applied sciences

MDPI

Article

What Are Lake Beaches Made of? An Assessment of Plastic Beach Litter on the Shores of Como Bay (Italy)

Arianna Bellasi [1], Gilberto Binda [1,2,*], Ginevra Boldrocchi [3], Andrea Pozzi [1] and Roberta Bettinetti [3,*]

1 Department of Science and High Technology, University of Insubria, Via Valleggio 11, 22100 Como, Italy; abellasi@uninsubria.it (A.B.); andrea.pozzi@uninsubria.it (A.P.)
2 Norwegian Institute for Water Research (NIVA), Økernveien 94, 0579 Oslo, Norway
3 Department of Human and Innovation for the Territory, University of Insubria, Via Valleggio 11, 22100 Como, Italy; ginevra.boldrocchi@uninsubria.it
* Correspondence: gilberto.binda@niva.no (G.B.); roberta.bettinetti@uninsubria.it (R.B.)

Featured Application: This research aims at understanding the sources of plastic litter in a freshwater body, assessing the possible production of secondary microplastics (MPs).

Abstract: Plastic waste dispersion is a well-recognized environmental threat, despite continuous efforts towards improving waste disposal management over the last few decades. Plastic litter is known to strongly impact upon water bodies and shorelines, affecting the health of ecosystems and impacting upon the aesthetic value of sites. Moreover, plastic waste that is abandoned on beaches contributes towards different degradation processes that potentially lead to the formation of secondary microplastics (MPs), with likely cascade effects upon the whole ecosystem. In this view, this study aims to characterize the plastic beach litter found on the shores of the western basin of Como Lake (Italy) to better understand the origin of MPs in littoral sediments, including the recognition of object typologies and the chemical characterization of polymers using Fourier-transformed infrared analysis (FTIR). The results highlighted that the most abundant polymers on beaches are polypropylene (PP) and polyethylene (PE), representing 73% of the collected polymers. This confirms that floating, low-density polymers are more likely to accumulate on beaches. Moreover, almost 66% of litter is represented by commonly used manufactured items (disposable objects, packaging, and everyday items). This evidence, combined with the analysis of the main environmental features of the sampling sites (the main winds, distance to urban areas, and the presence of tributaries) indicate that abundance of beached litter is mainly linked to beach accessibility and the local winds. These results highlight that multiple factors affect the environmental fate of plastic litter and give insights into the assessment of secondary microplastics in beach sediments.

Keywords: plastic litter; freshwater; sediments; secondary microplastics

Citation: Bellasi, A.; Binda, G.; Boldrocchi, G.; Pozzi, A.; Bettinetti, R. What Are Lake Beaches Made of? An Assessment of Plastic Beach Litter on the Shores of Como Bay (Italy). *Appl. Sci.* **2022**, *12*, 5388. https://doi.org/10.3390/app12115388

Academic Editor: Anna Annibaldi

Received: 13 April 2022
Accepted: 20 May 2022
Published: 26 May 2022

1. Introduction

Many scientists suggest that we are currently living in a "plastic age, the Plasticene" [1]. Plastic is a durable, versatile, and low-cost material that has completely changed human life and represents the most produced and used material in the world.

European plastic production represents 16% of global production, with a plastic demand of 50.7 million tons and an average consumption of 112 kg/person for the period of 2010–2019 [2,3]. Within Europe, Italy covers 13.8% of the total demand, preceded by Germany (which represents 24.2%).

Approximately 30,000 polymer materials are currently registered in the European Union [4]. However, the most commonly produced ones are polypropylene (PP), polyethylene (PE), polyvinylchloride (PVC), polyethylene terephthalate (PET), polyurethane (PUR) and polystyrene (PS). These polymers are increasingly being used for an abundant variety of objects. Moreover, the global SARS-CoV-2 pandemic has led to a further increase in plastic production and usage (e.g., for disposable personal protective equipment [5]).

Although waste disposal management has improved over the last decade (an increase of 19% of collected waste for recycling from 2006) [3], a large portion of these materials is still dispersed in natural ecosystems. An increasing number of reports on a global scale are warning of the threats derived from plastic litter in the environment [6–8].

After their spread, plastic polymers are subject to a series of ageing processes, such as: UV radiation exposure, oxidative reactions, mechanical deterioration (due to frictional forces), and biofouling [9]. These processes, in turn, can lead to polymer embrittlement [10,11], further promoting the release of secondary microplastics (MPs, [12]): particles with sizes ranging from a few micrometers to 5 mm [13,14]. These particles, due to their small dimensions and high reactive surface areas, can easily adsorb water contaminants and trace metals, as well as release chemical additives, causing adverse ecotoxicological effects on organisms [15–17].

The uptake of MPs by various aquatic organisms has been well demonstrated by several studies [18–22]. This effect can affect the entire food web [23], also posing threats to human health [24]. Specifically, MPs can enter organisms mainly via ingestion and respiration [25], causing physical damage and negatively affecting physiological functions. Indeed, ingested MPs induce a sense of satiety in organisms, causing a reduced uptake of food and a consequent decrease in energy intake [26]. Moreover, other risks are posed by MPs that are increased in already contaminated water environments (e.g., [27,28]) due to the interaction of plastic particles with waterborne micropollutants, which potentially enhances their accumulation in aquatic organisms. In this context, different studies have reported on the adsorption of hydrophobic pollutants onto plastic pellets [29,30], and show that MPs can act as vectors of contaminants and favor their ingestion [31,32].

In addition to its ecotoxicological and aesthetic consequences [33], plastic litter dispersion is an important issue in the context of environmental welfare and policies, and consequently, a guidance document was written in 2013 (within the Marine Strategy Framework Directive, MSFD) for the monitoring of marine litter in the European seas [34]. However, no effort has been made in the context of inland water systems [35].

The fate of plastic waste in water ecosystems depends on the plastics' properties (the dimension, shape, density, and level of ageing) and different natural processes (e.g., main currents, winds, or bioturbation). Normally, lighter plastics (e.g., PE) tend to float in water, while heavier ones (e.g., PVC) are expected to settle [36]. However, different processes can alter this behavior: the biofouling of plastic has been demonstrated to affect buoyancy rates and, on the other hand, bioturbation and water movements have been observed to re-suspend dense polymers [37]. Therefore, while reports of floating plastic waste deposed on beaches (i.e., beach litter) are increasingly being published [38–40], the dynamics leading to the release of plastic materials in beaches are generally overlooked, hindering the reconstruction of their sources and pathways [41–43]. Understanding the processes leading to the initial accumulation and preliminary degradation of plastics on shorelines can then lead to a better understanding of the likelihood of secondary microplastic generation, as well as their potential polymeric composition [7].

In this study, therefore, different beaches of the Como Bay (on the shores of lake Como, Lombardy, Italy) were surveyed. The present study area was selected because Lombardy region counts 20,000 tons of plastic waste produced every month [44] and present an abundant diffusion of lacustrine ecosystems (representing >50% of surface waters). Moreover, this area is also poorly investigated regarding the impacts of plastic pollution [39]. Thus, this study aims to (i) identify sources of plastic waste on beaches near Como city, (ii) assess the most likely processes causing its transport and accumulation, and (iii) understand the possible polymeric nature and the grade of degradation of collected plastic objects in beach sediments, in order to potentially understand the main sources and polymeric composition of secondary MPs in the study area. This will help to develop an efficient analytical protocol for extracting MPs from sediments [29,31].

2. Materials and Methods

2.1. Study Area and Litter Sampling

Sampling was carried out in Como Bay, which is located in the southern end of the western basin of Lake Como (Figure 1a,b). In this basin, Lake Como has no outlet [45]: the only outlet for Como Lake is the Adda river, that flows instead in the eastern basin (near Lecco city).

