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Review

Evolving Global Microplastic Research: History and Perspective

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Abstract: Although plastic was invented more than a century ago, massive production did not start until the early 1950s. The widespread use of plastic products since then has substantially improved the quality of life in modern society, but also created increasing environmental concerns. The earliest documented research was conducted in the early 1970s and was popularized following the discovery of floating plastic debris in the Pacific Ocean in the late 1990s. Tiny plastic particles were further defined as "microplastics" (MPs) in 2004, which greatly spurred research activities globally. Sampling methods for MPs mainly vary among environmental matrices, while FTIR and Raman spectroscopies, as well as pyrolysis-GC-MS remain the prevailing tools for MP analysis. Cross-compartment transfer of MPs is well documented, but uncertainties remain regarding their bioaccumulation and biomagnification. A more controversial issue is the occurrence of MPs in human organs, which has provoked intensive debates and concerns. Improved analytical protocols and quality-control procedures are essential to generate robust data and reduce uncertainties. On the other hand, discharge of terrestrial MPs to the global ocean has been estimated, but available estimates vary in several orders of magnitude, which poses difficulty in constraining the combined impact of anthropogenic activities on marine plastic pollution. Substantial uncertainties remain regarding human exposure, bioaccumulation and biomagnification processes, as well as the magnitude of land-to-sea MP fluxes, posing major challenges for future research and risk assessment.

Keywords: microplastic; analytical technology; cross-compartment transfer; bioaccumulation; human exposure; land-to-sea transport

1. Introduction

Over the last two decades since "microplastics" (MPs) was first used to describe "tiny plastic particles" [1], global MP research activity and publication output have been growing nearly exponentially, particularly in the last 10 years (Figure 1). This spectacular phenomenon has been driven by a multitude of factors such as the gradual recognition of the potential adverse impact of MP pollution by global governments and international organizations/agencies, as well as the public's increasing awareness of the importance to protect the environment. The drastic increase in research productivity by China's researchers has also greatly contributed to the upward trajectory. China was responsible for the largest share of peer-reviewed papers published in the period of 2020–2024, with five individuals from mainland China on the global top 10 list [2].

Despite the tremendous efforts in characterizing the fate, transport, toxicity, and human health risk of MPs, large knowledge gaps have remained [3]. Available data are frequently inconsistent or contradictory with one another, and even beyond common sense. Conflicting findings are often associated with technological limitations, procedural deficits, and inappropriate approaches in data analysis and interpretation. Methodological innovation and proficient execution are therefore important areas of improvement to reduce uncertainties in results from MP research. Caution

needs to be exercised in selecting data processing tools and how data are interpreted. Extrapolation of data from small to large scales and from short- to long-term exposure always requires rigorous checks and validations.

Given the rapid accumulation and complex nature of MP research data, we believe it is the appropriate time to reflect how MP (and plastic) research has evolved and where it is heading. We aim to (1) outline the timelines of when marine plastic debris were discovered and MPs were defined, which has served as the catalyst for subsequent growth of MP researches worldwide; (2) present the prevailing sampling and analytical technologies and their applicability in MP researches; (3) examine the environmental fate and ecotoxicity of MPs, particularly the suitability of the concepts for bioaccumulation and biomagnification in describing the behavior of MPs; and (4) assess current estimation of land-to-sea discharge of (micro)plastics and the relative importance of riverine and atmospheric transport pathways. It is our hope that this review will assist in understanding both the significance and uncertainties of MP research and aid in selecting future research areas for younger researchers.

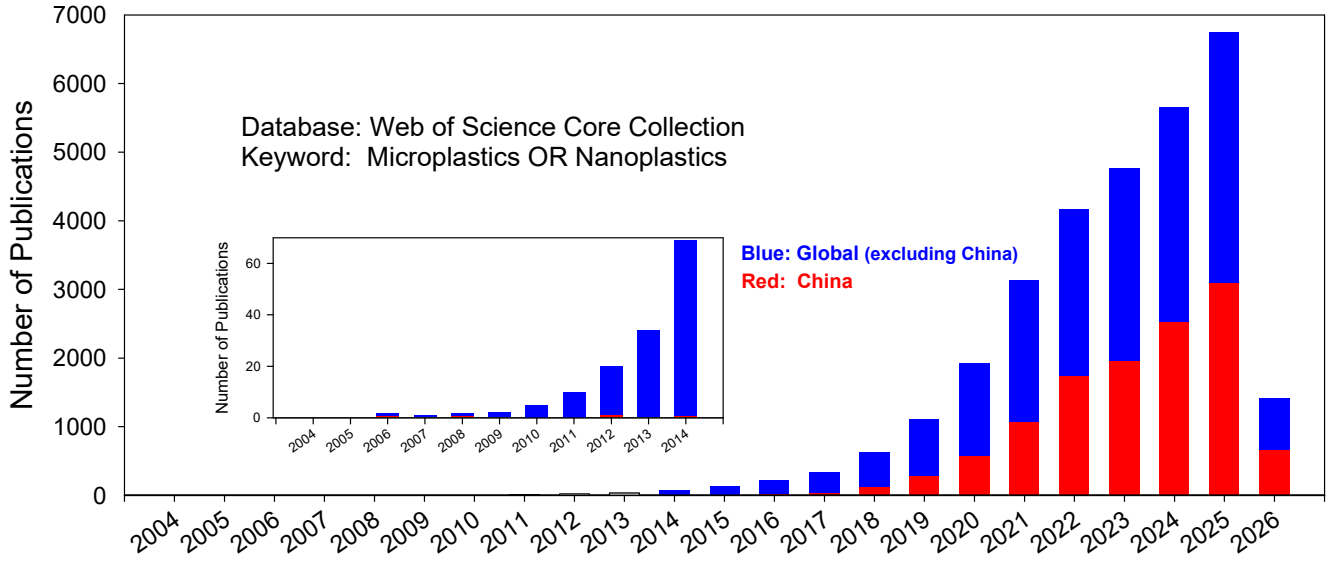

Figure 1. Number of published papers on microplastics from 2004 to March 2026. The data for China may be overestimated as the citations of any paper with at least one mainland China's affiliation are attributed to those for China. Data were updated on 22 March 2026.

