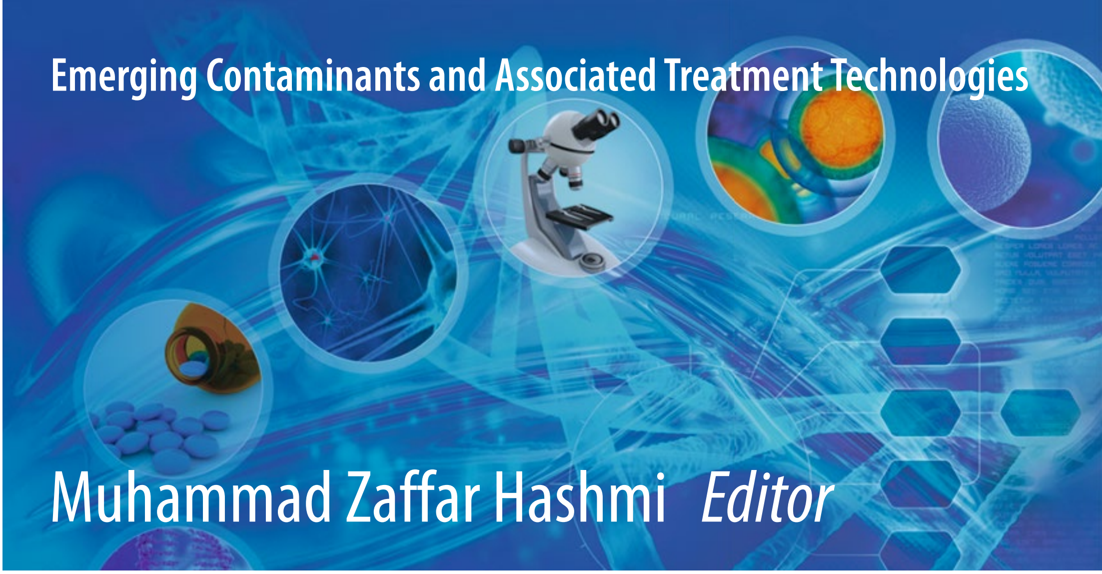
Emerging Contaminants and Associated Treatment Technologies



Muhammad Zaffar Hashmi *Editor*

Microplastic Pollution

Environmental Occurrence and Treatment Technologies

Springer

Emerging Contaminants and Associated Treatment Technologies

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Foreword

The future… Is this question going to be answered the way we imagined, or are we going to struggle with the unforeseen? It is both an easy and difficult question to answer.

The year 2030, as we have seen it in the games we used to play in our childhood, was the date when flying cars and even life in space would be possible in our dreams. Furthermore, the planets to travel were listed for the year 1999 that we associated with the space series. When the millennium was perhaps too distant to reach, we used to dream about how old we would be and what we would do in those years. We used to dream about scientific discoveries and design the technological devices we would use. The day has come, the dates have come closer, and dreams have come true one by one. The future we planned, and the realities of that future manifested themselves as “dreams that came true,” albeit in different ways. However, another issue, which is far from our dreams and unpredictable, has arisen; A “troubled” Earth… And unfortunately, one of the biggest problems of this world has been “plastics.”

Decades ago, warnings about the global food problem, energy problem, and water crisis were far-fledged. Forthcoming challenges and notable warnings went unnoticed and sometimes ignored. We also ignored the pollution that could be seen even with naked eyes. We were not worried about the future because we had very solid arguments like “nature does her job” or “nature cleans herself.” After all, nature has always been a friend to us, and she has always been a provider. However, nature has never liked or failed to love the most difficult equation to solve, namely plastics. Nowadays, this much-debated difficult equation has the smallest dimension: microplastics. So how has the plastic problem become inextricable day by day? And more importantly, how come that there is still no solution to this problem? What can human beings do under the title of “one health” in the human—food—ecosystem triad? How can human beings eliminate this problem, which nature does not like, with environmentally friendly solutions?

This book aims to examine the micro-size pollution caused in the ecosystems by the commonly used polymer materials, including special applications due to their light structure, durability, and low price based on environmental formation and treatment technologies. From Asia to Europe, freshwater to marine water, air to seafood, microplastic formations have been documented in every aspect of the ecosystem. Extraction, enumeration, and identification, which are among the most difficult methods, and the impacts of plasticizers have been detailed, and the difficulties encountered, and available gaps have been scrutinized. The importance of monitoring studies in microplastic pollution, microplastic transport in water-air-food, possible toxicity of microplastics, and risk assessments were interpreted, and their epidemiological, ecological, and public health effects were evaluated. It is aimed to explain possible solutions for microplastic pollution by not only chemical but also photo treatment and green treatment technologies. Since the challenges of preventing microplastic pollution are well-known, both national and international policies and legislations for microplastic pollution have been enforced. The chapters of this book, written by experts and distinguished authorities, present the studies carried out on microplastics and the measures that need to be taken.

An inter- and transdisciplinary endeavor is required to tackle the microplastics issue, given the magnitude of the problem in terms of material attributes, sampling and analytical difficulties, ecological interactions, ecotoxicity, and risk assessment. I hope this book will foster and inspire such a perspective. I would like to extend my gratitude to our editor, reviewers, the publisher, and all funders for supporting us so that this book could be successfully published (and open access).

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Preface

Microplastics are fragments of any type of plastic less than 5 mm (0.20 in) in length. They enter natural ecosystems from a variety of sources, including cosmetics, clothing, and industrial processes. Microplastics may be primary microplastics that include any plastic fragments or particles that are already 5.0 mm in size or less before entering the environment. These include microfibers from clothing, microbeads, and plastic pellets (also known as nurdles). Or secondary microplastics arise from the degradation (breakdown) of larger plastic products through natural weathering processes after entering the environment. Such sources of secondary microplastics include water and soda bottles, fishing nets, plastic bags, microwave containers, and tea bags. Both types are recognized to persist in the environment at high levels, particularly in aquatic and marine ecosystems. However, microplastics also accumulate in the air and terrestrial ecosystems. Because plastics degrade slowly (often over hundreds to thousands of years), microplastics have a high probability of ingestion, incorporation into, and accumulation in the bodies and tissues of many organisms. The toxic chemicals that come from both the ocean and runoff can also biomagnify up the food chain. The book consists of four parts: introduction to plastic, environmental occurrence, risk assessment and health impact, and treatment technologies. The book focuses on the emergence of plastic pollution, types, sources and fate, dynamic trends in the environment, occurrence in different environmental compartments, toxicity and risk assessment, and prevention strategies. Microplastic pollutants is currently an important topic in both industry and academia, as well as among legislative bodies, and research in this area is gaining considerable attention from both the worldwide media and scientific community on a rapidly increasing scale. Ultimately, this book provides an excellent source of reference and information on microplastics for scientists, engineers, students, industry, policy makers, and citizens alike.