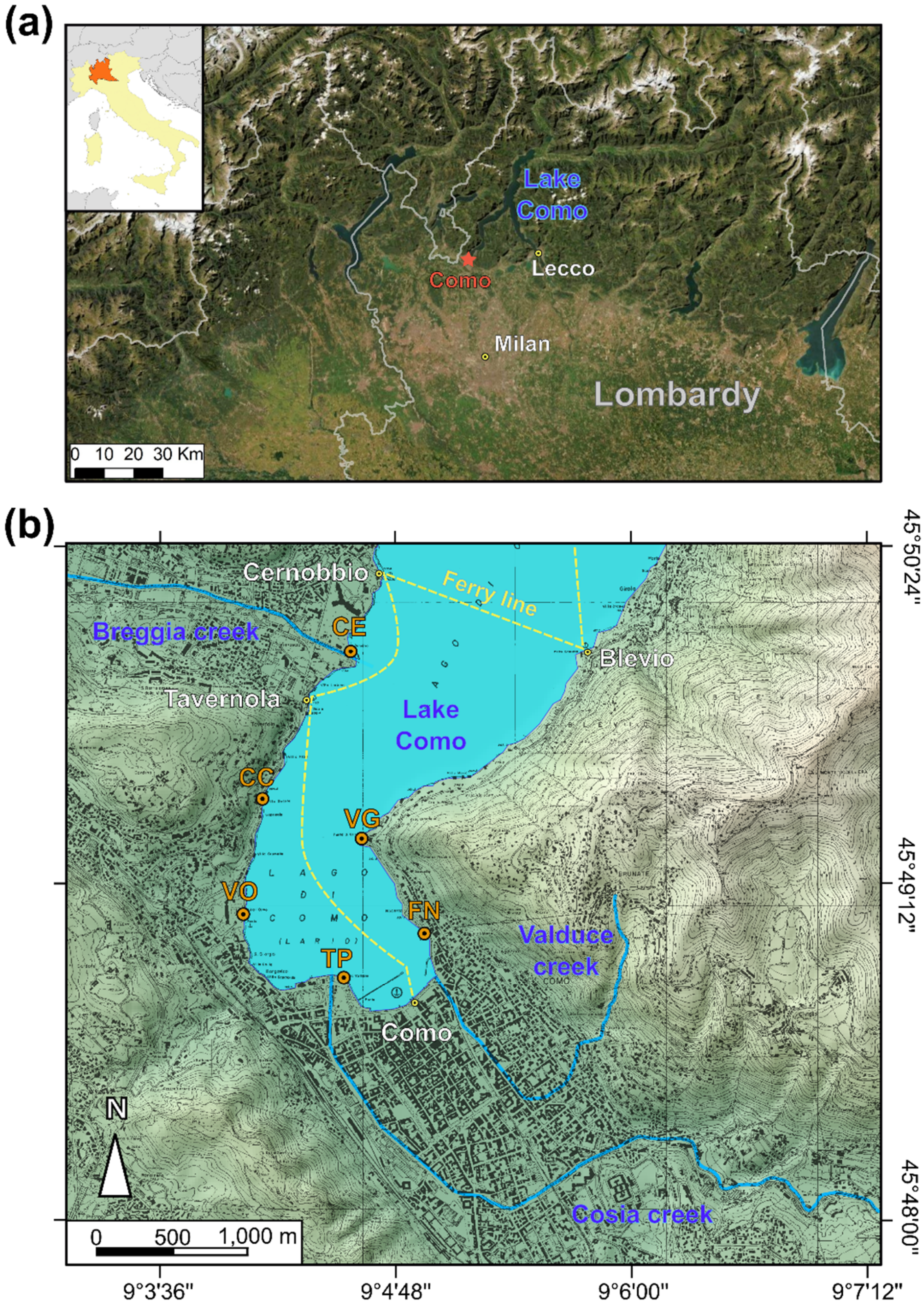

Figure 1. (**a**) Satellite view of the Lake Como area. The location of Lombardy region is highlighted in orange in the map in the upper left corner. The main cities are indicated in orange, and Como (the study area) is highlighted in red. (**b**) Detail of the sampling area of Como, located in the southern west Como branch: sampling points are highlighted in orange, main towns are indicated by yellow dots, and the yellow dashed line indicates the main route of the local ferries. Main tributaries of lake Como in the area are also highlighted.

Como city is instead drained by two streams, Valduce creek and Cosia stream (Figure 1b), which are presently artificially buried, and have an underground flow path through the urban area until it reaches the lake. These hydrological features can be important sources of plastic litter [46]. Valduce creek provides an ephemeral flow, and is mainly recharged by a small karst basin, whereas Cosia stream drains a larger watershed area (about 33.5 km^2) with measured flows of up to 33 m^3/s during its maximum flow regime. Moreover, in the area of Cernobbio town, the Breggia stream outflows into the lake after a 12 km path. This stream collects water from the western border of Lombardy and part of Switzerland (with a watershed area of about 47.4 km^2).

The demographic data of the study area can give insights into the littering sources: human pressure, in fact, has been identified as being the main cause of water pollution worldwide [47]. In this area, the principal municipalities are represented by Como and Cernobbio. According to the data for 2021, as reported by the National Institute of Statistics [48], the populations of these cities were 84,250 and 6498, with a density of 2300 and 536 inhabitants/km^2, respectively [48]. Moreover, this area is a known tourist destination: in the case of the Como Bay, the highest number of visitors occurs in spring and fall, with a weekly increase during the weekends [47].

According to the most widely applied dimensional categorization of plastic fragments [49], this research work exclusively focused on plastic litter with dimensions >5 mm [50] and that were recognizable to the naked eye. Plastic beach litter was collected in 5 different locations on the lake shores (Figure 1b and Table 1). All beaches were surveyed during May 2021 and, for each one, the extension (in square meter) was registered. Beach litter was sampled along parallel transects, until the whole beach area was inspected. To determine the predominant factors in the deposition of plastic beach litter, for each location, the following features were considered: the distance from the urban centers, public accessibility, the presence of tributary streams, and the exposure of the beach. The characteristics of the sampling sites are recorded in Table 1.

Table 1. Principal characteristics of sampling sites.

Sampling Site	Coordinates	Geographic Exposure (°N)	Area (m^2, ±Standard Deviation)	Distance from Principal Urban Centers (km)	Public Accessibility	Tributaries
VG	45°49′22″ N–9°04′39″	S–W	200 ± 15	1.5 (from Como city center)	Yes	-
FN	45°49′02″ N–9°04′57″ E	W	61.9 ± 8	0.8 (from Como city center)	Yes	-
TP	45°48′54″ N–9°04′30″ E	N–E	173 ± 20	1 (from Como city center)	Yes	Cosia creek
VO	VO1: 45°49′06″ N–9°04′00″ E; VO2: 45°49′09″ N–9°03′59″ E	E	391 ± 32	2 (from Como city center); 2.6 (from Cernobbio)	No (Private)	-
CC	45°49′22″ N–9°04′04″ E	E	101 ± 14	2.4 (from Como city center); 2.2 (from Cernobbio)	Yes	-
CE	45°50′04″ N–9°04′34″	E	2879 ± 43	900 (from Cernobbio)	No (Private)	Breggia creek

Since the data of the main flows are not available and the main currents are mostly constant during the season, the likely movement of the plastic litter was assessed, considering other natural and anthropic factors. Firstly, the wind speed and the velocity data in Como city (from Como station, located about 400 m away from the TP sampling point) were collected for the whole year of 2021 from the Lombardy environmental protection agency [51]. Another disturbing factor by waves, i.e., the passage of ferries, was also analyzed in this study. Indeed, this area of the lake is supported by an extensive navigation network that connects the city of Como with other towns on the lake shore (i.e., Cernobbio, Tavernola, and Blevio, Figure 1b), increasing wave rates toward the coast, especially in E–W direction.

2.2. Classification and FTIR Analysis of Collected Objects

All detectable plastic litter was collected in plastic bags on each beach, and the bags were labelled. Afterwards, plastic litter was brought into the laboratory to sort all the objects according to their polymer type (after visual sorting or FTIR analysis), size (measured as the length of the object on the longest axis), and the category of objects (e.g., food packaging, toys, bottles, Figure 2). More specifically, each sampled object was categorized in terms of the function of its use, as (i) disposable objects, (ii) everyday objects, and (iii) packaging. Polymeric material such as Styrofoam and polyurethane were classified as "packaging" and "building material", respectively. Plastic pieces and fragments of unclear sources were classified as "unknown material" instead.

Figure 2. Example of litter sorting and categorization (from sampling point CE).

According to the dimensions of the considered plastic objects (>5 mm), the possibility of sample contamination by the bags was ruled out after an evaluation of the bag integrity after sample categorization.

To characterize the polymer type, some objects were directly recognized using visual inspection, due to the presence of their recycling code. For other samples, attenuated total reflectance infrared spectroscopy (ATR-FTIR) was performed to recognize the polymer matrix, due to the fingerprinting of the absorption of specific infrared wavelengths by the polymeric functional groups [35]. Spectra were obtained after rinsing the objects with ultrapure water, to remove loosely adhered material. Then the samples were left to air-dry for 24 h and were analyzed with a Thermo Scientific™ (Waltham, MA, USA) Nicolet™ iS™ 10 FTIR Spectrometer, performing 32 scans in the 4000–650 cm^{-1} spectral range. The achieved spectra were then compared with those collected in the Open Specy database [52] to recognize the sample. To validate the polymer recognition, the robustness of the Pearson correlation coefficient (R) was considered: polymer matching was assumed to be accurate only for R values > 0.75.

A detailed list of sampled items is reported in Supplementary Materials (Table S1).

2.3. Data Analysis and Interpretation

After polymer type determination, the relative abundance of the different polymers was calculated for each sampling site, and for the total amount of plastic collected.

