/article/10.3390/microplastics5010015/s1, Figure S1: Percentage color composition of collected LMPs; Figure S2: Percentage color composition of LMPs across seasons: (a) summer; (b) autumn; (c) winter; (d) spring; Figure S3: Percentage color composition of LMPs across geomorphological zones: (a) swash zone; (b) lower backshore; (c) upper backshore; (d) dune foot; (e) dune crest; Figure S4: Percentage color composition of collected SMPs; Figure S5: Percentage color composition of SMPs across seasons: (a) summer; (b) autumn; (c) winter; (d) spring; Figure S6: Percentage color composition of SMPs across geomorphological zones: (a) swash zone; (b) lower backshore; (c) upper backshore; (d) dune foot; (e) dune crest; Figure S7: Fiber: colorless and transparent; Figure S8: Fiber: black and opaque; Figure S9: Fiber: blue and opaque; Figure S10: Fiber: black and

opaque; Figure S11: Fiber: black and opaque; Figure S12: Fiber: blue and opaque; Figure S13: Fiber: red and transparent; Figure S14: Fiber: multicolor and opaque; Figure S15: Fiber: blue and opaque; Figure S16: Fragment: white and opaque; Figure S17: Fragment: white and opaque; Figure S18: Fragment: multicolor and opaque; Figure S19: Fragment: black and opaque; Figure S20: Fragment: black and opaque; Figure S21: Fragment: white and opaque; Figure S22: Fragment: blue and opaque; Figure S23: Fragment: creme and opaque; Figure S24: Fragment: black and opaque; Figure S25: Fragment: brown and transparent; Figure S26: Fragment: blue and opaque; Figure S27: Fragment: white and opaque; Figure S28: Pellet: colorless and transparent; Figure S29: Pellet: gray and opaque; Figure S30: Pellet: black and opaque; Figure S31: Pellet: creme and opaque; Figure S32: Pellet: black and opaque; Figure S33: Pellet: black and opaque; Figure S34: Film: green and transparent; Figure S35: Film: colorless and transparent; Figure S36: Film: black and transparent.

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Abbreviations

The following abbreviations are used in this manuscript:

DC	Dune Crest
DDT	Dichlorodiphenyltrichloroethane
DF	Dune Foot
dGPS	Differential Global Positioning System
GZ	Geomorphological Zone
LB	Lower Backshore
LMP	Large Microplastic
MP	Microplastic
MSFD	Marine Strategy Framework Directive
NaCl	Sodium Chloride
PA	Polyamide
PAH	Polycyclic Aromatic Hydrocarbons
PCB	Polychlorinated biphenyl
PE	Polyethylene
PET	Polyethylene Terephthalate
POP	Persistent Organic Pollutants
PP	Polypropylene
PS	Polystyrene
PTEs	Potentially Toxic Elements

PVC	Polyvinyl Chloride
QA/QC	Quality Assurance and Quality Control
SD	Standard Deviation
SMP	Small Microplastic
SZ	Swash Zone
UB	Upper Backshore
UTM	Universal Transverse Mercator
UV	Ultraviolet Radiation
WGS	World Geodetic System
WWTPs	Wastewater treatment plants
χ^2	Pearson's chi-square test

Appendix A

Table A1 reports the Raman spectroscopy analysis data. In some cases, the pigments used to color the polymers were detected. This information is included in the notes.

Table A1. Specifications of each LMP item analyzed via Raman spectroscopy, including sample identifiers, item typology, polymer type, and any additional notes.

ID	Type	Polymer	Note
01	Pellet	PE	
02	Pellet	PE	
03	Pellet	PA	
04	Pellet	PE	
05	Pellet	PS	
06	Pellet	PET	
07	Pellet	PE	
08	Pellet	PE	
09	Pellet	PE	
10	Pellet	PP	
11	Pellet	PP	
12	Pellet	PP	Hostasol Green G-K
13	Pellet	PE	
14	Pellet	PP	
15	Pellet	PA	
16	Pellet	PE	
17	Pellet	PE	
18	Pellet	PA	
19	Pellet	PE	
20	Pellet	PE	
21	Pellet	PE	
22	Pellet	PP	
23	Pellet	PP	
24	Pellet	PE	
25	Pellet	PE	
26	Pellet	PE	
27	Pellet	PET	Hostasol Green G-K
28	Pellet	PE	
29	Pellet	PET	
30	Pellet	PE	
31	Pellet	PS	
32	Fragment	PP	
33	Fragment	PE	
34	Fragment	PET	
35	Fragment	PP	
36	Fragment	PE	
37	Fragment	PET	
38	Fragment	PET	
39	Fragment	PET	Pigmosol Green
40	Fragment	PE	
41	Fragment	PP	
42	Fragment	PP	
43	Fragment	PE	
44	Fragment	PET	
45	Fragment	PP	Hostasol Green G-K
46	Fragment	PET	
47	Fragment	PE	

Table A1. *Cont.*

ID	Type	Polymer	Note
48	Fiber	PE	Pigment red
49	Fiber	PP	
50	Fiber	PP	
51	Fiber	PP	
52	Film	PE	
53	Film	PE	
54	Film	PE	

Figure A1 displays the Raman spectra of the most common polymers identified in the analyzed LMPs.