2. Historical Evolution of Plastic and Microplastic Research: From Discovery to Definition

Plastic was invented more than 100 years ago, with various polymer forms synthesized during different time periods [4]. However, large-scale industrial production did not begin until 1950, when annual production reached approximately 2 million metric tons (mMT) [5]. Since then, plastic production has increased dramatically, reaching 430 mMT in 2024, approximately one-third of which was produced in China [6]. Polyethylene (PE) remains the dominant polymer type, accounting for 30–36% of the global production, followed by polypropylene (PP; 16–21%), polyvinyl chloride (PVC; 9–12%), polyethylene terephthalate (PET; 8–10%), and polystyrene (PS; 6–7%) [5]. When plastic products reach the end of their service lifetime, the majority of obsolete plastics is landfilled and recycled; however, certain amounts of plastic waste are inevitably released into the environment [5]. Plastic waste discharged to the environment may eventually enter aquatic systems and subsequently the global oceans. An estimate suggested that around 0.5% of plastic waste ends up in the ocean, corresponding to about 1–2 mMT annually [7].

Plastic waste in the form of fishing line was believed to be first detected in 1957 [8], while the first detection of a plastic bag was recorded in 1965 off the coast of Ireland [9]. The very first meaningful field study of marine plastics was conducted in 1971 at a small area of the western Sargasso Sea within the Atlantic Ocean [10]. This was the first time a 0.33-mm neuston net was used for sampling plastic particles, which, as shown later in this review, had profound implications for subsequent field studies. Another large-scale and comprehensive survey of marine plastics was reported in 1974, which collected plastic debris from the North Atlantic and suggested that polyethylene (PE) fragments, polystyrene (PS) spheres, and foams were the dominant constituents [11]. These pioneering studies laid the foundation for future investigations into the occurrence and distribution of marine plastics (and MPs), with influences extending to generations of subsequent research.

Surprisingly, the scientific interest in marine plastic pollution remained relatively stagnant over the following

two decades [12]. A major turning point occurred in 1997 when Charles Moore and his crew, while returning to California after the Los Angeles-to-Hawaii Transpacific sailing race, observed abundant plastic debris floating in the North Pacific Gyre, including bottles, bottle caps, wrappers, and fragments [13]. They subsequently collected samples from an area near the central pressure cell of the North Pacific subtropical high in August 1999 using a manta trawl equipped with a 0.333-mm mesh net (3.5 meters long). Thin films, polypropylene/monofilament line, and unidentified plastic accounted for 98% of the total number of plastic pieces collected. The abundance and mass of neustonic plastic were 334,271 pieces km^{-2} and 5114 g km^{-2}, respectively [14]. The endeavor by Moore and colleagues opened a new chapter in marine plastic research and greatly stimulated global interest in plastic pollution. The major milestones in the historical development of plastic and microplastic research are summarized in Figure 2.

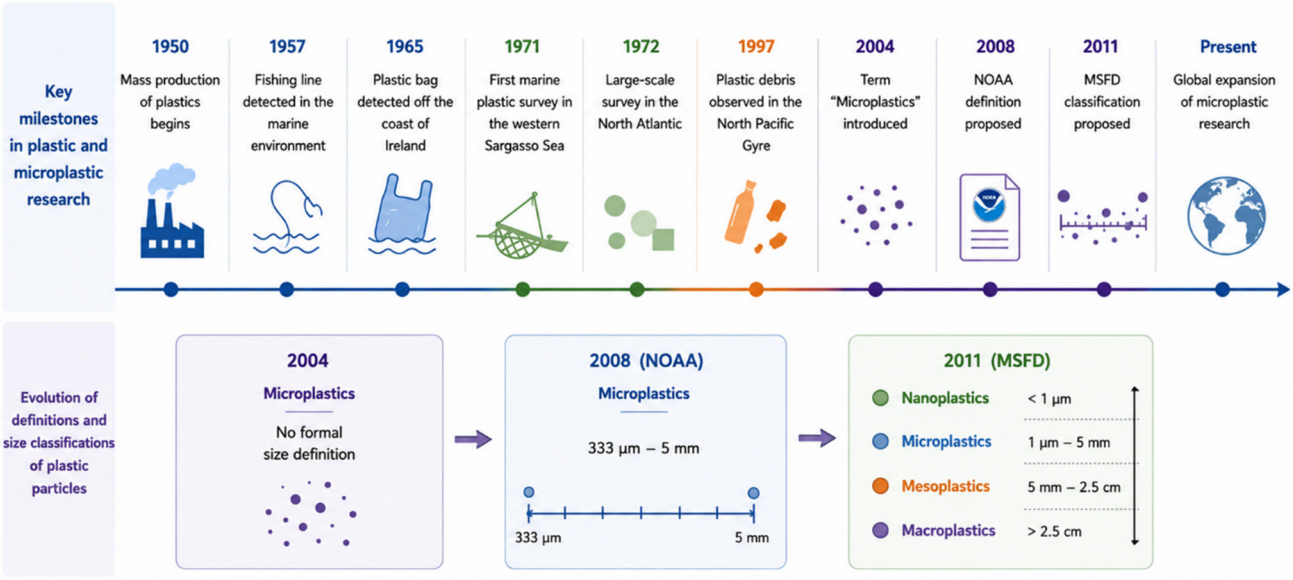

Figure 2. Evolution of microplastic definitions and size classifications.

As research on plastic pollution expanded, increasing attention was directed toward small plastic particles generated through the fragmentation and degradation of larger plastic debris. Thompson et al. [1] first introduced the term "microplastics" (MPs) to describe these tiny plastic particles. Ironically, Thompson et al. [1] did not define the size range for MPs, yet many subsequent studies incorrectly cited this paper as the first formal definition of the microplastic size. In 2008, the United States National Oceanic and Atmospheric Administration held an international symposium, and later published a technical report, which tentatively define MPs as plastic particles >333 μm and <5 mm [15]. The selection of 333 μm as the lower boundary was obviously in consistency with the use of a 0.33-mm neuston net in the pioneering field survey of Carpenter and Smith, Jr. [10]. The technical report further suggested that MPs could be subdivided into additional size classes as more monitoring data became available, while retaining the concept that MPs referred to plastic particles within the micrometers-to-millimeter range [15]. A few years later, the European Marine Strategic Framework Directive proposed a more comprehensive classification scheme for plastic particles [16], including nanoplastics (<1 μm) ; MPs (>1 μm and <5 mm), mesoplastics (>5 mm and <2.5 cm); and macroplastics (>2.5 cm). Microplastics were further divided into small MPs (>1 μm and <1 mm) and large MPs (>1 mm and <5 mm).