Each sampling site was geolocated, and the areas in m^2 were estimated using Google Earth. To rule out the effects of water level changes on beach extensions, measures were performed using different satellite images over different time periods (as well as the on-site measurements, see Section 2.1), taking into account the standard deviation of the beach extension as derived by the seasonal changes in level (Table 1). The density of the plastic litter (expressed as items/m^2 of beach) was then calculated. This parameter was then interpreted, considering its distance from principal urban centers, public accessibility to the sampled areas, the presence/absence of tributaries, and the exposure of the sites (which can give insights that are related to related to winds and main currents). A correlation matrix was then computed with Origin 2018 software (OriginLab—Northampton, MA, USA).

Then, for further support of the evaluation of the environmental factors affecting litter dispersion, the spatial distribution of the plastic litter and the specific abundance was spatially analyzed through QGIS software.

3. Results and Discussion

3.1. Overall Litter Density and Sample Composition

A total of 352 items were collected from the six sampling sites (Table S1). By considering the absolute number of items, the selected beaches showed different amounts of plastic litter, as follows: CE > VG > VO > TP > FN > CC. However, the normalization of the results in terms of litter abundance for m^2 demonstrated the actual impact of the plastic litter in the Como area. In fact, the beach showing the highest absolute number of collected items (CE, 120 items) was instead a less-impacted beach in terms of its normalized abundance (0.06 items/m^2). On the other hand, the most threatened site was represented by VG, with a plastic beach litter density of 0.94 items/m^2 (which instead showed an absolute abundance of 112 plastic objects).

Since data on beach plastic litter in freshwater ecosystems are still limited [53], we compared our results with some other findings in both freshwater [54–58] and marine environments [41–43] (Table 2). However a direct comparison is still complex, due to the differences in the sampled litter and the selected ecosystems [42,43,57]. As for other plastic-related environmental analyses, there is a need for harmonization regarding the sampling and data treatments. Considering these limitations, our data appear to be generally consistent with those reported in other studies [41,54]. Specifically, the abundance of PL detected in the present study is in line with those observed in Lake Tollense, which among the considered sites, is the closest to our study area.

Table 2. Comparison between data on plastic litter (PL) and anthropogenic litter (AL) detected in other studies.

Location	Type of Litter	Litter Density	References
Como Bay, Italy	PL	0.06–0.94; mean 0.14 items/m^2	Present study
Black Sea	AL	0.085–5.058 items/m^2	Topçu et al., 2013 [43]
Lake Tollense, Germany	AL	mean 0.2 ± 0.1 items/m^2 (72% plastic litter)	Hengstmann and Fischer, 2020 [54]
Lake Lewisville, Texas	PL	0.024 items/m^2	Chapman, 2019 [55]
Lake Michigan	AL	mean 0.0092 items/m^2	Hoellein et al., 2015 [56]
Central Caribbean Coast, Colombia	PL	mean 5.38 items/m^2	Rangel-Buitrago et al., 2020 [42]
Ceuta, Spain	AL	0.063–1.96 items/m^2 (32.5% plastic litter)	Asensios-Montesinos et al., 2021 [41]
Puerto Mishualli, Amazonia	PL	mean 0.045 items/m^2	Lucas-Solis et al., 2021 [57]
Setúbal Lake, Argentina	PL	1.26 items/m^2	Blettler et al., 2017 [58]

Considering the polymeric composition of plastics elucidated via FTIR analyses (Figure 3a), the most abundant polymer in the environment was PE (40.34%), followed by PP (32.95%), in accordance with previous findings from marine and freshwater environments [59,60]. Other common polymers found in quantities <10% were PS (6.25%), PU (4.55%), and PET (4.26%). A small portion of biodegradable polymer, PLA (0.6%), was also detected: this issue was in line with the increasing use of biodegradable plastic materials, but it also raised some concerns regarding the possible persistence of this degradable polymer in natural environments [61]. Moreover, part of the analyzed fragments were identified as being "vegetal matter": this is possibly linked to superficial colonization by biological matter (see Section 3.2).

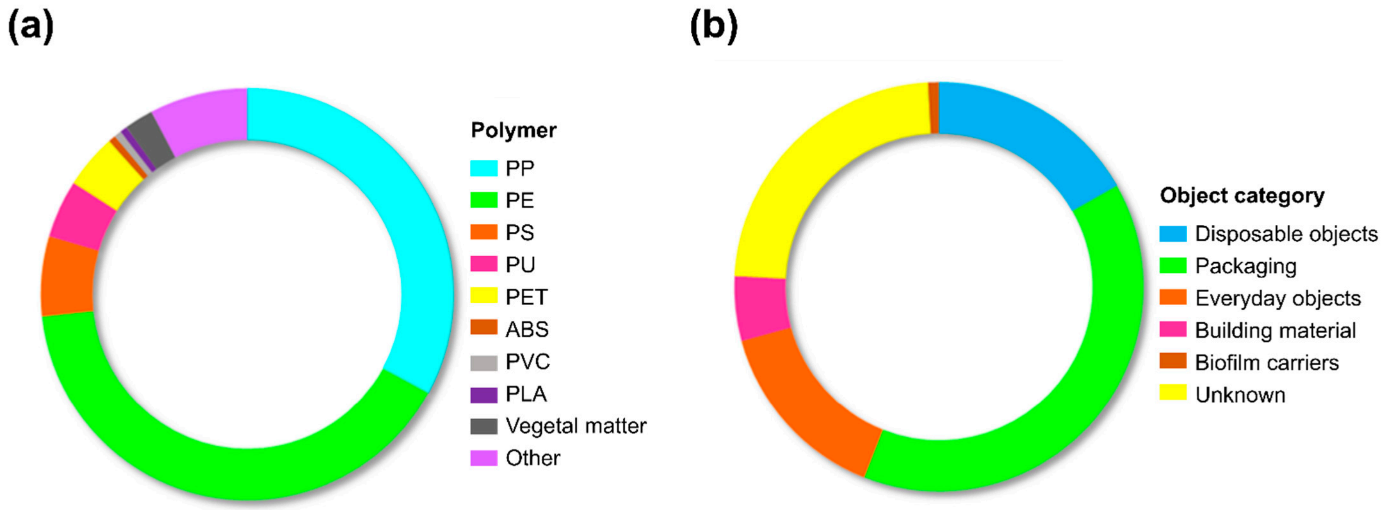

Figure 3. Pie chart showing polymeric composition (**a**) and category (**b**) of the collected objects.

Considering instead the categories of the objects collected, a similar composition was observed in all the sampling sites. In accordance with previous findings by Rizzo et al. [40], most of the plastic litter was related to activities conducted in the proximity of beaches. As a matter of fact, the main categories of beach litter were packaging (39%), everyday objects (15%), and disposable objects (17%). With regard to the latter, 3% of the objects consisted of disposable face masks, highlighting a worryingly increasing trend of plastic dispersion derived from the recent and ongoing global pandemic [5]. These three categories together represented approximately 71% of beach litter (Figure 3b), while unknown objects and building material represented 23% and 5% of found objects, respectively. Moreover, even if this was not a great contribution to the litter, it is interesting to note that about 1% of the collected items (n = 3 detected in VG and TP) was represented by biofilm carriers from wastewater treatment plants (WWTPs).

Considering the size distribution, the majority of the collected objects were of small size (less than 5 cm on the maximum axis) in almost all the sites. Only one site (CE) showed a similar number of litter objects in the <5 cm and 5–10 cm size classes (Figure S1). This is in accordance with other studies, both in the seawater and freshwater environments, and indicates that the fragmentation of plastic objects after their dispersion leads to a higher abundance of small particles [62,63].

3.2. Polymer Degradation and Alteration of Collected Samples: Problems in Recognition and Possible Environmental Consequences

Considering the comparison with the Open Specy database [52] for polymeric recognition, the most common polymers generally presented a reliable method of identification (especially PP and PE, with average R^2 values of 0.93 for PP and of 0.94 for PE), while other abundant polymers (i.e., PS and PU) showed a lower value for the Pearson coefficient (with average R^2 values of between 0.75 and 0.8, Table S2).

In fact, since the reference databases of the plastics were largely composed of pure polymers, the chemical identification of plastic objects (often containing multi-polymer blends and a differential load of additives) might show different hindrances [64]. Moreover, different physicochemical processes are known to affect the FTIR spectra of aged plastic samples, such as UV radiation, thermo-oxidative processes, mechanical abrasion, and colonization by different microbial consortia (i.e., biofouling) [17]. All these issues negatively affect recognition efficiency.