Islamabad, Pakistan — Muhammad Zaffar Hashmi

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Part I
Introduction to Plastic pollution

Chapter 1
Emerging Issue of Microplastic in Sediments and Surface Water in South Asia: A Review of Status, Research Needs, and Data Gaps

Jalal Bayar, Muhammad Zaffar Hashmi, Muhammad Abdullah Khan, Siwatt Pongpiachan, Xiaomei Su, and Paromita Chakaraborty

Abstract Plastic particles <5 mm come under the category of microplastics (MPs) that can be primary or secondary in nature. Microplastic pollution is a major concern because the world's shores served as a major sink. Previously, the researchers focused on marine ecosystem, whereas the data on beach sediments and water are limited, especially in South Asia. Several research articles have been published in South Asia, including India, Pakistan, Sri Lanka, Maldives, and Bangladesh. Furthermore, Nepal, Afghanistan, and Bhutan lack information regarding MPs on beach sediments and surface water. Therefore, the review in South Asia will help mitigate and raise awareness on the severity of MPs within the research community and local public. Here, we review the abundance, fate, spatial distribution, research need, and gaps regarding MPs. In Pakistan, only two research articles are published despite the higher concentration of MPs there compared with the other South Asian

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countries. High concentrations of MPs on sediments (3726 particles/m^2) and fresh water (2074 particles/m^3) were observed in Ravi River, Lahore, whereas low MP concentrations on sediments (22.8 particles/m^2) and fresh water (0.32/m^3) were observed in Faafu Atoll, Maldives. In terms of shape, fragments were dominant with the polyethylene polymer type. MP pollution mainly depends on population density, where land-based sources, i.e., industrial, municipal, fishing, tourism, and recreational activities, are the major contributors. Consumption of MPs is life-threatening. The chronic biological effects in aquatic organisms are due to the accumulation of MPs in their cells and tissues. Understanding of these areas is essential to raise awareness and establish management policies for decision-making in this perspective.

Keywords Distribution · Sources · Microplastics · South Asia

1.1 Introduction

Microplastics (MPs) refer to particles that are less than 5 mm and are not visible to the naked eye proclaim a major component as worldwide environmental challenge (De Falco et al. 2018; Koelmans et al. 2019). They can be classified into primary and secondary MPs based on their sources (Strungaru et al. 2019; Cole et al. 2011). Primary MPs are used in several cosmetic products, i.e., scrubs (Napper et al. 2015), while secondary MPs are produced by the breakdown of large MPs in the environment (Wagner et al. 2014).

MPs enter the coastal water through winds, currents, tides, and waves, which causes them to be carried out over long distances from their original point (Veerasingam et al. 2016a; Kim et al. 2015; Eriksen et al. 2014) and be found in remote areas (Saliu et al. 2018) where beaches are considered to be a major sink for plastic debris (GESAMP 2015). It is projected that 80% of plastics in the marine water originate from land-based sources and are transported by rivers and eventually flow through the sea (Jambeck et al. 2015). Disasters, such as flood and tsunami, can also transport plastics to the aquatic environment (Veerasingam et al. 2016b).

Different polymer types of MPs (i.e., polyethylene, polyvinyl chloride, polyethylene terephthalate, polystyrene, and polypropylene) depicts additional information regarding its physicochemical properties and degradation rate of MPs (Wagner and Lambert 2018).

It has been reported that the degradation of plastic from large to small particles (Yonkos et al. 2014; Wessel et al. 2016) increases susceptibility to aquatic organisms (Cole et al. 2011) and the ingestion of MPs by freshwater organisms(Imhof et al. 2013; Sanchez et al. 2014). The adsorption of organic pollutants on MPs also increases contamination of the food web of aquatic organisms (Sruthy and Ramasamy 2017). Plastics comprised of different additives, i.e., bisphenol A (BPA) and phthalates. These additives may adversely affect the aquatic environment and ultimately the consumers if released during the degradation process.

MPs from waste generated by industrial plants, wastewater treatment plants, and municipal solid waste (Lechner et al. 2014; Mason et al. 2016; Fendall and Sewell 2009) might entangle with different terrestrial and aquatic organisms, which can cause mortalities (Ryan 2018; Borunda 2019; Gregory 2009; Votier et al. 2011; Nunes et al. 2018). MPs are pollutants consisting of different additives, i.e., hydrocarbons and flame retardants, which might be released into the environment via degradation. Moreover, MPs also act as vectors that adsorb POPs from the environment, are consumed by aquatic biota, and reduce health and biodiversity (Veerasingam et al. 2018).

Environmental pollution caused by MPs (<5 mm) is an increasing concern that has not been tackled on in South Asia. Several studies on freshwater and beach sediments have been conducted in South Asia, including India (Veerasingam et al. 2016a, b; Sruthy and Ramasamy 2017; Dowarah and Devipriya 2019; Jeyasanta et al. 2020; Karthik et al. 2018; Vidyasakar et al. 2018; Robin et al. 2020; Gopinath et al. 2020; Vidyasakar et al. 2020; Patchaiyappan et al. 2020), Sri Lanka (Viraj et al. 2019; Weerakoon et al. 2018a; Athawuda et al. 2020; Koongolla et al. 2018; Weerakoon et al. 2018b), Bangladesh (Hossain et al. 2020; Karim et al. 2020; Sarker et al. 2020), Pakistan (Irfan et al. 2020a, b), and Maldives (Saliu et al. 2018; Imhof et al. 2017), which indicates the significant amount of MPs in the environmental compartments. The current review is the first to highlight the status, research needs, and data gaps of MPs in environmental matrices in South Asia that pose tremendous health hazard to aquatic and terrestrial lives. It was found that South Asia is the main source of plastic debris worldwide (Duhec et al. 2015).