With fast accumulation of MPs data, it has become clear that adopting a unified size ranges for plastic particles has the utmost benefit for interested researchers, government agencies, international organizations, as well as the public. This would allow hard-to-get field data to be analyzed and interpreted under the same framework (or umbrella), and maximize their utility in advancing the science of plastic pollution On the other hand, despite the importance of classifying plastic particles into clearly defined size ranges, the term "microplastic" has been often used to broadly refer to plastic particles of all sizes. Such usage reflects the fast growing popularity of the term "microplastic", particularly among the scientific community, media, general public, etc. and provides considerable convenience in communicating research findings. Therefore, MPs are also loosely used in the present review paper to refer to plastic particles of various sizes. Nevertheless, the widespread use of different size definitions may contribute to inconsistencies in data comparison and interpretation among studies. Future progress in MP research would therefore benefit from internationally harmonized definitions and standardized reporting criteria.

3. Field sampling Approaches

Field sampling represents one of the first and most critical steps in MP research. A typical workflow consists of sample collection, pretreatment, particle identification, and quantification (Figure 3). The reliability of each downstream step depends strongly on the representativeness and quality of the collected samples. As mentioned above, the first meaningful field sampling [10] employed a 0.33-mm Neuston net, invented by Russian scientist P.V. Zaitsev in the 1960s to capture biological samples in the sea-surface layer [17]. Manta trawl, an upgraded and more stable version of Neuston net, was adapted and popularized by Moore et al. [14] who established protocols for trawling speed, distance, and 0.333-mm mesh size. It has now become one of the standard sampling tools for collecting MPs in large water bodies and in fact is beneficial for conducting data analysis and interpretation on a common ground. Other variants of Manta trawl have also been developed, such as Plankton net, Bongo net, and microplastics trap [18]. Besides dynamic sampling with Manta trawl popularized by Moore et al. [14], microplastic samples can also be collected stationarily. For example, a string bag can be deployed in the middle of a river or stream; suspended particles carrying MP debris can be intercepted by the string bag due to water flows. Stationary sampling is suited for small water bodies which do not support net trawling by boats and does not require power supply. The main drawback of stationary sampling is perhaps the relatively small sample volume that can be processed. Another viable sampling approach is a combination of filtration and sieving. Filtration and sieving can be operated on-site or in the laboratory; however, on-site operation is inefficient and laboratory processing requires transport and storage of large-volume water that may be costly and labor-intensive [19].

Aside from all the benefits and deficits of different sampling techniques, one of the major challenges facing current aquatic sampling methods is the lack of standardized sampling protocols. Differences in mesh size, sampling volume, trawling speed, sampling duration, and sampling depth may substantially influence the abundance and size distribution of MPs reported by different studies. Consequently, methodological variability may contribute to some of the discrepancies observed among datasets collected from similar environments. Setting a commonly accepted sampling protocol would further homogenize datasets acquired by different research groups and ease the assessment of patterns and trends on regional and/or global scales.

Atmospheric MPs sampling is very similar to that used for collecting aerosol particles. Passive samplers collect naturally depositing MP particles and hence are advantageous in areas without power supply [20]. However, the efficiency of passive sampling can be adversely impacted by weather conditions. Another deficit with passive sampling is that low-density MPs are difficult to deposit and therefore not easily collected. Active sampling strategies typically employ an air pump to draw air through a holder housing one or more filters. Active samplers can process large volumes of air in a short period of time, but are not feasible at sites with no power supply [21]. Apparently, the choice of atmospheric sampling strategies hinges on the ambient conditions, levels of MP pollution, and the type of data desired. For example, passive sampling methods are more suitable than active ones in acquiring naturally depositing fluxes of MPs, whereas active samplers are generally more effective for detecting MPs present at low concentrations than passive ones. The absence of standardized atmospheric sampling protocols further complicates direct comparisons across studies, particularly when passive and active sampling approaches are used to quantify different metrics such as deposition fluxes and airborne concentrations.

Methods for collecting soil, sediment, and biota samples are largely similar to those for other pollutants [22,23]. Typically, soil samples are collected using stainless steel tools under random or composite sampling designs, followed by basic preprocessing such as drying, sieving, and density separation prior to analysis. Representativeness is a key factor for the success of a sampling campaign in attaining robust and useful data. Compared with aquatic environments, standardized protocols for sampling MPs in soils, sediments, plants, and terrestrial organisms remain less developed. Given that terrestrial ecosystems are increasingly recognized as important reservoirs and potential sources of MPs to aquatic environments, further methodological development in these compartments deserves greater attention.

4. Sample Pretreatment and Analytical Technologies

The most prevailing pretreatment methods are digestion and density separation through floatation [24]. For water samples, pretreatment typically involves filtration to collect particles, followed by oxidative or enzymatic digestion (e.g., H_2O_2 or Fenton reaction) to remove organic matter, and density separation through floatation [25]. For air samples, particles are commonly collected on filters using active or passive samplers, followed by visual sorting and digestion to eliminate organic interferences [26]. For solid samples (e.g., soil, sediment, and biota), pretreatment generally includes drying, sieving, and homogenization, followed by chemical or enzymatic digestion and density separation to isolate plastic particles from complex matrices [27]. These pretreatment procedures are designed to reduce matrix interferences and improve the reliability of subsequent particle identification and quantification (Figure 3).

Fourier transform infrared (FTIR) spectroscopy is one of the most important optical detection methods for qualitative analysis of MPs. The first application of FTIR in MP analysis can date back to the early 2000s when Thompson et al. [1] coined the term "microplastics" by presenting FTIR spectra of microscopic plastic debris collected in sediment from beaches and from estuarine and subtidal sediments around Plymouth, UK. Subsequent development of related techniques includes μFTIR and laser direct infrared (LDIR). FTIR techniques are generally reliable, efficient, time effective, and suitable for samples of various matrices, but they are costly, require complex sample pretreatment, and are incapable of distinguish closely adjacent particles [28,29].

Another popular optical detection method is Raman spectroscopy (and its variants), which was first employed by Cole et al. in 2013 [30] to identify MP debris extracted from zooplankton. Traditional Raman spectroscopy, though being better in terms of resolution than FTIR, is still constrained by the diffraction limit of laser and generally incapable of detecting nanoplastics. Its detection capability is also severely undermined by long measurement time and fluorescence and organic matter within nanoplastics [31]. Surface-enhanced Raman spectroscopy (SERS), through surface plasmon resonance of noble metals such as Au and Ag, can detect plastic debris of sizes as low as 20 nm [32]. But SERS also suffers from poor quantitative reliability, high cost, and limited spectral range, etc., which limits the utility of SERS in measurement of MPs in environmental samples. Amid these deficits, scanning electron microscope-Raman spectrometer has been a great alternative for qualitative and quantitative analyses and morphological characterization of nanoplastics [33]. Other coupling techniques such as photothermal manipulation-SERS and shrinking surface bubble deposition technology are also developed to separate, concentrate, and detect nanoplastics [34]. More recently, emerging techniques such as hyperspectral imaging, nano-FTIR, and machine learning-assisted image analysis have shown considerable potential for improving the sensitivity, resolution, and throughput of MP detection.