Specifically, oxidation processes can easily alter the FTIR spectra in the regions of the carboxylic (1800–1700 cm^{-1}) and hydroxyl wavelengths (3500–3000 cm^{-1}): this process is known to especially affect carbon backbone polymers (e.g., PE and PP [65], Figure 4) and it can be induced by thermo-oxidative or UV-induced oxidation and biodegradation [66,67]. These are also the mechanisms that are known to lead to plastic embrittlement and mechanical instability, inevitably leading to the production of secondary MPs [68,69].

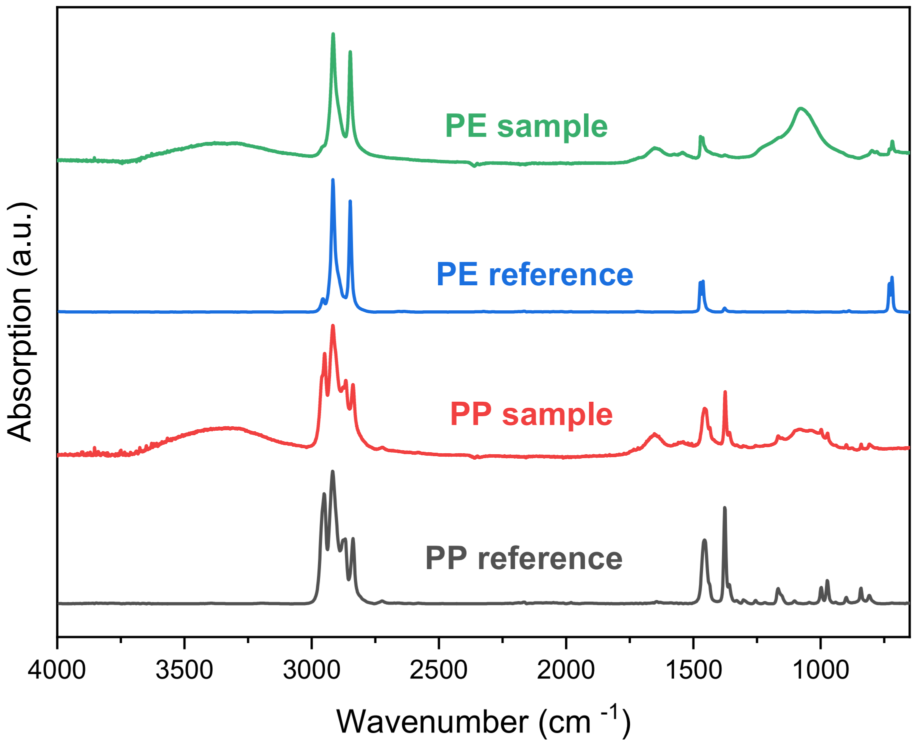

Figure 4. Example of FTIR spectra of collected samples showing altered peaks derived from polymer degradation and biofouling. PE sample (in green) shows the spectrum of a fragment collected in TP, while PP sample (in red) shows the spectrum of a film collected in VG. PE reference (in blue) and PP reference (in grey) show the reference spectra of these polymers from Openspecy database [52].

In addition to polymer matrix oxidation, other chemical alterations can occur in plastic objects, such as the leaching of additives. The FTIR spectra of PET, for example, are observed to vary in relation to environmental deterioration [70], whereas native peaks decrease, the adsorption or release of additives may originate in new ones (Figure 4). Beyond changes in the FTIR spectra, the leaching of (toxic) additives can lead to increasing ecotoxicological effects in plastic polluted environments [71].

Additionally, the mechanical degradation of plastic polymers can lead to problems in their recognition, enhancing the surface roughness of the materials, and thus negatively affecting the signal-to-noise ratio [72]. In our study, the mechanical ageing of plastic objects is also observable by the high percentage of the unknown typologies of objects (fragments and degraded objects, Figure 3b), which account for almost 25% of the sampled litter.

Finally, biofouling is another phenomenon that is known to affect plastic after its dispersion in the environment (and especially in water). Different bacteria, algae, and fungi can initiate colonization and release extracellular polymeric substances onto the plastic surfaces. This process can alter different windows of the FTIR spectra, such as in the regions of 1650 cm^{-1} and 1550 cm^{-1} (amide I and amide II stretching, respectively) due to the presence of proteins, or in the 1040 cm^{-1} band (C–O stretching), derived from polysaccharides [73]. This effect altered the recognition of a few samples in our study, which were identified as being vegetal matter by spectral matching. Despite all the samples being rinsed with deionized water, this process can still cause problems in polymer recognition, due to the high resistance of biofilm-derived organic matter. This process, moreover, has increasingly attracted the attention of researchers aiming to understand the possible cascade effects on plastic environmental behavior: biofouling is observed to affect the buoyancy rates of polymers and change the surface properties of plastics (e.g., reducing hydrophobicity), and it can selectively concentrate antibiotic-resistant bacteria and pathogens [74–76].

3.3. Litter Distribution and Possible Sources

After a global characterization of the main trends of plastic objects and polymers composing the plastic litter, the specific distribution of objects in the sampled sites was observed and compared with the local features of each sampled shore. This analysis permitted us to understand the likely mechanisms causing the accumulation of litter, including environmental (e.g., currents, winds, transport by creeks) and anthropic (e.g., ferry lines, on-site littering) processes.

An initial assessment of the factors affecting litter accumulation was performed through a correlation analysis of the litter density, and the local features of the sampling sites, such as the absence/presence of tributaries, the distance from the closest urban center for public accessibility, and the aspect of the beach (Table S3). Generally, the analyzed factors were correlated in a limited fashion with the litter density (all the Pearson's R values were below 0.75). The only variables potentially influencing the litter concentration in the sampled beaches were public accessibility ($R = 0.538$, p-value = 0.27) and the exposure of the beach ($R = 0.504$, p-value = 0.308). On the other hand, tributaries seemed to have a negligible influence on the dispersion of plastic beach litter, in accordance with previous reports [77]. The slightly negative R value ($R = -0.413$, p-value = 0.415) possibly indicates that the presence of a tributary in the beach can instead flush the plastic objects out from the beach and cause their dispersion into the lake. Indeed, in our study, one of the locations presenting a tributary (CE) was less impacted in term of litter density, while the other (TP) presented a litter density value that was below the average of all the other sampling sites, possibly due to direct littering, since it was located in the urban area of Como.

When observing the spatial trend of plastic litter abundance (in terms of density, Figure 5a), a general increase was observable on all the shores on the west side of the lake, with a N–S trend (CE < CC < VO). Moreover, the site in the inner part of Como bay (TP) and the two sites in the eastern side of Como bay (FN and VG) showed the highest impacts among the studied sites. However, the pie charts indicated only minor changes in terms of the composition of the most abundant polymers, with almost 75% of the plastic being composed of PE and PP, and a general almost equivalent portion of these polymers (only FN showed a lower abundance of PE, of less than 25%, Figure 5a).