Almost all countries in South Asia are third-world or developing countries. Thus, the pollution caused by MPs is high in undeveloped areas due to the lack of proper waste management techniques, which will cause a large amount of plastic from land to be transported to the ocean by 2025 (Jambeck et al. 2015). Extensive and in-depth studies are urgently needed to justify the knowledge gaps through scientific research to raise awareness among the people of the threat of MPs that are significantly hazardous to marine biota and terrestrial life, as well as to support the development of policies addressing this issue.

The objective of this review is to determine (1) the abundance of MPs, (2) their distribution and sources in South Asia, (3) the possible health risks, (4) and the research gaps regarding MP pollution in South Asia.

1.2 Microplastic Distribution in the Water Bodies of South Asia

Fresh waters consist of only 0.01% of the water on Earth, with rivers, reservoirs, and lakes covering around 2.3% (and freshwater wetlands estimated at 5.4%–6.8%) of the global land surface area, excluding large ice sheets (Lehner and Döll 2004).

Surface waters are contaminated with an increasing diversity of anthropogenic compounds, giving rise to the presence of complex contaminant mixtures that can cause serious harm to aquatic ecosystems (Vörösmarty et al. 2010; Schwarzenbach et al. 2006; Bernhardt et al. 2017).

Fresh water is important to sustain life, and it mainly comes out from the surface and underground aquifers near rivers and canals. The quality of surface water has been declining continuously due to the accumulation of raw municipal and industrial effluents in water (Chilton 2000). Most of the rivers are extended and diluted and cannot sustain aquatic life. These water bodies are highly contaminated and need proper management to free them from contaminants, i.e., MPs, so that they can be used by humans.

MPs generated from anthropogenic, industrial, and municipal activities that enter the water through winds, currents, tides, and waves could be carried out over long distances from their original point (Veerasingam et al. 2016a; Kim et al. 2015; Eriksen et al. 2014). MPs were studied in different countries in South Asia. The results suggested that the MP level is high in water. For example, MPs in freshwater were studied by (Irfan et al. 2020a, b) in Pakistan, and it was found that the MP concentration is higher in the Ravi River (2074 MPs/m^3) compared with Rawal Lake (1420 MPs/m^3); moreover, the concentration of MP is higher off Colombo (140.34 MPs/m^3) than in the southeast coast of Sri Lanka (0–29 items/m^3) (Athawuda et al. 2020; Koongolla et al. 2018). MPs were quantified by (Robin et al. 2020) in India and by (Saliu et al. 2018) in Maldives, and it was found that the MP concentration is high in the southwest coast of India (1.25 particles/m^3) compared with Faafu Atoll, Maldives (0.32 particles/m^3) (Fig. 1.1). The results suggested that these high levels could influence the water quality in these countries.

1.3 Microplastics in the Sediments of South Asia

Sediments are an important part of the aquatic environment as they transport a large portion of nutrients and contaminants. Sediments facilitate nutrient uptake, storage, release, and transfer between environmental matrices. The erosion of bedrock and organic component while undergoing degradation derives the sediments (WHO 1996). The deposition of MPs in sediments has an adverse effect on microbial community and nitrogen cycling (Seeley et al. 2020).

There are different pathways through which the most persistent pollutants enter the ocean waters and beaches (Villarrubia-Gómez et al. 2018). The plastics that accumulate in sediments might originate from land and sea-based sources that adversely affect life through entanglement with different organisms and ingestion, which could affect the reproduction system and lead to transportation of nonnative species and habitat degradation (Oehlmann et al. 2009). Poor waste management techniques play an important role in the deposition of plastic debris in the aquatic environment, including oceans, deep-sea sediments, beaches, and freshwater lakes (Jayasiri et al. 2013; Ballent et al. 2016).

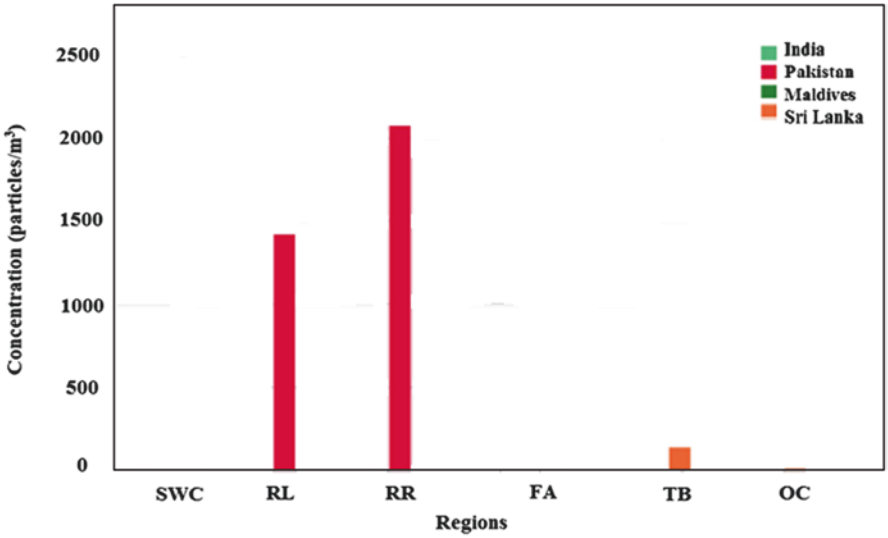

Fig. 1.1 Bar graph comparing the mean concentration of MPs in water (m^3) across different countries. *SWC* Southwest coast, *RL* Rawal Lake, *RR* Ravi River, *FA* Faafu Atoll, *TB* Thumpalai Beach, *OC* off Colombo

MPs act as toxicants to life in two ways. One is that MPs adsorb toxic chemicals in the ecosystems, thus serving as vectors of transport. The second is that MPs are mixed with dangerous chemicals to increase their elastic characteristics and prolong their shelf life. To date, there is a lack of knowledge on the major additives that are used in the plastic industry, on their fate once MPs are released into the environment, and on their consequent effects on human health when associated with micro- and nanoplastics (Campanale et al. 2020).