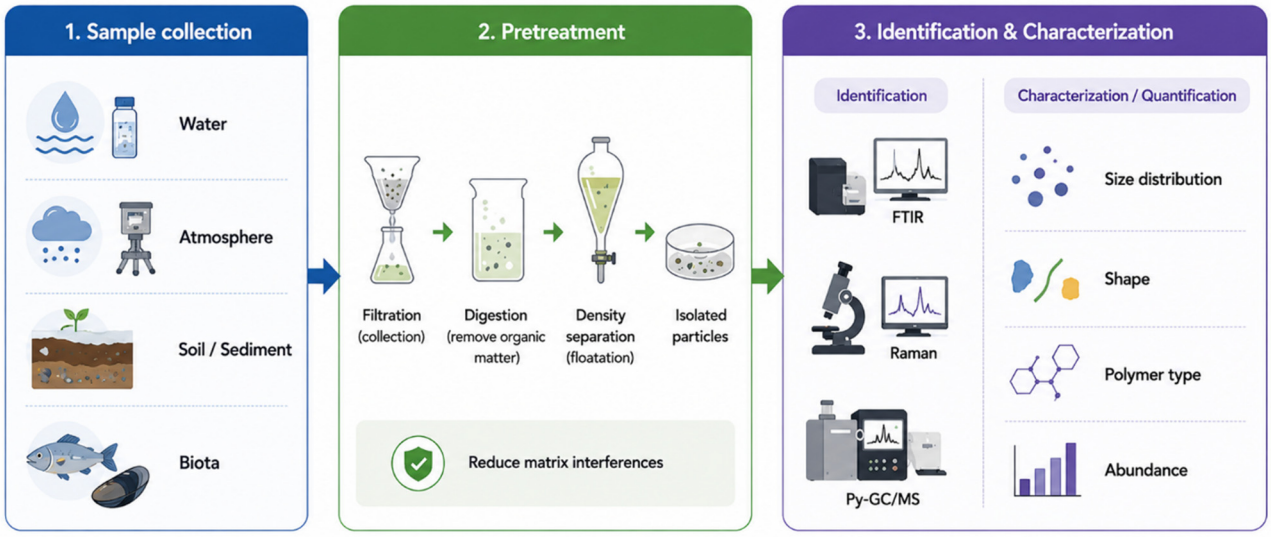

Figure 3. General workflow for microplastic sampling and analysis.

Pyrolysis-GC-MS, which was modernized in the 1960s–1970s to fingerprint non-volatile compounds and identify pyrolytic products [35], was first applied in measurement of MPs by Fries et al. [36]. Due to its high sensitivity and selectivity, pyrolysis-GC-MS is capable of qualitative and quantitative measurement of trace MPs in various environmental matrices. It is particularly advantageous to analyze samples with low concentrations of MPs, such as human specimens. On the other hand, pyrolysis-GC-MS is a destructive technique and therefore repeating analyses cannot be performed on the same sample. Pyrolysis-GC-MS is also unable to obtain information about particle size and morphology. Matrix interferences can also be significant, so substantial sample pretreatment is needed. Another method also based on weight loss is thermogravimetric analysis (TGA) [37], which is typically coupled with FTIR or mass spectrometry and able to process large samples from complex matrices. Similar to pyrolysis-GC-MS, TGA-based methods are also destructive techniques and are unable to acquire data on particle sizes and morphology.

5. Multi-Media Transport and Transformation of Microplastics

Upon entering to the environment, MPs may be subject to a series of physical, chemical, and biological alterations within and across various environmental compartments (Figure 4). As plastics are produced exclusively by humans and predominantly used inland, plastic debris are primarily sourced terrestrially and may undergo transport process, some of which find their way into the ocean. Understanding these transport pathways

is essential for identifying major sources of MPs and developing effective mitigation strategies before they enter aquatic and marine environments.

As plastics are produced, used, and discarded predominantly on land, terrestrial environments constitute the primary source and reservoir of plastic debris. Soil is the largest reservoir of plastic debris [38], while inland aquatic sediment is also an important sink [39]. The majority of plastic wastes entering the ocean originates from terrestrial sources and is transported through interconnected land–water systems. During precipitation events, MPs accumulated in soils can be mobilized by surface runoff and transferred into streams and rivers. Rivers are therefore widely recognized as the dominant pathway transporting land-derived MPs to estuaries and oceans. In aquatic systems, the transport of MPs is further influenced by hydrological conditions such as flow velocity, turbulence, and tides. Suspended MPs tend to migrate with water currents, whereas larger or denser particles are more likely to settle or become associated with sediments and aquatic organisms [40,41]. In addition to riverine transport, surface runoff and wastewater discharge can facilitate the redistribution of MPs from terrestrial to aquatic environments. Wastewater treatment plants receive large quantities of plastic debris from domestic and industrial sources. Although modern treatment facilities can remove a large fraction of MPs, with overall removal efficiency generally exceeding 90%, a considerable portion may still enter receiving waters through treated effluents or remain in sewage sludge that is later applied to agricultural soils, thereby promoting the continuous cycling of MPs between terrestrial and aquatic systems [42–44]. Atmospheric migration appears to play a relatively minor role in moving land-derived MPs to the ocean despite contradictory results being reported. Since atmosphere is highly mobile, atmospheric transport of MPs is dominated by diffusion. As a result, only light-density MPs (e.g., fibers) are profoundly detected in the atmosphere. Nevertheless, current estimates of atmospheric transport and deposition fluxes of MPs vary substantially across studies. The relative contribution of atmospheric deposition to oceanic plastic inputs remains uncertain compared with the better-established riverine transport pathway [45].