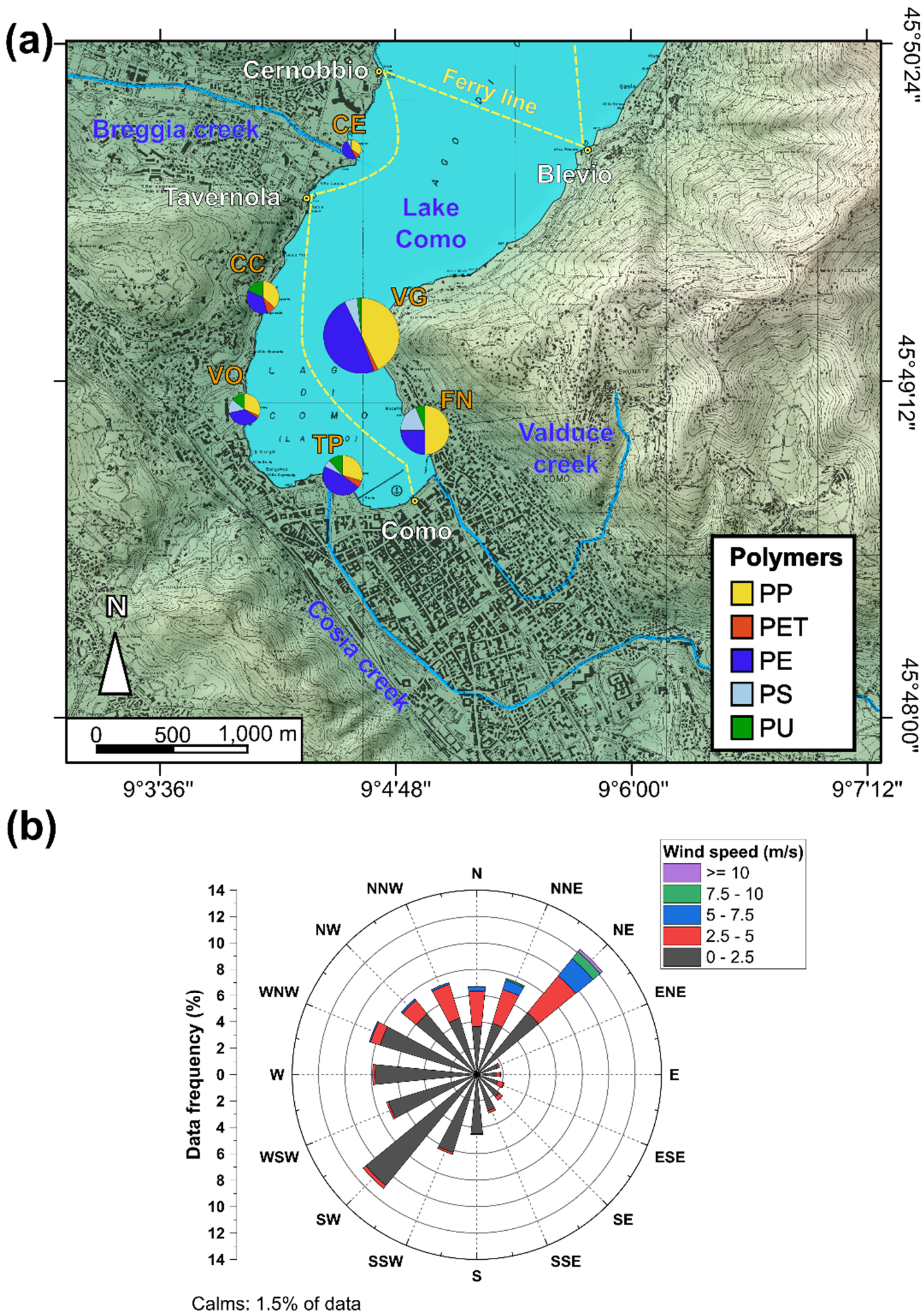

Figure 5. (**a**) Distribution of litter in the different beaches sampled in the study area. The pie chart indicates the proportions of the most frequent polymers (PP in yellow, PET in red, PE in blue, PS in light blue, and PU in green) and the dimensions of the pie are proportional to the litter density (in items/m^2). (**b**) Rose diagram of the main winds measured at Como station, with data frequency for the period January–December 2021 [51]. The different colors indicate the registered wind speeds.

Considering the low densities of all the polymers analyzed in the plastic litter, the most likely process for its spatial distribution was the surface longitudinal transport. Therefore, since most of the local winds (especially at high speeds) come from the NE in the study area (Figure 5b), a possible explanation for this observed spatial trend is derived from the transport of litter by wind. Plastic sourced from the northern part of the lake can be moved towards the south, leading to a higher concentration in the eastern part of the city (FN and VG sites), which is morphologically covered by these winds, creating a possibly stagnant zone of water. Wind current, therefore, is confirmed as being a major factor affecting the spatial distribution of plastic in lakes [78]. As a side note, the uneven distribution of the east and west sides of the lake indicates a marginal effect by the main ferry line, which presents a S–N main direction route.

Moreover, the composition of the item categories (Figure S2) showed marked changes among the different sites. While all sites showed a similar polymeric composition, the types of objects changed across the beaches. Specifically, disposable objects, packaging, and everyday objects were the prevalent categories on the most highly accessed beaches (VG and TP). However, this trend was also observable in CC, highlighting the importance of water transportation together with public accessibility and direct littering.

3.4. Implications for the Environmental Fate of Litter: Sourcing of Secondary Microplastics

As previously highlighted by different studies [22,54,79–81] the presence of MPs in water environments may pose several threats to water ecosystems. Despite the existence of different sources of MPs, this study aims to point out the sources and processes for sourcing secondary MPs that are mostly derived from the degradation of larger items [82]. This choice was made because these MPs are the most abundant in water systems [83]. Their presence, in fact, can lead to various negative effects upon water environments, such as their uptake by different organisms [50,84]. Moreover, due to the consequences of deterioration processes on the surface properties, these could act as sources and sinks for other contaminants (releasing additives and adsorbing waterborne pollutants) [85,86], potentially enhancing negative effects upon aquatic organisms. This aspect could be worsening in sediments where, due to the local chemical conditions, anthropogenic contaminants can easily concentrate [87,88]. It is therefore crucial to obtain a clearer vision regarding the sources of secondary MPs in sediments.

This study highlighted that both local (direct littering, and urbanization) and regional (main wind and currents) conditions can alter the fate of plastic litter in freshwater bodies. Moreover, further confirmation of the accumulation of mostly light polymers in beaches, derived from longitudinal transportation on the water surface, was observed. This process is widely recognized in seashores [36], but it is analyzed in a more limited fashion in freshwater bodies [78]. The FTIR analyses of these plastic polymers contributed to their recognition and also indicated an abundant degree of ageing for plastic objects, with the presence of surface oxygen containing functional groups and biofouling. These processes are known to affect all dispersed litter, and they may possibly lead to the formation of secondary microplastics, which can be directly sourced from the aged litter in the beach sediment [89,90]. Although MPs are not exclusively derived from the deterioration of larger objects, and the deterioration processes are still poorly understood, these observations provide key insights into the formation of secondary microplastics in beach sediments, suggesting that even these particles will likely be represented by a similar polymeric composition, and a greater abundance of low-density polymers (especially PE and PP). In this way, an awareness of the presence of low-density polymers can assist with the selection of an efficient extraction method for microplastics through density separation, with the selection of less hazardous and more cost-effective saline solutions (e.g., saturated NaCl), to ensure a good degree of recovery [91].

4. Conclusions

This work aimed to quantify the presence of plastic beach litter in the Como Bay (Italy) to track the main sources and processes of plastic contamination in freshwater bodies, and to assess the likely nature of secondary MPs. Moreover, since data regarding MPs in Como Lake sediments are still lacking, this could be a key factor for supporting further research on the distribution of MPs in littoral sediments, giving insights into the possible composition of (secondary) MPs. Indeed, according to our data, the most widespread polymers are PE (>40%) and PP (>32%), while other commonly used polymers represent approximately 17% of beach litter. Considering the types of sampled objects, in all the studied sites, packaging represents the most abundant category (>34%).

Since no previous data regarding the distribution of plastic litter are available for this study area, it is not possible to recognize a temporal trend in plastic distribution along the coast. However, our study allowed us to identify the most plausible contamination routes for plastic litter accumulation. A general accordance between the most accessible beaches and pollution loads highlights that in-site littering plays an important role. Moreover, the distribution of plastics in the study area suggests that part of the plastic waste can accumulate via longitudinal transport on the water surface, in accordance with the main regional wind currents. This information can be used to better address waste management strategies and the control of litter production in the neighboring coastal area, as well as giving important insights into the possible composition of secondary microplastics on the lake shoreline. In addition, local governments could promote dissemination campaigns through educational activities, such as local beach cleaning, which could increase awareness in citizens. Such activities could also offer opportunities for collecting data on plastic abundance, distribution, and composition, to be used for scientific purposes.

Supplementary Materials: The following supporting information can be downloaded at: https://www.mdpi.com/article/10.3390/app12115388/s1, Table S1: Detailed list of sampled objects. The type of object, size (in cm) and the polymeric composition are reported for each item; Table S2: Correlation values (R) for FTIR obtained by comparison with Openspecy database; Table S3: Correlation table of different site properties and plastic litter densities. Values > 0.5 are highlighted in bold. The obtained *p*-values are also included (in italics); Figure S1: Number of objects for different dimensional classes in all the sampled sites; Figure S2: Objects and polymers recovered from different beaches. On the left side: pie graphs reporting the polymeric composition of recovered objects; on the right side: sample composition in terms of "object categories".

Author Contributions: Conceptualization: A.B.; methodology: A.B. and G.B. (Ginevra Boldrocchi); data curation: G.B. (Gilberto Binda) and A.B.; writing—original draft preparation: A.B.; writing—review and editing: G.B. (Ginevra Boldrocchi), G.B. (Gilberto Binda), and R.B.; supervision: R.B. and A.P. All authors have read and agreed to the published version of the manuscript.

Funding: This research received no external funding.

Institutional Review Board Statement: Not applicable.

Informed Consent Statement: Not applicable.

Data Availability Statement: Not applicable.

Conflicts of Interest: The authors declare no conflict of interest.