The MPs in India were evaluated in the sediments by (Sruthy and Ramasamy 2017; Dowarah and Devipriya 2019; Jeyasanta et al. 2020; Karthik et al. 2018; Vidyasakar et al. 2018; Robin et al. 2020; Vidyasakar et al. 2020; Tiwari et al. 2019; Sarkar et al. 2019), in Pakistan by (Irfan et al. 2020a, b), in Sri Lanka by (Viraj et al. 2019; Koongolla et al. 2018), in Maldives by (Saliu et al. 2018; Imhof et al. 2017), and in Bangladesh by (Balasubramaniam and Phillott 2016), which are shown in Table 1.1. The highest concentration in India was found in Vembanad Lake (296 particles/m^2), whereas the least was found in the southwest coast of India (40.7 particles/m^2) (Fig. 1.2). In terms of kg, the highest concentration was found in Palolem Beach (520 particles/kg), whereas the least was found in the coast of Danushkodi in India (45 particles/kg) (Fig. 1.3). In terms of m^2, among all other countries, the highest concentration was found in Pakistan (3726 particles/m^2) (Fig. 1.2). In Sri Lanka, the concentration is relatively high as well in Thumpalai Beach, i.e., 384 particles/kg (Fig. 1.3). In Maldives, the concentration was found least in Faafu Atoll (22.8 particles/m^2) compared with the Indian Ocean (1029 particles/m^2). In Bangladesh, at the beach of Cox's Bazar, the concentration was found

Table 1.1 The abundance of plastics/plastic debris/microplastics in beach sediments and surface water around the South Asia

Location	Sample type	Concentration	Polymer type	Shape	References
Rawal Lake, Pakistan	Water	0.142 items/0.1 L	PE, PP, PET, PVC, polyester	Fibers, fragments	Irfan et al. (2020a)
Rawal Lake, Pakistan	Sediments	1.04 items/0.01 kg	PE, PP, PET, PVC, polyester	Fibers, fragments	Irfan et al. (2020a)
Southeast Coast, India	Sediments	2–178/m^2	PE, PP	Fragments	Karthik et al. (2018)
Vembanad Lake, India	Sediments	96–496 particles/m^2	LDPE, HDPE, PP, PS	Films, foams	Sruthy and Ramasamy (2017)
Lahore, Pakistan	Sediments	3726 ± 9030 MPs/m^2	PE, PP, PS	Fragments, fibers	Irfan et al. (2020b)
Lahore, Pakistan	Water	2074 ± 3651 MPs/m^3	PE, PP, PS	Fragments, fibers	Irfan et al. (2020b)
India Ocean, Maldives	Sediments	1029 ± 1134 particles/m^2	PE, PP, PS	Expanded polystyrene, fragments	Imhof et al. (2017)
Silver Beach, India	Sediments	204 particles/kg	PVC, PE	Fragments, pellets, fibers	Vidyasakar et al. (2020)
Dhanushkodi Coast, India	Sediments	45 ± 12 particles/kg	PE, PP, PET, PS	Fibers, granules	Tiwari et al. (2019)
Mumbai Coast, India	Sediments	220 ± 50 particles/kg	PE, PP, PET, PS	Fibers, granules	Tiwari et al. (2019)
Tuticorin Coast, India	Sediments	181 ± 60 particles/kg	PE, PP, PET, PS	Fibers, granules	Tiwari et al. (2019)
Puducherry Coast, India	Sediments	720.30 ± 191.60 particles/kg	PP, HDPE, LDPE, PS, polyurethane	Fragments, fibers, films, foams, pellets	Dowarah and Devipriya (2019)
South Andaman Beach, India	Sediments	414.35 ± 87.4 particles/kg	PVC, melamine, PP, polysulfide, etc.	Fragments, fibers	Patchaiyappan et al. (2020)
Rameswaram Coral Island, India	Sediments	403 pieces	PP, PE, PS, nylon, PVC	Irregular shapes, fibers, pellets	Vidyasakar et al. (2018)
Thumpalai Beach, Sri Lanka	Sediments	9.6/25 g	None found	Fibers	Balasubramaniam and Phillott (2016)
Hawke's Bay Beach, Pakistan	Sediments	12.0/25 g	None found	Fibers	Balasubramaniam and Phillott (2016)

(continued)

Table 1.1 (continued)

Location	Sample type	Concentration	Polymer type	Shape	References
Dhunikolhu, Maldives	Sediments	4.3/25 g	None found	Fibers	Balasubramaniam and Phillott (2016)
Palolem Beach, India	Sediments	13.0/25 g	None found	Fibers	Balasubramaniam and Phillott (2016)
Cox's Bazar, Bangladesh	Sediments	12.3/25 g	None found	Fibers	Balasubramaniam and Phillott (2016)
Ganga River, India	Sediments	99.27–409.86 items/kg	PE, PP, PET, PS	Fibers, filaments, foam, films, fragments	Sarkar et al. (2019)
Off Colombo, Sri Lanka	Water	140.34 ± 15.23 items/m^3	None found	Filaments, fragments, films	Athawuda et al. (2020)
Tuticorin district, India	Sediments	25 ± 1.58 to 83 ± 49 items/m^2	PE, PP, PET, NY, PVC, PS	Fragments, film, fibers	Jeyasanta et al. (2020)
Southeast coast, Sri Lanka	Water	0–29 items/m^3	PE, PP, PS	Fragments, pellets, line, foams, films	Koongolla et al. (2018)
Southeast coast, Sri Lanka	Sediments	0–157 (±94) items/m^2	PE, PP, PS	Fragments, pellets, line, foams, films	Koongolla et al. (2018)
Southwest coast, India	Water	1.25 ± 0.88 particles/m^3	PE, PP	Fragments, fibers/line, foams	Robin et al. (2020)
Southwest coast, India	Sediments	40.7 ± 33.2 particles/m^2	PE, PP	Fragments, fibers/line, foams	Robin et al. (2020)
Faafu Atoll, Maldives	Water	0.32 ± 0.15 particles/m^3	PE, PP, PS, PVC, PS, polyamide	Fragments, foams, filament, pellets, films, char/tar	Saliu et al. (2018)
Faafu Atoll, Maldives	Sediments	22.8 ± 10.5 particles/m^2	PE, PP, PS, PVC, PS, polyamide	Fragments, foams, filaments, pellets, films, char/tar	Saliu et al. (2018)