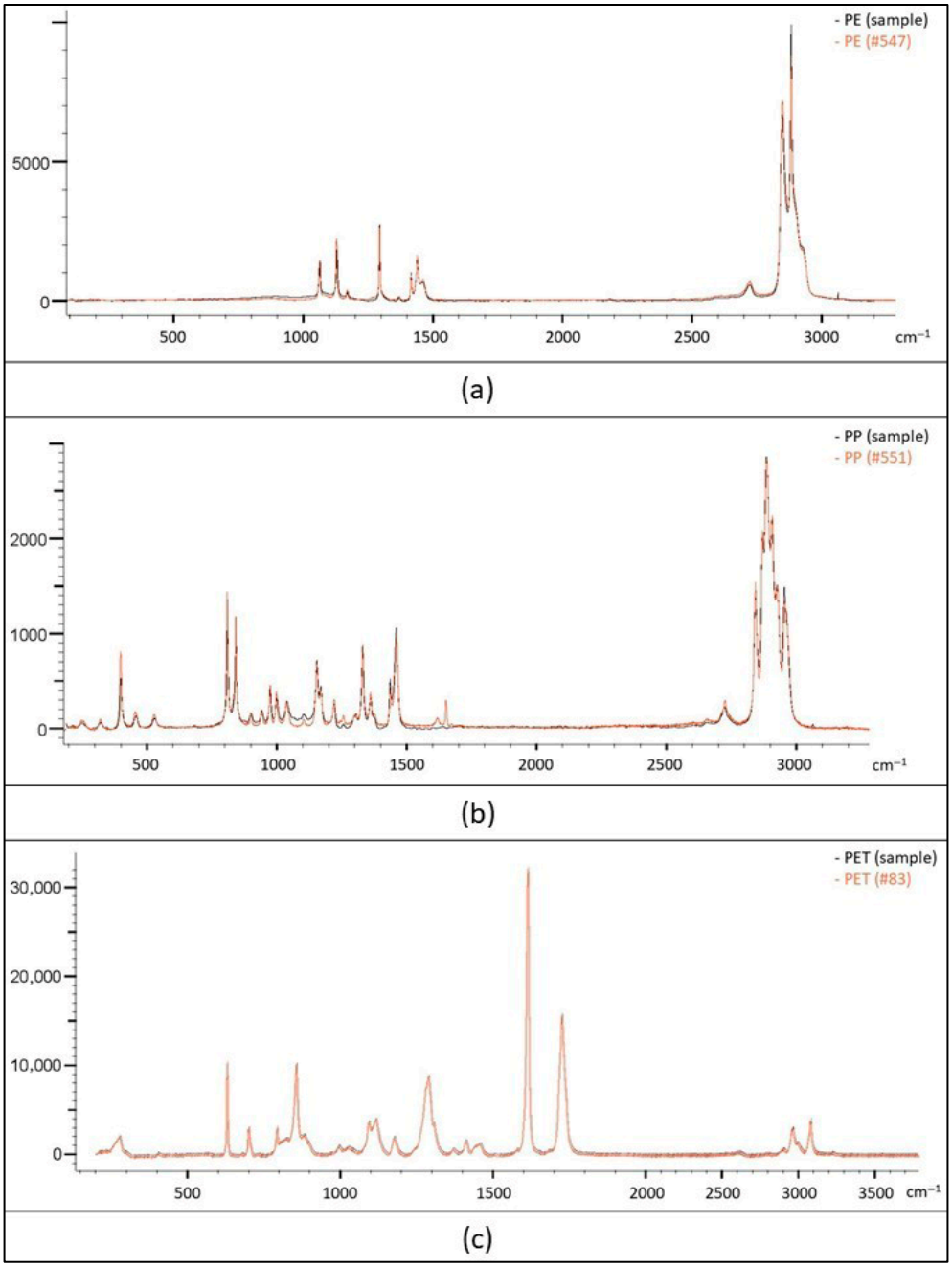

Figure A1. Representative Raman spectra of the most common polymers identified among the analyzed LMPs: (**a**) PE; (**b**) PP; (**c**) PET.

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