References

1. Thompson, R.C.; Swan, S.H.; Moore, C.J.; Vom Saal, F.S. Our plastic age. *Philos. Trans. R. Soc. B Biol. Sci.* **2009**, *364*, 1973–1976. [CrossRef] [PubMed]
2. Martino, A.; Sany, E.; Sala, S. Resources, Conservation & Recycling Modeling the EU plastic footprint: Exploring data sources and littering potential. *Resour. Conserv. Recycl.* **2022**, *178*, 106086.
3. Association of Plastic Manufacturers (Organization). Plastics—The Facts 2020. *PlasticEurope* **2020**, *1*, 1–64.
4. Horton, A.A.; Walton, A.; Spurgeon, D.J.; Lahive, E.; Svendsen, C. Science of the Total Environment Microplastics in freshwater and terrestrial environments: Evaluating the current understanding to identify the knowledge gaps and future research priorities. *Sci. Total Environ.* **2017**, *586*, 127–141. [CrossRef] [PubMed]
5. Binda, G.; Bellasi, A.; Spanu, D.; Pozzi, A.; Cavallo, D.; Bettinetti, R. Evaluating the Environmental Impacts of Personal Protective Equipment Use by the General Population during the COVID-19 Pandemic: A Case Study of Lombardy (Northern Italy). *Environments* **2021**, *8*, 33. [CrossRef]
6. Carson, H.S.; Colbert, S.; Kaylor, M.J.; McDermid, K.J. Small plastic debris changes water movement and heat transfer through beach sediments. *Mar. Pollut. Bull.* **2011**, *62*, 1708–1713. [CrossRef]
7. Efimova, I.; Bagaeva, M.; Bagaev, A.; Kileso, A.; Chubarenko, I. Secondary Microplastics Generation in the Sea Swash Zone With Coarse Bottom Sediments: Laboratory Experiments. *Front. Mar. Sci.* **2018**, *5*, 313. [CrossRef]
8. Fortibuoni, T.; Amadesi, B.; Vlachogianni, T. Composition and abundance of macrolitter along the Italian coastline: The first baseline assessment within the european Marine Strategy Framework Directive. *Environ. Pollut.* **2021**, *268*, 115886. [CrossRef]
9. Waldschläger, K.; Born, M.; Cowger, W.; Gray, A.; Schüttrumpf, H. Settling and rising velocities of environmentally weathered micro- and macroplastic particles. *Environ. Res.* **2020**, *191*, 110192. [CrossRef]

10. Bornman, J.F.; Paul, N.D.; Shao, M. Environmental effects of ozone depletion and its interactions with climate change: 2014 assessment. Introduction. *Photochem. Photobiol. Sci. Off. J. Eur. Photochem. Assoc. Eur. Soc. Photobiol.* **2015**, *14*, 9. [CrossRef]
11. Ren, S.-Y.; Sun, Q.; Ni, H.-G.; Wang, J. A minimalist approach to quantify emission factor of microplastic by mechanical abrasion. *Chemosphere* **2020**, *245*, 125630. [CrossRef]
12. Chubarenko, I.; Esiukova, E.; Bagaev, A.; Isachenko, I.; Demchenko, N.; Zobkov, M.; Efimova, I.; Bagaeva, M.; Khatmullina, L. Chapter 6—Behavior of Microplastics in Coastal Zones. In *Microplastic Contamination in Aquatic Environments*; Zeng, E.Y., Ed.; Elsevier: Amsterdam, The Netherlands, 2018; ISBN 978-0-12-813747-5.
13. Newman, S.; Watkins, E.; Farmer, A.; ten Brink, P.; Schweitzer, J.-P. The Economics of Marine Litter. In *Marine Anthropogenic Litter*; Bergmann, M., Gutow, L., Klages, M., Eds.; Springer International Publishing: Cham, Switzerland, 2015; pp. 367–394. ISBN 978-3-319-16509-7.
14. Duis, K.; Coors, A. Microplastics in the aquatic and terrestrial environment: Sources (with a specific focus on personal care products), fate and effects. *Environ. Sci. Eur.* **2016**, *28*, 2. [CrossRef] [PubMed]
15. Kwon, J.-H.; Chang, S.; Hong, S.H.; Shim, W.J. Microplastics as a vector of hydrophobic contaminants: Importance of hydrophobic additives. *Integr. Environ. Assess. Manag.* **2017**, *13*, 494–499. [CrossRef] [PubMed]
16. Fu, L.; Li, J.; Wang, G.; Luan, Y.; Dai, W. Adsorption behavior of organic pollutants on microplastics. *Ecotoxicol. Environ. Saf.* **2021**, *217*, 112207. [CrossRef] [PubMed]
17. Binda, G.; Spanu, D.; Monticelli, D.; Pozzi, A.; Bellasi, A.; Bettinetti, R.; Carnati, S.; Nizzetto, L. Unfolding the interaction between microplastics and (trace) elements in water: A critical review. *Water Res.* **2021**, *204*, 117637. [CrossRef] [PubMed]
18. Winkler, A.; Antonioli, D.; Masseroni, A.; Chiarcos, R.; Laus, M.; Tremolada, P. Following the fate of microplastic in four abiotic and biotic matrices along the Ticino River (North Italy). *Sci. Total Environ.* **2022**, *823*, 153638. [CrossRef]
19. Galafassi, S.; Sighicelli, M.; Pusceddu, A.; Bettinetti, R.; Cau, A.; Temperini, M.E.; Gillibert, R.; Ortolani, M.; Pietrelli, L.; Zaupa, S.; et al. Microplastic pollution in perch (Perca fluviatilis, Linnaeus 1758) from Italian south-alpine lakes. *Environ. Pollut.* **2021**, *288*, 117782. [CrossRef]
20. Setälä, O.; Fleming-Lehtinen, V.; Lehtiniemi, M. Ingestion and transfer of microplastics in the planktonic food web. *Environ. Pollut.* **2014**, *185*, 77–83. [CrossRef]
21. Watts, A.J.R.; Lewis, C.; Goodhead, R.M.; Beckett, S.J.; Moger, J.; Tyler, C.R.; Galloway, T.S. Uptake and Retention of Microplastics by the Shore Crab *Carcinus maenas*. *Environ. Sci. Technol.* **2014**, *48*, 8823–8830. [CrossRef]
22. Ory, N.C.; Gallardo, C.; Lenz, M.; Thiel, M. Capture, swallowing, and egestion of microplastics by a planktivorous juvenile fish. *Environ. Pollut.* **2018**, *240*, 566–573. [CrossRef]
23. Eerkes-Medrano, D.; Thompson, R.C.; Aldridge, D.C. Microplastics in freshwater systems: A review of the emerging threats, identification of knowledge gaps and prioritisation of research needs. *Water Res.* **2015**, *75*, 63–82. [CrossRef] [PubMed]
24. Piyawardhana, N.; Weerathunga, V.; Chen, H.-S.; Guo, L.; Huang, P.-J.; Ranatunga, R.; Hung, C.-C. Occurrence of microplastics in commercial marine dried fish in Asian countries. *J. Hazard. Mater.* **2022**, *423*, 127093. [CrossRef] [PubMed]
25. Weis, J.S. Improving microplastic research. *AIMS Environ. Sci.* **2019**, *6*, 326–340. [CrossRef]
26. Lei, L.; Wu, S.; Lu, S.; Liu, M.; Song, Y.; Fu, Z.; Shi, H.; Raley-Susman, K.M.; He, D. Microplastic particles cause intestinal damage and other adverse effects in zebrafish Danio rerio and nematode *Caenorhabditis elegans*. *Sci. Total Environ.* **2018**, *619–620*, 1–8. [CrossRef]
27. Boldrocchi, G.; Monticelli, D.; Omar, Y.M.; Bettinetti, R. Trace elements and POPs in two commercial shark species from Djibouti: Implications for human exposure. *Sci. Total Environ.* **2019**, *669*, 637–648. [CrossRef] [PubMed]
28. Mazzoni, M.; Buffo, A.; Cappelli, F.; Pascariello, S.; Polesello, S.; Valsecchi, S.; Volta, P.; Bettinetti, R. Perfluoroalkyl acids in fish of Italian deep lakes: Environmental and human risk assessment. *Sci. Total Environ.* **2019**, *653*, 351–358. [CrossRef]
29. Rios, L.M.; Moore, C.; Jones, P.R. Persistent organic pollutants carried by synthetic polymers in the ocean environment. *Mar. Pollut. Bull.* **2007**, *54*, 1230–1237. [CrossRef] [PubMed]
30. Velzeboer, I.; Kwadijk, C.J.A.F.; Koelmans, A.A. Strong Sorption of PCBs to Nanoplastics, Microplastics, Carbon Nanotubes, and Fullerenes. *Environ. Sci. Technol.* **2014**, *48*, 4869–4876. [CrossRef]
31. Amelia, T.S.M.; Khalik, W.M.A.W.M.; Ong, M.C.; Shao, Y.T.; Pan, H.-J.; Bhubalan, K. Marine microplastics as vectors of major ocean pollutants and its hazards to the marine ecosystem and humans. *Prog. Earth Planet. Sci.* **2021**, *8*, 12. [CrossRef]
32. Hartmann, N.B.; Rist, S.; Bodin, J.; Jensen, L.H.S.; Schmidt, S.N.; Mayer, P.; Meibom, A.; Baun, A. Microplastics as vectors for environmental contaminants: Exploring sorption, desorption, and transfer to biota. *Integr. Environ. Assess. Manag.* **2017**, *13*, 488–493. [CrossRef]
33. Wang, J.; Wang, M.; Ru, S.; Liu, X. Science of the Total Environment High levels of microplastic pollution in the sediments and benthic organisms of the South Yellow Sea, China. *Sci. Total Environ.* **2019**, *651*, 1661–1669. [CrossRef]
34. Joint Research Centre, Institute for Environment and Sustainability. *Guidance on Monitoring of Marine Litter in European Seas*; Publications Office of the European Union: Luxemburg, 2014; Available online: https://data.europa.eu/doi/10.2788/99816 (accessed on 28 April 2022).
35. Bellasi, A.; Binda, G.; Pozzi, A.; Galafassi, S.; Volta, P.; Bettinetti, R. Microplastic Contamination in Freshwater Environments: A Review, Focusing on Interactions with Sediments and Benthic Organisms. *Environments* **2020**, *7*, 30. [CrossRef]
36. Li, Y.; Wolanski, E.; Dai, Z.; Lambrechts, J.; Tang, C.; Zhang, H. Trapping of plastics in semi-enclosed seas: Insights from the Bohai Sea, China. *Mar. Pollut. Bull.* **2018**, *137*, 509–517. [CrossRef]