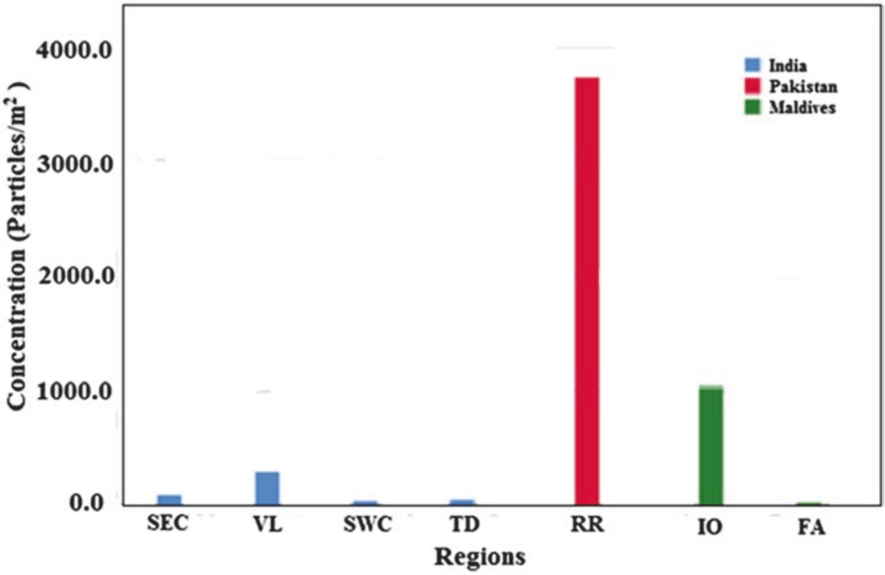

Fig. 1.2 Bar graph comparing mean concentration of MPs in sediments (m^2) across different countries. *SEC* Southeast coast, *VL* Vembanad Lake, *SWC* Southwest coast, *RR* Ravi river, *IO* Indian Ocean, *FA* Faafu Atoll)

to be 492 particles/kg (Fig. 1.3). Pakistan is contributing highly in contamination of MPs. Only two publication have published in Pakistan on MPs contamination.

1.4 Abundance of Microplastics in South Asia

South Asia is a large, unique landmass and is the southern region of Asia. It consists of eight countries—Sri Lanka, Nepal, Pakistan, Bhutan, India, Bangladesh, Maldives, and Afghanistan (Sivakumar and Stefanski 2010). Its total area covers almost 5.1 million km^2 (1.9 million mi^2), and this is 11.51% of the Asian continent. The landmass of this region gives a ground for near 1.749 billion people that covers about one-fourth of the world's population (Lanka 2000).

Here, different industries like cotton, textiles, pharmaceutical, carpets, chemical, food processing, tourism, iron and steel, and leather produce plastics that converts into MPs through different processes like weathering and highly contributes to pollution.

MPs were observed by (Balasubramaniam and Phillott 2016) in Australia, Bangladesh, Sri Lanka, Indonesia, India, Maldives, Myanmar, Pakistan, and Tanzania, but they are observed to be higher in Pakistan, Sri Lanka, India, and Bangladesh (Balasubramaniam and Phillott 2016). Its concentration has been reported to be high in sediments than in water (Robin et al. 2020; Koongolla et al.

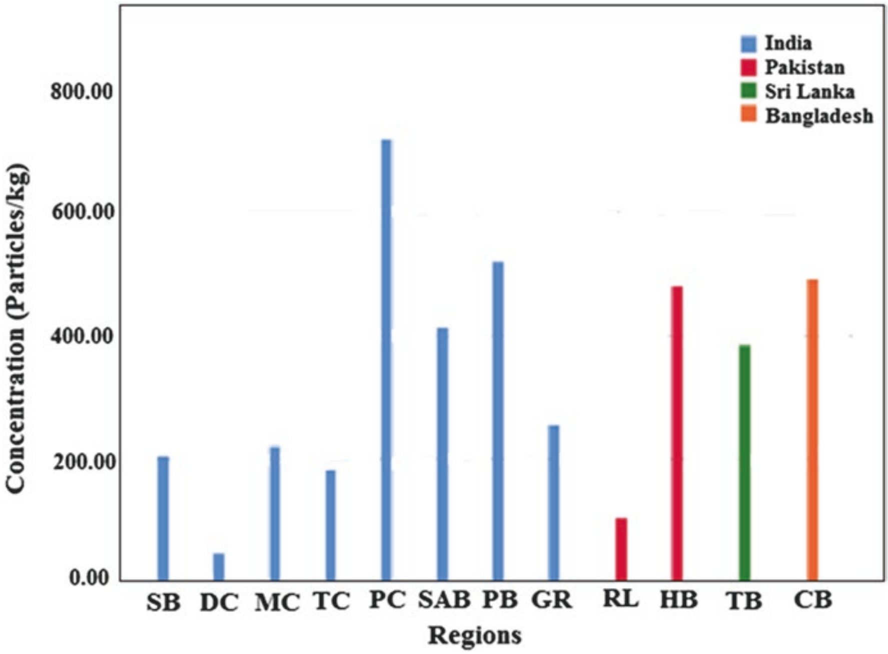

Fig. 1.3 Bar graph comparing mean concentration of MPs in sediments (Kg) across different countries. *SB* Silver Beach, *DC* Dhanushkodi coast, *MC* Mumbi coast, *TC* Tuticorin coast, *PC* Puducherry coast, *SAB* South Andaman beach, *PB* Palolem Beach, *GR* Ganga River, *RL* Rawal Lake, *HB* Hawke's Bay Beach, *TB* Thumpalai Beach, *CB* Cox's Bazar)

2018; Irfan et al. 2020a, b). This could be due to the biofouling that might sink MPs at the bottom (Irfan et al. 2020a). MPs were evaluated based on their origin from three different locations along Indian coast where the order of MPs abundance were found as Mumbai (megacity, Arabian seacoast), Tuticorin (industrial city) followed by Dhanushkodi (tourist spot, Bay of Bengal coast) (Tiwari et al. 2019). To identify the extent of flood, MPs pellets were quantified before and after the flood along the Chennai coast in India. The abundance increased up to threefold due to flood (Veerasingam et al. 2016b). In terms of size, plastic litter in the surface water were quantified off Colombo, where 99.60% of MPs were found (0.3–1 mm) among the total plastics (0.3–100 m) observed that might increase susceptibility to aquatic life (Athawuda et al. 2020). Sri Lanka ranked fifth among 192 countries that discharge plastic waste to the world's oceans (Jambeck et al. 2015). Based on region MPs were quantified along different coastal matrices in southwest coast of India, where MPs were found highly in southern coast in both beach sediments and surface water followed by central sector. While in northern sector least concentration of MPs were observed (Robin et al. 2020). Charred MPs were firstly investigated in Faafu Atoll in Maldives from beach sediments and surface seawater. Despite the remoteness and less population of the area, MPs were found in a considerable amount (Saliu et al. 2018). MPs were also quantified according to tide lines where in high tide line, MPs are observed to be high (Karthik et al. 2018). In low tide line (LTL), MPs remain whelmed during most part of the day and are transported to the oceans by the high-energy waves. This is one of the reasons why the concentration of MPs are less abundant in LTL (Karthik et al. 2018).