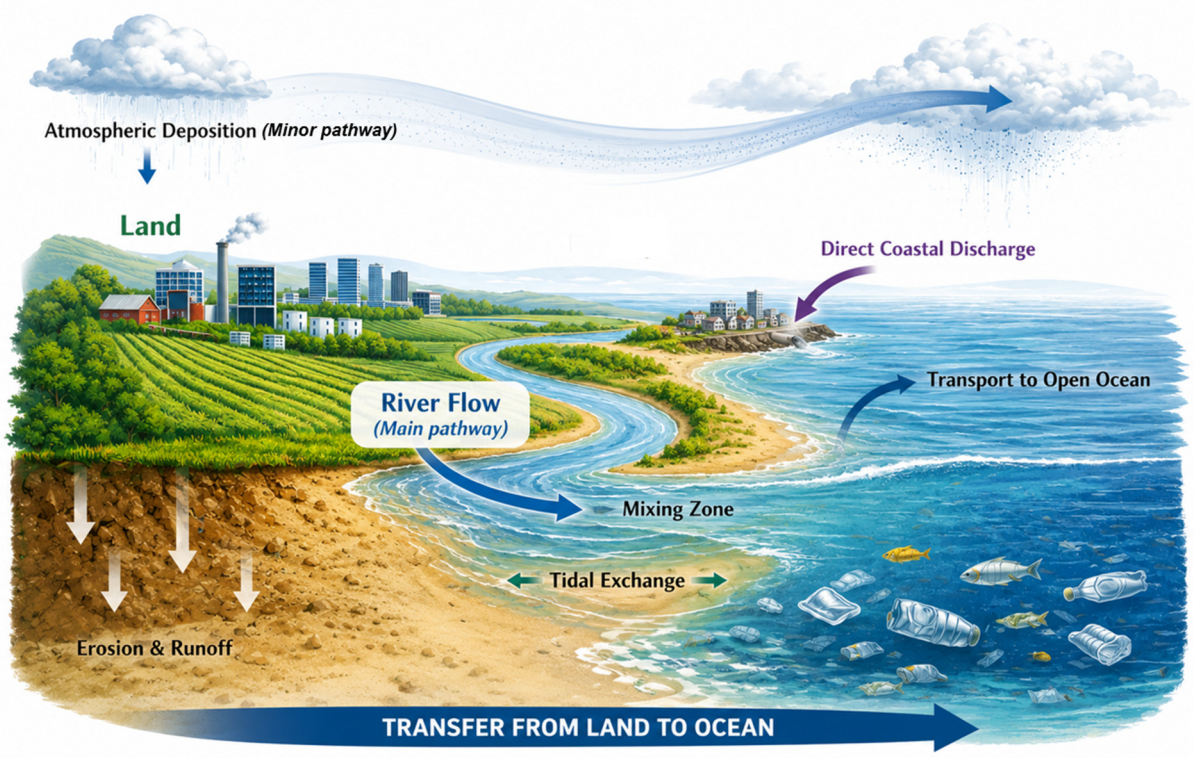

Figure 4. Schematic showing the multi-compartmental transfer of microplastics.

Plastic debris are subject to a number of aging processes such as physical fragmentation and/or photolytic degradation in the environment, resulting in enhanced hydrophilicity and chemical activity for MPs [46]. Such enhancement paves the way for subsequent abiotic and biotic transformation. Continuous exposure to ultraviolet radiation, temperature fluctuations, and mechanical stress may induce polymer chain scission and oxidative reactions, introducing oxygen-containing functional groups (e.g., carbonyl and hydroxyl groups) onto particle surfaces and increasing surface polarity and roughness [47]. As a result, aged MPs often interact more strongly with surrounding environmental media, including water molecules, natural organic matter, and mineral particles. Environmental aging may also modify the density and aggregation of MPs. Microbial

colonization on plastic surfaces can lead to the formation of biofilms, commonly referred to as the "plastisphere", which may increase the apparent density of particles and promote their vertical transport from surface waters to deeper layers or sediments. In addition, interactions between MPs and natural colloids or mineral particles may lead to hetero-aggregation and influence the environmental residence time and transport pathways of MPs [48,49]. Aging processes can also enhance the capacity of MPs to sorb contaminants by increasing the hydrophobic surfaces and specific surface areas of MPs, allowing MPs to act as vectors for transporting organic pollutants and heavy metals across environmental compartments [50,51].

Since the occurrence of MPs in marine organisms was first reported by Thompson et al. [1], abundant studies have accumulated large quantity of data to highlight the importance of bioaccumulation to MPs in aquatic species. For example, one of the earliest studies by Pegado et al. [52] demonstrated bioaccumulation of MPs in fishes (n = 26) from Amazon River estuary by linear correlation between the number of MP articles in their gastrointestinal tracts and fish standard lengths. Similar exposure has also been widely reported in avian species, where MPs are frequently detected in the digestive tracts of seabirds across different regions [53]. By contrast, biomagnification of MPs has been a debatable issue [49]. One clear piece of evidence is illustrated in the study of Alava et al. [54] who obtained contradictory results on projected loadings of MPs versus trophic levels in the cetaceans' food web of the Northeastern Pacific. The prevailing opinion about biomagnification of MPs suggests that it occurs with small MP particles [55,56]. Compared to conventional organic pollutants, MPs are particles and hence behave differently from that of organic pollutants which are of molecular sizes. The physiology is also distinctly different from that governing the biological processes of organic pollutants. These observations suggest that conventional concepts of bioaccumulation and biomagnification, originally developed for dissolved contaminants, may not fully apply to MPs and require further investigation.

Migration of MPs from the environment to human body is another hot research topic. Since the first detection of MPs in human stool by Schwabl et al. [57] and subsequently in human blood [58], the number of studies on MPs in human specimens have increased exponentially [59]. In addition to environmental exposure pathways, plastic food packaging materials have been identified as potential sources of MPs in food items. During food processing, storage, and preparation, packaging materials may release MPs through mechanical abrasion, thermal degradation, and/or repeated stress, particularly under conditions such as heating or prolonged storage [60–62]. These particles may migrate into food matrices and potentially carry plastic additives or sorbed contaminants, providing a pathway linking plastic pollution with human dietary exposure. However, results from different studies varied substantially, posing challenges for reliable human health risk assessments. The probable causes include small sample sizes, lack of standardized analytical protocols, insufficient quality control procedures, and potential cross contamination during sample collection and analysis [63]. Consequently, considerable uncertainties remain regarding the actual levels of human exposure and the associated health risks.

Taken together, the transport, transformation, and biological interactions of MPs are governed by a complex interplay of physical, chemical, and biological processes. However, substantial uncertainties remain regarding the relative importance of different transport pathways, the mechanisms controlling long-range migration, and the occurrence of trophic transfer and human exposure. Addressing these uncertainties will require greater harmonization of monitoring approaches and closer integration of field observations, laboratory experiments, and modeling studies.