37. Cai, Y.; Li, C.; Zhao, Y. A Review of the Migration and Transformation of Microplastics in Inland Water Systems. *Int. J. Environ. Res. Public Health* **2021**, *19*, 148. [CrossRef] [PubMed]
38. Mecho, A.; Sellanes, J.; Aguzzi, J. Seafloor litter at oceanic islands and seamounts of the southeastern Pacific. *Mar. Pollut. Bull.* **2021**, *170*, 112641. [CrossRef] [PubMed]
39. Andriolo, U.; Gonçalves, G.; Sobral, P.; Bessa, F. Spatial and size distribution of macro-litter on coastal dunes from drone images: A case study on the Atlantic coast. *Mar. Pollut. Bull.* **2021**, *169*, 112490. [CrossRef] [PubMed]
40. Rizzo, A.; Rangel-Buitrago, N.; Impedovo, A.; Mastronuzzi, G.; Scardino, G.; Scicchitano, G. A rapid assessment of litter magnitudes and impacts along the Torre Guaceto marine protected area (Brindisi, Italy). *Mar. Pollut. Bull.* **2021**, *173*, 112987. [CrossRef]
41. Asensio-Montesinos, F.; Anfuso, G.; Aguilar-Torrelo, M.T.; Ramírez, M.O. Abundance and Temporal Distribution of Beach Litter on the Coast of Ceuta (North Africa, Gibraltar Strait). *Water* **2021**, *13*, 2739. [CrossRef]
42. Rangel-Buitrago, N.; Velez-Mendoza, A.; Gracia, C.A.; Neal, W.J. The impact of anthropogenic litter on Colombia's central Caribbean beaches. *Mar. Pollut. Bull.* **2020**, *152*, 110909. [CrossRef]
43. Topçu, E.N.; Tonay, A.M.; Dede, A.; Öztürk, A.A.; Öztürk, B. Origin and abundance of marine litter along sandy beaches of the Turkish Western Black Sea Coast. *Mar. Environ. Res.* **2013**, *85*, 21–28. [CrossRef]
44. ISPRA. *Municipal Waste Report—Edition 2019*; Istituto Superiore per la Protezione e la Ricerca Ambientale: Rome, Italy, 2018; Volume 313, ISBN 978-88-448-0971-3.
45. Binda, G.; Frascoli, F.; Spanu, D.; Ferrario, M.F.; Terrana, S.; Gambillara, R.; Trotta, S.; Noble, P.J.; Livio, F.A.; Pozzi, A.; et al. Geochemical Markers as a Tool for the Characterization of a Multi-Layer Urban Aquifer: The Case Study of Como (Northern Italy). *Water* **2022**, *14*, 124. [CrossRef]
46. Lebreton, L.C.M.; van der Zwet, J.; Damsteeg, J.-W.; Slat, B.; Andrady, A.; Reisser, J. River plastic emissions to the world's oceans. *Nat. Commun.* **2017**, *8*, 15611. [CrossRef]
47. Vavassori, A.; Oxoli, D.; Brovelli, M.A. Population Space–Time Patterns Analysis and Anthropic Pressure Assessment of the Insubric Lakes Using User-Generated Geodata. *ISPRS Int. J. Geo-Inform.* **2022**, *11*, 206. [CrossRef]
48. ISTAT. Available online: http://dati.istat.it/Index.aspx?DataSetCode=DCIS_POPRES1# (accessed on 28 April 2022).
49. Barnes, D.K.A.; Galgani, F.; Thompson, R.C.; Barlaz, M. Accumulation and fragmentation of plastic debris in global environments. *Philos. Trans. R. Soc. Lond. B Biol. Sci.* **2009**, *364*, 1985–1998. [CrossRef] [PubMed]
50. GESAMP Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection. *Sources, Fate and Effects of Microplastics in the Marine Environment: A Global Assessment*; Kershaw, P.J., Ed.; IMO/FAO/UNESCO-IOC/UNIDO/WMO/IAEA/UN/UNEP/UNDP International Maritime Organization (IMO): London, UK, 2015; 96p.
51. ARPA. Available online: https://www.arpalombardia.it/Pages/Meteorologia/Richiesta-dati-misurati.aspx (accessed on 2 May 2022).
52. Cowger, W.; Steinmetz, Z.; Gray, A.; Munno, K.; Lynch, J.; Hapich, H.; Primpke, S.; De Frond, H.; Rochman, C.; Herodotou, O. Microplastic Spectral Classification Needs an Open Source Community: Open Specy to the Rescue! *Anal. Chem.* **2021**, *93*, 7543–7548. [CrossRef] [PubMed]
53. Aslam, S.; Tzoraki, O.; Krasakopoulou, E. Anthropogenic litter in freshwater bodies and their estuaries: An empirical analysis in Lesvos, Greece. *Environ. Sci. Pollut. Res.* **2022**, *29*, 16563–16575. [CrossRef] [PubMed]
54. Hengstmann, E.; Fischer, E.K. Anthropogenic litter in freshwater environments—Study on lake beaches evaluating marine guidelines and aerial imaging. *Environ. Res.* **2020**, *189*, 109945. [CrossRef] [PubMed]
55. Chapman, A. The Sources of Macroplastic Waste in Freshwater Ecosystems. *Environ. Res.* **2019**, *10*, 860–866.
56. Hoellein, T.J.; Westhoven, M.; Lyandres, O.; Cross, J. Abundance and environmental drivers of anthropogenic litter on 5 Lake Michigan beaches: A study facilitated by citizen science data collection. *J. Great Lakes Res.* **2015**, *41*, 78–86. [CrossRef]
57. Lucas-Solis, O.; Moulatlet, G.M.; Guamangallo, J.; Yacelga, N.; Villegas, L.; Galarza, E.; Rosero, B.; Zurita, B.; Sabando, L.; Cabrera, M.; et al. Preliminary Assessment of Plastic Litter and Microplastic Contamination in Freshwater Depositional Areas: The Case Study of Puerto Misahualli, Ecuadorian Amazonia. *Bull. Environ. Contam. Toxicol.* **2021**, *107*, 45–51. [CrossRef]
58. Blettler, M.C.M.; Ulla, M.A.; Rabuffetti, A.P.; Garello, N. Plastic pollution in freshwater ecosystems: Macro-, meso-, and microplastic debris in a floodplain lake. *Environ. Monit. Assess.* **2017**, *189*, 581. [CrossRef] [PubMed]
59. Issac, M.N.; Kandasubramanian, B. Effect of microplastics in water and aquatic systems. *Environ. Sci. Pollut. Res.* **2021**, *28*, 19544–19562. [CrossRef] [PubMed]
60. Schwarz, A.; Ligthart, T.; Boukris, E.; van Harmelen, T. Sources, transport, and accumulation of different types of plastic litter in aquatic environments: A review study. *Mar. Pollut. Bull.* **2019**, *143*, 92–100. [CrossRef]
61. Weinstein, J.E.; Dekle, J.L.; Leads, R.R.; Hunter, R.A. Degradation of bio-based and biodegradable plastics in a salt marsh habitat: Another potential source of microplastics in coastal waters. *Mar. Pollut. Bull.* **2020**, *160*, 111518. [CrossRef]
62. Schirinzi, G.F.; Köck-Schulmeyer, M.; Cabrera, M.; González-Fernández, D.; Hanke, G.; Farré, M.; Barceló, D. Riverine anthropogenic litter load to the Mediterranean Sea near the metropolitan area of Barcelona, Spain. *Sci. Total Environ.* **2020**, *714*, 136807. [CrossRef] [PubMed]
63. Fok, L.; Cheung, P.K.; Tang, G.; Li, W.C. Size distribution of stranded small plastic debris on the coast of Guangdong, South China. *Environ. Pollut.* **2017**, *220*, 407–412. [CrossRef]