1.5 Types of Microplastics Based on Shape in South Asia

The abundance of fragments is due to the degradation of municipal waste (Ravi River), whereas fibers in water could originate from domestic activities (Kang et al. 2018; Wang et al. 2017). Films are found in the aquatic system because of the breakdown of plastic carry bags, lines/fibers are likely from the pieces of fishing nets or ropes. Microbeads are used in personal care products, for example, facial scrubs (Fendall and Sewell 2009).

MPs were sorted into films, foams, fragments, pellets, and fibers/lines in which film- and foam-shaped MPs were high in the studied area (Table 1.1). Films are found in the aquatic system because of the breakdown of plastic carry bags, lines/fibers are likely from the pieces of fishing nets or ropes (Sruthy and Ramasamy 2017), while fragments were observed as dominant type of MPs (Saliu et al. 2018; Dowarah and Devipriya 2019; Karthik et al. 2018; Patchaiyappan et al. 2020; Irfan et al. 2020a; Jayasiri et al. 2013) followed by fibers (Patchaiyappan et al. 2020; Irfan et al. 2020b) that are secondary in nature (Sruthy and Ramasamy 2017; Irfan et al. 2020a) as fragments were high in sediments while fibers were high in water (Irfan et al. 2020b). Fragments are formed due to the breakdown of larger plastics, while fibers originate from shipping, fishery, and textile (Xiong et al. 2018). Expanded polystyrene with fragments of larger plastic debris shape were found to be high followed by raw pellets (Imhof et al. 2017), while microfibers were observed that is secondary in nature, formed from larger MPs (Gopinath et al. 2020; Tiwari et al. 2019; Balasubramaniam and Phillott 2016). The irregular fragments were highly observed (Vidyasakar et al. 2018) that majorly correspond to fishing practices and household materials (Vidyasakar et al. 2018) with white in color comprised of PVC polymer type (Vidyasakar et al. 2020) followed by polyethylene and nylon that comprised of pellets and fibers, respectively (Vidyasakar et al. 2020). Polyethylene and polypropylene belonged to film and fiber shapes and were observed to be high in Ganga River (Sarkar et al. 2019).

1.6 Sources of Microplastics in South Asia

MP sources mostly arise from anthropogenic activities (Patchaiyappan et al. 2020), i.e., direct disposal of plastic bags and abandoned fishing nets, foams, and disposable plastic materials (Karthik et al. 2018). MPs are also transported from one area to another due to unplanned burning, incomplete burning, and lack of proper plastic management techniques (Karthik et al. 2018; Patchaiyappan et al. 2020; Imhof et al.

2017). Industrial and commercial areas are also major contributors of MP pollution (Sruthy and Ramasamy 2017). According to (Irfan et al. 2020b), the abundance of MPs is due to the degradation of municipal waste and domestic activities, for example, washing of clothes (Balasubramaniam and Phillott 2016) nevertheless of fishing activities (Karthik et al. 2018; Vidyasakar et al. 2020; Irfan et al. 2020b; Jayasiri et al. 2013) concluded that land-based sources contributed mostly to MP pollution than sea-based sources. Beaches adjacent to a river mouth have high MPs than tourism and fisheries, respectively (Karthik et al. 2018). MP concentration were high in residential and densely populated areas (Robin et al. 2020; Irfan et al. 2020a; Imhof et al. 2017; Tiwari et al. 2019), and tourist activities (Robin et al. 2020; Tiwari et al. 2019) and fishing practices majorly contributed (Dowarah and Devipriya 2019; Vidyasakar et al. 2018; Robin et al. 2020; Gopinath et al. 2020; Patchaiyappan et al. 2020; Sarkar et al. 2019) to the pollution where fragmentation occurs due to high UV radiations and high wave currents (Dowarah and Devipriya 2019). Plastic debris could be transported to remote areas by wind and ocean currents from densely populated regions (Imhof et al. 2017). It is transported over long distances from the original point (Imhof et al. 2017). Fibers may originate from carpets, discarded and weathered PP materials, fishing materials, diapers, and air filters (Balasubramaniam and Phillott 2016). The abundance of MPs depends on oceanographic conditions and originates primarily from sea-based sources according to (Veerasingam et al. 2016a) along the Goa coast in India. The winds and ocean currents also play a major role in the deposition of MPs. MPs released into the environment from both manufacturing and transport are carried out by surface runoff through streams and rivers and eventually to the oceans (Veerasingam et al. 2016a). Monsoon also plays an important role in transportation of MPs from sediments to water (Vidyasakar et al. 2020). The distribution of MPs in southeast coast of India was predominantly controlled by coastal urbanization and river inputs (Karthik et al. 2018; Vidyasakar et al. 2018). MP concentration also depends on weather conditions. High plastic pollution was reported in August due to the sea and weather conditions (Athawuda et al. 2020). A variety of charred MPs were highlighted by (Saliu et al. 2018) for the first time in this literature, which are in the proximity of an inhabited island due to combustion of waste at the shoreline (Saliu et al. 2018; Gopinath et al. 2020). MP pellets released into the environment due to manufacturing and transport are carried out by surface runoff and eventually to the ocean where wind and ocean currents affect the distribution (Veerasingam et al. 2016b). In the surface water, MP concentration is also high due to recreational usage of beaches (Koongolla et al. 2018; Imhof et al. 2017; Jayasiri et al. 2013), tidal flux (Sarkar et al. 2019), and religious and fishing activities (Jayasiri et al. 2013). Moreover, fishery harbor is the major contributor in surface water MP pollution (Koongolla et al. 2018).