6. How Hazardous Microplastics Really Are: From Ecotoxicity to Human Health Risk

Although it is difficult to identify the very first ecotoxicity test of MPs, a meaningful number of studies appeared only after the early 2010s. The toxicological disparities of microplastics induced by aging status, polymer composition and particle size are systematically summarized in Table 1. Earlier studies, however, often employed much higher concentrations of MPs (at mg L^{-1} level) than environmentally found levels in toxicity tests [55,64]. Use of environmentally unrealistic concentrations, as well as non-representative particle characteristics, of MPs has produced results that are often inconsistent with those observed in natural environment, which could partly explain why some early toxicity findings differ substantially from more recent studies conducted under environmentally relevant conditions.

More recent toxicity studies have generally used environmentally relevant concentrations of MPs for organism exposure. For example, Li et al. [65] examined the developmental toxicity of carboxylated PS MPs, generated from environmental aging processes, on zebrafish. The exposure experiment was conducted under environmentally relevant concentrations of carboxylated PS MPs, ranging from 0.1 to 100 μg L^{-1}. They found that exposure to these MPs could induce dose-dependent adverse effects, including growth inhibition and reduced tail coiling. Xiang et al. [66] demonstrated that exposure to UV-aged MPs resulted in greater adverse effects and a broader range of toxicity thresholds compared to virgin MPs. Another study by Ephsy and Raja [67] compared the toxicity of virgin and biodegraded MPs on fish, with MP concentrations at 1, 10, and 100 μg L^{-1} (environmentally

relevant concentrations). They found that virgin MPs could cause stronger dose-dependent toxicity, behavioural stress, and growth inhibition than biodegraded MPs. This finding demonstrated that microbial degradation of MPs has the potential to reduce MPs toxicity in the aquatic environment. These studies highlighted the importance of environmentally relevant exposure scenarios and particle aging status in determining MP toxicity.

Although most recent toxicity studies have applied environmentally relevant concentrations of MPs, toxic effects obtained are often inconsistent and sometimes even contradictory due to various characteristics of MPs (e.g., shape, size, and polymer type). Liu et al. [68] reported that PS significantly subdued the body weight of zebrafish, while PE considerably inhibited zebrafish's body length. The different toxicity effects may be caused by different toxic mechanisms of the two types of MPs on zebrafish. The toxicity of PS is mainly driven by the adsorption capacity conferred by its rigid benzene ring structures. In contrast, PE is more likely to accumulate in myocardial tissue due to its strong hydrophobicity, thereby intensifying cell apoptosis through mechanisms involving physical damage. Currently, nano- and micro-PS spheres are the most prevalent MPs used in toxicity studies [69–71], without considering the effects of shape, size, and polymer type on toxicity. A systematic meta-analysis demonstrated that the effects of biodegradable MPs on aquatic organisms were highly polymer-specific. For instance, polybutylene succinate and polyhydroxybutyrate impaired growth and behavior across multiple aquatic taxa (including plankton, invertebrates, and fish), while polylactic acid showed strong size-dependent toxicity [72]. Overall, apparent contradictions among toxicity studies may often reflect differences in particle characteristics and experimental design rather than genuinely conflicting biological responses.

Most ecotoxicity studies to date have focused on aquatic organisms, particularly fish and invertebrates. However, growing evidence suggests that MP exposure is not limited to aquatic ecosystems. MPs have also been detected in terrestrial organisms (e.g., plants and livestock), and even in human tissues, indicating that their environmental occurrence and biological effects extend across multiple ecosystems via various exposure pathways [73]. Compared with aquatic species, studies on MP exposure to terrestrial organisms remain relatively limited, and therefore ecological implications of MPs require further investigation.

Table 1. Summary on toxicity effects of microplastics (MPs) with different aging status, polymer types, and sizes.

Microplastic Characteristic	Toxicity Effect	References
Aging status	Compared with virgin MPs, aged MPs caused more severe intestinal mucosal damage, with additional vacuolization, inflammatory cell infiltration, and reduced mucus coverage on *gibel carp*	[66]
	Compared with virgin MPs, biodegraded MPs caused less toxicity on *Catla catla* fish, with less growth inhibition and weaker behavioral stress	[67]
Polymer type	Polystyrene (PS) MPs had a stronger inhibitory effect on body weight, whereas polyethylene (PE) MPs more strongly inhibited body length on *marine medaka*. Mechanistically, PS toxicity was linked more to its adsorption capacity from rigid benzene ring structures, while PE toxicity was associated more with its hydrophobicity, greater accumulation in myocardial tissue, and stronger induction of apoptosis through physical damage pathways.	[68]
	Polymer-specific risks were observed in biodegradable MPs: polybutylene succinate (PBS) and polyhydroxybutyrate (PHB) mainly impaired growth and behavior, while PHB and polyglycolic acid (PGA) more strongly reduced reproduction and survival. In contrast, polylactic acid (PLA) toxicity showed a clear size-dependent pattern and was less pronounced at environmentally relevant concentrations.	[72]
Size	Smaller PS beads were generally more toxic on *marine copepod*: 0.05 μm caused the strongest mortality, 0.5 μm had moderate delayed effects, and 6 μm had little effect on survival, though both 0.5 and 6 μm reduced fecundity.	[55,56]
	Smaller particle size increased toxicity on *sea cucumber*. Compared with larger PS microplastics, PS nanoplastics showed greater accumulation and caused stronger oxidative damage, immune suppression, disruption of cell proliferation and differentiation, and more severe disturbance of the intestinal microbiota–host symbiosis.	[69–71]

A major pathway connecting ecotoxicity to human health is through food systems. A recent review revealed the pathway for sorption of other toxic pollutants by MPs in agricultural soils. These loaded MPs can be taken up by

crops or ingested by grazing animals, carrying the toxic cocktail from the environment to our food tables. This process can directly link soil ecotoxicity to human dietary exposure [74]. In addition to dietary intake, inhalation of airborne MPs has also been recognized as an important human exposure pathway, as MPs are ubiquitous in indoor and outdoor air and can be directly inhaled into the respiratory system [75]. Microplastics can enter animals via ingestion, respiration, and contact, persisting in animal products and posing potential risks to human respiratory, digestive, and immune systems [76]. While current detection and removal technologies help ensure product safety, more efficient and eco-friendly methods are needed. The marine environment, where plastic debris were first discovered, provides some of the most well-documented evidences for linkage between ecotoxicity and human health. Bhuiyan et al. [77] showed that MPs can accumulate in humans through seafood intake. For example, MPs at levels of 0.2–5 particles g^{-1} tissue and 1–10 particles per individual were detected in bivalves (e.g., mussels and oysters) and commercial fish, respectively. In human feces, MPs at 5–10 particles per 10 gram feces were found. Although direct comparison of these findings is impossible, the detection of MPs in human feces reflects MPs exposure via food intake. In conclusion, these findings suggest a close linkage between environmental MP contamination, food systems, and human exposure. Contaminated environments (e.g., soil and water) would result in contaminated foods and consequently human bodies, with health hazards driven by identical toxicological mechanisms.