64. ter Halle, A.; Ladirat, L.; Martignac, M.; Mingotaud, A.F.; Boyron, O.; Perez, E. To what extent are microplastics from the open ocean weathered? *Environ. Pollut.* **2017**, *227*, 167–174. [CrossRef]
65. Zbyszewski, M.; Corcoran, P.L. Distribution and Degradation of Fresh Water Plastic Particles Along the Beaches of Lake Huron, Canada. *Water Air Soil Pollut.* **2011**, *220*, 365–372. [CrossRef]
66. Fairbrother, A.; Hsueh, H.-C.; Kim, J.H.; Jacobs, D.; Perry, L.; Goodwin, D.; White, C.; Watson, S.; Sung, L.-P. Temperature and light intensity effects on photodegradation of high-density polyethylene. *Polym. Degrad. Stab.* **2019**, *165*, 153–160. [CrossRef]
67. Wei, R.; Zimmermann, W. Microbial enzymes for the recycling of recalcitrant petroleum-based plastics: How far are we? *Microb. Biotechnol.* **2017**, *10*, 1308–1322. [CrossRef]
68. Ainali, N.M.; Bikiaris, D.N.; Lambropoulou, D.A. Aging effects on low- and high-density polyethylene, polypropylene and polystyrene under UV irradiation: An insight into decomposition mechanism by Py-GC/MS for microplastic analysis. *J. Anal. Appl. Pyrolysis* **2021**, *158*, 105207. [CrossRef]
69. Fayolle, B.; Audouin, L.; Verdu, J. Initial steps and embrittlement in the thermal oxidation of stabilised polypropylene films. *Polym. Degrad. Stab.* **2002**, *75*, 123–129. [CrossRef]
70. Ioakeimidis, C.; Fotopoulou, K.N.; Karapanagioti, H.K.; Geraga, M.; Zeri, C.; Papathanassiou, E.; Galgani, F.; Papatheodorou, G. The degradation potential of PET bottles in the marine environment: An ATR-FTIR based approach. *Sci. Rep.* **2016**, *6*, 23501. [CrossRef]
71. Turner, A.; Filella, M. Hazardous metal additives in plastics and their environmental impacts. *Environ. Int.* **2021**, *156*, 106622. [CrossRef] [PubMed]
72. Henriksen, M.L.; Karlsen, C.B.; Klarskov, P.; Hinge, M. Plastic classification via in-line hyperspectral camera analysis and unsupervised machine learning. *Vib. Spectrosc.* **2022**, *118*, 103329. [CrossRef]
73. Rahman, M.M.; Al-Sulaimi, S.; Farooque, A.M. Characterization of new and fouled SWRO membranes by ATR/FTIR spectroscopy. *Appl. Water Sci.* **2018**, *8*, 183. [CrossRef]
74. Kalčíková, G.; Bundschuh, M. Aquatic Biofilms—Sink or Source of Microplastics? A Critical Reflection on Current Knowledge. *Environ. Toxicol. Chem.* **2022**, *41*, 838–843. [CrossRef]
75. He, S.; Jia, M.; Xiang, Y.; Song, B.; Xiong, W.; Cao, J.; Peng, H.; Yang, Y.; Wang, W.; Yang, Z.; et al. Biofilm on microplastics in aqueous environment: Physicochemical properties and environmental implications. *J. Hazard. Mater.* **2021**, *424*, 127286. [CrossRef]
76. Pham, D.N.; Clark, L.; Li, M. Microplastics as hubs enriching antibiotic-resistant bacteria and pathogens in municipal activated sludge. *J. Hazard. Mater. Lett.* **2021**, *2*, 100014. [CrossRef]
77. Schöneich-Argent, R.I.; Dau, K.; Freund, H. Wasting the North Sea?—A field-based assessment of anthropogenic macrolitter loads and emission rates of three German tributaries. *Environ. Pollut.* **2020**, *263*, 114367. [CrossRef]
78. Fischer, E.K.; Paglialonga, L.; Czech, E.; Tamminga, M. Microplastic pollution in lakes and lake shoreline sediments—A case study on Lake Bolsena and Lake Chiusi (central Italy). *Environ. Pollut.* **2016**, *213*, 648–657. [CrossRef] [PubMed]
79. Thompson, R.C.; Olsen, Y.; Mitchell, R.P.; Davis, A.; Rowland, S.J.; John, A.W.G.; McGonigle, D.; Russell, A.E. Lost at Sea: Where Is All the Plastic? *Science* **2004**, *304*, 838. [CrossRef]
80. Collard, F.; Gilbert, B.; Compère, P.; Eppe, G.; Das, K.; Jauniaux, T.; Parmentier, E. Microplastics in livers of European anchovies (*Engraulis encrasicolus*, L.). *Environ. Pollut.* **2017**, *229*, 1000–1005. [CrossRef]
81. Slootmaekers, B.; Carteny, C.C.; Belpaire, C.; Saverwyns, S.; Fremout, W.; Blust, R.; Bervoets, L. Microplastic contamination in gudgeons (Gobio gobio) from Flemish rivers (Belgium). *Environ. Pollut.* **2018**, *244*, 675–684. [CrossRef] [PubMed]
82. Wagner, M.; Scherer, C.; Alvarez-Muñoz, D.; Brennholt, N.; Bourrain, X.; Buchinger, S.; Fries, E.; Grosbois, C.; Klasmeier, J.; Marti, T.; et al. Microplastics in freshwater ecosystems: What we know and what we need to know. *Environ. Sci. Eur.* **2014**, *26*, 12. [CrossRef] [PubMed]
83. Wong, J.K.H.; Lee, K.K.; Tang, K.H.D.; Yap, P.-S. Microplastics in the freshwater and terrestrial environments: Prevalence, fates, impacts and sustainable solutions. *Sci. Total Environ.* **2020**, *719*, 137512. [CrossRef] [PubMed]
84. Wright, S.L.; Rowe, D.; Thompson, R.C. Correspondences Microplastic ingestion decreases energy reserves in marine worms. *Curr. Biol.* **2013**, *23*, R1031–R1033. [CrossRef] [PubMed]
85. Endo, S.; Yuyama, M.; Takada, H. Desorption kinetics of hydrophobic organic contaminants from marine plastic pellets. *Mar. Pollut. Bull.* **2013**, *74*, 125–131. [CrossRef]
86. Sun, R.; Luo, X.; Li, Q.X.; Wang, T.; Zheng, X.; Peng, P.; Mai, B. Legacy and emerging organohalogenated contaminants in wild edible aquatic organisms: Implications for bioaccumulation and human exposure. *Sci. Total Environ.* **2018**, *616–617*, 38–45. [CrossRef]
87. Zalasiewicz, J.; Waters, C.N.; Sul, J.I.D.; Corcoran, P.L.; Barnosky, A.D.; Cearreta, A.; Edgeworth, M.; Gałuszka, A.; Jeandel, C.; Leinfelder, R.; et al. The geological cycle of plastics and their use as a stratigraphic indicator of the Anthropocene. *Anthropocene* **2016**, *13*, 4–17. [CrossRef]
88. Kowalski, N.; Reichardt, A.M.; Waniek, J. Sinking rates of microplastics and potential implications of their alteration by physical, biological, and chemical factors. *Mar. Pollut. Bull.* **2016**, *109*, 310–319. [CrossRef] [PubMed]
89. Qin, J.; Liang, B.; Peng, Z.; Lin, C. Generation of microplastic particles during degradation of polycarbonate films in various aqueous media and their characterization. *J. Hazard. Mater.* **2021**, *415*, 125640. [CrossRef] [PubMed]

90. Barlow, D.E.; Biffinger, J.C.; Estrella, L.; Lu, Q.; Hung, C.-S.; Nadeau, L.J.; Crouch, A.L.; Russell, J.N., Jr.; Crookes-Goodson, W.J. Edge-Localized Biodeterioration and Secondary Microplastic Formation by Papiliotrema laurentii Unsaturated Biofilm Cells on Polyurethane Films. *Langmuir* **2020**, *36*, 1596–1607. [CrossRef] [PubMed]
91. Bellasi, A.; Binda, G.; Pozzi, A.; Boldrocchi, G.; Bettinetti, R. The extraction of microplastics from sediments: An overview of existing methods and the proposal of a new and green alternative. *Chemosphere* **2021**, *278*, 130357. [CrossRef] [PubMed]