The cracks and surface roughness indicate the long-term exposure to the environment that the MPs have undergone, which is through different weathering processes, i.e., mechanical and oxidative, etc. (Tiwari et al. 2019). The structure of an island is not influenced by anthropogenic activities, so due to long-term accumulation where the plastic abundance increased through on-site fragmentation that degraded by strong wind and heavy rains etc. that increased the threat to aquatic ecosystem (Imhof et al. 2017).

1.7 Microplastic Trends Based on Polymer Type in South Asia

Polymers are materials made of long, repeating chains of molecules. The materials have unique properties, depending on the type of molecules being bonded and how they are bonded. Some polymers bend and stretch, like rubber and polyester. Others are hard and tough, like epoxies and glass. Low-density polyethylene (LDPE) were observed as a dominant-type MP polymer that might be due to its excessive use in industrialized area (Sruthy and Ramasamy 2017). LDPE was also found frequently in (Imhof et al. 2013; Ballent et al. 2016; Vianello et al. 2013; Noik and Tuah 2015). FTIR analysis showed that polyethylene (PE) and polypropylene (PP) are found to be high among all other polymer types (Veerasingam et al. 2016a; Saliu et al. 2018; Veerasingam et al. 2016b; Karthik et al. 2018; Vidyasakar et al. 2018; Robin et al. 2020; Koongolla et al. 2018; Imhof et al. 2017) with white in color. The white color showed the virginity and weathering of MP pellets due to environmental factors, i.e., exposure to sun (Veerasingam et al. 2016b). Fragments comprising PE were highly found due to the disintegration and weathering of large MPs that transported over large distances in water bodies (Robin et al. 2020). PE and PP float on the surface water due to their low densities (Zhang et al. 2018), whereas PVC, PET, and polyvinyl alcohol are deposited as a sink in sediments due to their high densities (Vidyasakar et al. 2020). Meso- and microplastics were quantified in the sediments of Ganga River in India where polyethylene terephthalate was found to be abundant among the polymer types followed by PE (Sarkar et al. 2019). Thirteen polymer types have been identified, viz., polypropylene, poly(dimer acid-co-alkyl polyamine), melamine, polyvinyl chloride, polyvinyl formal, polybutadiene, polysulfide, poly(butadiene-acrylonitrile acrylic acid), poly(per fluoroethylene oxide), nylon-6, polyvinyl-benzoate, epoxy epichlorohydrin, and acrylonitrile butadiene styrene (Patchaiyappan et al. 2020). PE polymer types were highly found from the samples from beach sands which indicates the excess use of food packaging materials especially bottled water (Tiwari et al. 2019). All polymer types, i.e., PE, PP, PET, PS, nylon, PVC, etc., have different capabilities to adsorb chemical pollutants, in which PE sorbs higher concentration of pollutant than all other types (Irfan et al. 2020a).

1.8 Research Gaps Found in South Asia

The association between MPs and various other pollutants (polychlorinated biphenyls (PCBs), persistent organic pollutants (POPs), heavy metal, etc.) can be understood through extensive sampling and research and their adverse ecotoxicological impacts on the food web along the coastal ecosystems. Detailed investigations to assess the effect of MPs on sensitive organisms and coral bodies would help future research studies (Karthik et al. 2018; Vidyasakar et al. 2018; Vidyasakar et al. 2020; Irfan et al. 2020a).

A study on the improvement of negative effects, which are the result of anthropogenic pollutants, i.e., MPs on aquatic life, is also required because this issue is associated with the use of plastics and its disposal that can be resolved through 3Rs concept and proper management. The distribution of MPs is reduced when controlled at source because after release into the environment, it is hard to manage. Further seasonal fluctuation effects on the distribution of MPs are also of keen importance. Awareness among public and relevant policies can help to mitigate this problem (Irfan et al. 2020a).

MPs that humans take through either water or food and that may accumulate in the body that adversely damage the organs and lead to mortality are not under much consideration (Karthik et al. 2018).

1.9 Possible Research Solution

Controlling MPs/plastics at the source is the option to be explored seriously because once MPs are released into the environment, there is much little that can be done to limit their distribution and impacts. Concerted efforts in improving and monitoring waste management programs, emphasizing on the three “Rs” principle (reduce, reuse, and recycle) for the plastic management, may reduce the influx of plastics/MPs in the lake (Sruthy and Ramasamy 2017).

The single-day sampling does not quantify the abundance accurately; therefore, multiple replicates from multiple sites after equal interval of time would help to precise the data regarding water sampling (Saliu et al. 2018).

Most importantly, public awareness and public motivation to use biodegradable bags and nonplastic materials through government approach and nongovernment organizations should be introduced, and the strict law should be amended if there is any violation of it by anybody (Laskar and Kumar 2019).

1.10 Research Needs Regarding MP Quantification

The practical research regarding MP identification and quantification improves through amalgamation of different techniques and protocols. Certain protocols have specific limitations that can be filled by another. The handling of instrument affects the research as well, so proper practice on blank samples should be done to improve handlings. Certain clothes like cotton or nylon type should be avoided to reduce contamination.

1.11 Conclusion

Our results indicated the presence of MP in water and sediments that will lead to further study of MP presence in biota and MP pollution in freshwater systems. The present study concluded the fate, distribution, and sources of MPs on the beach sediments and fresh water in South Asia and their possible health hazard to life. The major studies on MPs in South Asia on beach sediments and fresh water are carried out in India, Sri Lanka, Bangladesh, Pakistan, and Maldives. The highest concentration of MPs was found in Pakistan, and the least was found in Maldives, while Bhutan and Nepal lack information regarding freshwater and beach sediments. Among shapes, fragmented MPs were highly observed followed by fibers. Other shapes like foams, films, pellets, and filaments were also observed. Polyethylene polymer type was dominant followed by polypropylene and other polymers like polyvinyl chloride and polyethylene terephthalate. Research on association of MPs with other pollutants will help to give awareness to the people regarding severity of MP pollution and might reduce anthropogenic contamination. The development of strong methodology and amalgamation with all other authentic techniques lead to accuracy and precision. MPs pose serious threat to aquatic and terrestrial life through consumption. It enters the food chain of organisms due to its resemblance with food particles and biomagnifies from lower trophic level to higher trophic level. Further reviews also are to be published related to MPs and their association with other contaminants to improve waste management techniques and local bodies on a small scale.