Toxicity endpoints used have varied greatly among different studies [72]. Thus, a same set of appropriate toxicity endpoints should be chosen for a certain species to allow a better comparison among different studies. To ensure the robustness and quality of MP toxicity data, toxicity studies should be conducted under environmental conditions. Therefore, MPs used in toxicity studies should not be pristine, spherical, single-sized beads. The target materials and concentrations must reflect the conditions of environmental MPs [78]. The majority of risk assessments note that environmentally observed MP concentrations are lower than those causing effects in laboratory studies. Microplastics at environmentally relevant exposure levels caused no effect on the survival, reproduction, and growth of freshwater benthic macroinvertebrates [79]. Although research into adverse effect mechanisms of MPs on organisms is progressing, more rigorous experimental designs are essential. For example, de Ruijter et al. [78] suggested that comparing MPs to similarly sized natural particles (e.g., fine stones) could effectively isolate the toxicity mechanism of food digestion (decomposition of MPs), where particle number concentration, rather than composition, drives toxic effects. Therefore, environmentally realistic exposure conditions and standardized toxicity endpoints are essential for improving comparability among studies and reducing uncertainty in risk assessment.

In recent years, nanoplastics have emerged as a new research focus in toxicity studies. Due to their much smaller size than MPs, nanoplastics exhibit enhanced mobility and can more readily penetrate biological barriers, including the intestinal epithelium, blood–brain barrier, and placental barrier, leading to wider distribution within organisms [80]. Nanoplastics also show higher potential for bioaccumulation in tissues and organs and may induce more pronounced toxic effects, partly due to their larger surface area-to-volume ratio and higher reactivity. Recent studies have further demonstrated that nanoplastics and MPs can accumulate in multiple organs and may be transferred across trophic levels, raising increasing concerns about their long-term ecological and health impacts [81].

7. Land-to-Sea Migration of Microplastics

As plastics are synthetic materials, MPs are almost exclusively generated on-land. As previously mentioned, soil is the largest reservoir of MPs. But a small portion of plastic debris still manage to find their ways to aquatic environments, and then pass through the coastlines and enter the oceans. As such, the ocean is considered the ultimate sink of MPs globally. Understanding land-to-sea transport pathways is important for identifying major sources of MPs and developing effective strategies to prevent plastic pollution before it enters aquatic and marine environments. As direct measurement of global land-to-sea plastic fluxes is extremely difficult, existing estimates rely primarily on modeling approaches. However, published estimates vary substantially among studies owing to differences in predictor selection, transport assumptions, and model calibration. Some models estimate plastic emissions based on mismanaged plastic waste (MPW), or socioeconomic variables, whereas other employ river-based transport models constrained by monitoring data. These methodological differences have contributed substantially to the large discrepancies among published estimates of global plastic emissions. Various estimates of global riverine plastic inputs to the ocean derived from MPW and HDI parameters are systematically summarized in Table 2.

A benchmark accomplishment on the subject was published in 2015 by Jamback et al. [82], who modeled the migration of plastic waste from land-originated sources to the ocean based on geographical locality, socioeconomic prosperity, and waste management efficiency which were collectively used to derive as a new concept of Mismanaged Plastic Waste (MPW). Defining the coastal zones as those within 50 km from the coastlines and assuming the precent of plastic waste disposed in river basins, due to inadequate waste management, that would

eventually enter the ocean was 15%, 25%, and 40%, Jamback et al. [82] were able to estimate the annual global land-sea input of plastic waste to be 4.8–12.7 mMT, with a median value of 8 mMT. As it has turned out, 8 mMT has been widely quoted by a large number of media coverages and research publications to indicate the severity of marine plastic pollution. However, the estimate was derived largely from assumptions regarding MPW generation and transport efficiency rather than direct observational constraints, which has led to substantial debate in subsequent studies. Numerous subsequent studies have intentionally or unintentionally contradicted this result, resulting in ongoing debates and investigations into the topic, which is undoubtedly healthy for promoting better science.

As rivers are the predominant pathways for transporting land-generated MPs to the ocean [82,83], a large number of field measurements and modeling studies have been conducted and already accumulated abundant data. The earliest modeling study by Lebreton et al. [84] built on the concept of MPW, the mass of which for each watershed was estimated from the rate of generating MPW, population density, and topography of artificial barriers within the watershed. They employed a regression model ($M_{out} = (k\ M_{mpw}\ R)^a$, where M_{out} is the plastic mass discharged at the river mouth (kg day^{-1}), M_{mpw} is the mass of MPW produced inside the river basin downstream of any artificial barriers, R is the monthly averaged river water flow runoff, and k and a are the regression parameters) calibrated against prescreened field data, yielded a range of 1.15–2.41 mMT for global riverine inputs of plastic waste to the ocean every year.

Almost simultaneously, Schmidt et al. [85] adopted a similar approach based on MPW to estimate river inputs of plastic waste to the ocean. They developed two models which differed in the ways of how field data were used to regress the prediction parameters. The two models obtained considerably different estimates. Median values of 0.47 and 2.75 mMT were obtained for Model 1 (25–75% prediction intervals: 0.21–1.12 mMT) and Model 2 (25–75% prediction intervals: 1.72–4.38 mMT), respectively. Despite the substantial difference between the two models, the predictions by Schmidt et al. [85] bracketed those by Lebreton et al. [84]. This result should not be unexpected, as both studies employed MPW as the main predictor with variations probably stemming from the use of different datasets. Another MPW-based model by Meijer et al. [86] yielded a range of 0.8 and 2.7 mMT per year emitted from more than 1000 large rivers, which accounted for 80% of the global riverine plastic emissions.

A different model predictor, Human Development Index (HDI) published annually by the United Nations Development Program [87], was employed by Mai et al. [88] to estimate global riverine plastic outflows. The HDI is a measure of education level, economic prosperity, and living standard for a country/region, and therefore can be more reflective of the extent of adequacy in waste management. Along with data of riverine discharge, population density, etc. from 1518 river across the globe, the HDI-based model predicted the amounts of riverine plastic input from rivers to the ocean to range from 0.055–0.245 (midpoint: 0.127) mMT. An update of the HDI-based model including all rivers of 161 countries expanded the range to 0.15–0.53 mMT in 2016 [89]. The HDI-based model generally yields much smaller estimates of the global riverine plastic emissions than the MPW-based models, which was attributed largely to the mechanistically different strategies in assigning HDI or MPW values to countries across the globe [88,89]. It should be noted that the HDI-based model tends to obtain lower overall riverine inputs than the MPW-based models. One of the causes is probably the smaller range of 1-HDI used in the regression than that of MPW [88]. Nevertheless, estimates of individual riverine plastic inputs have remained highly uncertain [90].

Riverine runoff is believed to transport the majority of land-derived plastic waste to the ocean [84,85]. Therefore, there is clearly a large gap between the model estimates of land-to-sea inputs (4.8–12.7 mMT) [82] and riverine emissions (0.15–2.75 mMT) [84–86,89]. Among all technical deficits, one main limitation perhaps is the lack of field data to calibrate/validate Jamback et al.'s model. This seemed to cause two technical issues. The first issue is the assignment of divergent MPW values to countries/regions of different economic prosperities, which seem to be more favorable to developed countries than underdeveloped ones. Another issue is the percentage of MPW generated in a watershed that can be carried into the ocean was determined based on monitoring data collected in the San Francisco Bay watershed (71 municipalities) and set as 15%, 25%, and 40% for all watersheds targeted in the model.

The large discrepancies among these estimates are closely linked to differences in model assumptions, predictor selections, and calibration strategies. For example, the model of Jambeck et al. [82] assumes that 15–40% of MPW generated within coastal watersheds ultimately reaches the ocean, whereas river-based models estimate transport efficiencies indirectly from river discharge and field observations. In addition, MPW-based and HDI-based models differ fundamentally in how plastic leakage is assigned across countries and regions. These methodological differences can propagate through the modeling framework and ultimately result in order-of-magnitude differences in predicted land-to-sea plastic fluxes.

Research on atmospheric transport of MPs began much later than those in other areas. The study of Allen et al. [45] on atmospheric MPs unveiled the significance of atmospheric deposition of MPs even in a remote region, paving the way for subsequent investigations into the

migration of MPs from land to the ocean. A summary of available data on atmospheric transport of MPs obtained a range of 0.013–25 mMT for annual fluxes of MPs to the ocean [91]. The wide range of the amounts of MPs transported atmospherically to the ocean probably reflect the large variability and uncertainty related to the model parameters employed. The extraordinarily high maximum value is out of proportion of common-sense expectation. A subsequent modeling study [92] suggests a much lower value of 0.025 mMT. Other regional studies also obtained lower estimates, such as Chen et al. [93] who obtained approximately 0.009 mMT of atmospheric transport of MPs to the ocean in the south hemisphere.

Overall, accumulated evidence suggests that riverine transport remains the dominant pathway for land-to-sea migration of MPs, whereas atmospheric transport likely represents a secondary pathway. Large discrepancies among published estimates arise primarily from differences in predictor selection, transport assumptions, and model calibration. MPW-based models generally yield higher estimates than HDI-based models and river-calibrated models, highlighting the uncertainty that remains in estimating global plastic emissions.

Table 2. Various estimates of global riverine plastic inputs to the ocean based on Mismanaged Plastic Waste (MPW) and Human Development Index (HDI).

References	Transport	Lower Point (Mt yr^{-1})	Mid Point (Mt yr^{-1})	Higher Point (Mt yr^{-1})	Main Predictor	Major Uncertainty
Jambeck et al. 2015 [82]	Plastics from land to ocean	4.8	–	12.7	MPW	MPW estimation; ocean-entry assumptions
Lebreton et al. 2017 [84]	Plastics from rivers to oceans	1.15	–	2.41	MPW	River export parameters; calibration data limitations
Schmidt et al. 2017 [85]	Plastics from rivers to seas	0.47	–	2.75	MPW	River data selection sensitivity
Lebreton and Andrady 2019 [94]	Plastics from land to seas	3.1	5.1	8.2	MPW	Future waste-generation assumptions
Mai et al. 2020 [88]	Plastics from rivers to seas	0.057	0.134	0.265	HDI	HDI-based socioeconomic assumptions
Meijer et al. 2021 [86]	Plastics from rivers to oceans	0.8	–	2.7	MPW	River transport parameterization
Weiss et al. 2021 [83]	Microplastics from rivers to oceans	–	6.1×10^{-3}	–	Population density and drainage intensity	Emission factor uncertainty
Zhang et al. 2023 [95]	Plastic emissions to oceans	0.13	0.7	3.8	Observed dataset of sea surface plastic concentrations and an ensemble of ocean transport models	Ocean transport model uncertainty; observational data limitations

*Note: Mt yr^{-1} = million metric tons per year.

8. Conclusions

So far numerous data have been accumulated on MP pollution, which has greatly advanced our understanding of the sources, transport pathways, environmental fate, and health effects of plastic particles. At the same time, we have also witnessed numerous publications with contradictory or contrasting findings, some of which are even out of proportion of common knowledge. Many of these discrepancies arise from differences in definitions, sampling strategies, analytical methodologies, experimental designs, and modeling assumptions. Recognizing and critically evaluating these sources of uncertainty is essential for developing a more coherent understanding of MP pollution at regional and global scales. The historical evolution of MP research demonstrates a transition from the discovery of plastic debris in the environment to increasingly sophisticated investigations of transport, transformation, ecotoxicity, and human exposure. While caution is necessary in assessing the reliability of large influxes of new data on the topic, additional and precise efforts should be made to continue the journey of plastic research. This not only fulfills the curiosity of mankind but more importantly also unveils the fundamental characteristics of MP pollution and its spatiotemporal trends, propelling the adoption of integrated scientific and regulatory measures for better utilization of plastic products while minimizing their negative consequences. Addressing MP pollution will require not only scientific advances but also effective mitigation strategies and coordinated regulatory efforts at regional and global scales. In light of these challenges, three priority areas for future research are proposed: (i) Standardization of definition, sampling, and analytical protocols to enhance data comparability; (ii) toxicity assessments incorporating environmentally aged MPs to better capture realistic exposure conditions; and (iii) field validation of cross-compartment transport and fluxes across environmental systems, and improved model development to reduce uncertainties in global MP flux estimates.

Author Contributions

E.Z.: conceptualization, supervision, project administration, funding acquisition; Z.X. and L.M.:

investigation, literature collection, writing—original draft preparation; L.B., Y.R. and E.Z.: reviewing and editing. All authors have read and agreed to the published version of the manuscript.

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Use of AI and AI-Assisted Technologies

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