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Chapter 2
Extraction, Enumeration, and Identification Methods for Monitoring Microplastics in the Aquatic Environment

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Abstract Plastic products in different sizes and with different features are in widespread use to make day-to-day life easier. Until now, these materials have not been considered as a threat to terrestrial and aquatic environments; however, the accumulation of these products in these environments is now considered to cause plastic pollution. Consequently, plastic wastes and their degradation byproducts have become a more serious problem to overcome. The level of anthropogenic pollution defines as the level of plastic pollution. Efforts encompassing all known methods for plastic pollution mitigation have been attempted, but as yet, an effective solution has not been found. In the last 40 years, MPs have gained a significant attention as one of the emerging pollutants in the aquatic environment. The expansion of the occurrence of MPs in fresh water and seawater globally results in having a great deal of attention by scientists, policymakers, and the public. The determination and comparison of MPs abundance and characteristics are still not understood well as MPs study is in its early years. The studies on MPs' spatial and temporal variations in aquatic systems should be increased worldwide. In this chapter of the book, principal methods in microplastic isolation such as sampling, separation, and identification have been attempted to comply with microplastic research to be performed in various matrices of freshwater and marine ecosystems. Also, new techniques and methodologies that are preferred for use in microplastic studies will be explained.

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2.1 Introduction

Studies on plastic pollution in ecosystems focus mostly on marine environments rather than terrestrial and freshwater ecosystems due to the demand for determining the effects of the microplastics (MPs), that is, the degradation of products of plastics, on the aquatic ecosystems. Microplastics have attracted attention due to their occurrence in the aquatic environment, their persistence, and their possible ecological risk of ecosystems (Yukioka et al. 2020; Yang et al. 2021). Plastic fibers, films, and particles smaller than 5 mm (primary and secondary microplastics) refer to microplastic. However, the definition of MPs by researchers has not been consistent over the years. In 2014, they were often referred to as "small plastic particles" if they had a size of less than 5 mm. Then more recently, the definition was "small plastics" with a size of less than 1 mm in the studies from 2011 to 2014 (Browne et al. 2011; Andrady 2011; Lambert et al. 2014). Small particles with a size of less than 5 mm can be consumed by organisms because of their size, and they are still regarded as "*microplastics*" by some researchers. As long as the scope of studies continues to evolve from MPs with the same size as planktonic forms to "*nanoplastics* (NaPs)" that can be consumed by planktonic forms and thus easily added to the food web, the need for new approaches to define the particles with smaller than micro-size will continue to increase (Lusher et al. 2020).

Owing to the lack of a standardized method for the detection of microplastics in aquatic environments, defining the method and road map for the purpose of the study plays important role in carrying out the study. Optimized methods are required to acquire comparable tracking data to the results conducted by various research groups in different locations (Ryan et al. 2009; Lusher et al. 2020; Lv et al. 2021). There are many published protocols regarding the detection of MPs, which have

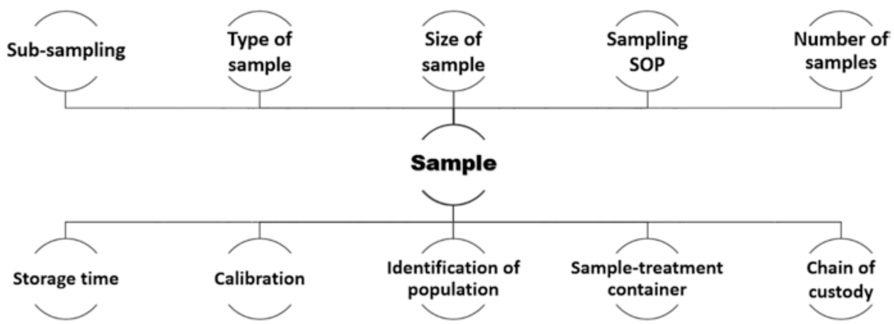

Fig. 2.1 MPs sampling fundamental modules in aquatic systems. (SOP stands for Standard Operation Procedure) (Adapted from Madrid and Zayas 2007; Yang et al. 2021)

already been used by the researchers (Bessa et al. 2019a; European Commission 2013; Gago et al. 2019; Kovač Viršek et al. 2016; Liu et al. 2018a); however, these protocols have different properties among them. The fundamental modules carried out in a sample study of MPs were summarized in Fig. 2.1 by examining the studies on MPs. It is proposed that a work plan should be prepared accordingly considering related fundamental steps which are (1) how, when, and where to gather samples; (2) maintained and calibrated water sampling equipment; (3) sample containers, their cleaning and stabilizers addition and storage, and sample treatment procedures (e.g., drying, mixing, and handling before measurements); (4) subsampling procedures; and (5) sample recording (e.g., labeling, auxiliary information, and chain of custody requirements).

In this chapter of the book, fundamental methods and techniques in microplastic isolation such as sampling, separation, and identification have been attempted to comply with microplastic research to be performed in various matrices of aquatic ecosystems (river, lake, estuary, lagoon, and sea) within the framework. Furthermore, new methodologies that are preferred for use in studies will be provided. For this purpose, the topics of the proposed methods to monitor the detection of MPs in aquatic ecosystems are grouped as (i) MPs sampling (water surface, water column, and sediment) and fundamental considerations to be taken into account through sampling (sampling strategy, contamination), (ii) separation of MPs from samples, and (iii) identification, chemical characterization, and quantification. These groups are given as main headings by considering particular considerations of aquatic ecosystems and are divided into subheadings and figures in detail.

2.2 Data Collection

2.2.1 Selection of Sampling Areas and Main Parameters During the Process

The necessary standards for sampling, pretreatment, quantification, and identification have not been regulated even though several studies on MPs have been conducted in the last 20 years (Silva et al. 2018; Prata et al. 2019). However, studies are performed with the hypothesis that "every environment having MPs actually refers to matrices where the isolation of MPs can be realized and examined." The determination of the abundance, the morphological properties (e.g., type, color, and size) of MPs in different environmental matrices, and the polymer types were investigated (Chanpiwat and Damrongsiri 2021; Fan et al. 2021; Çullu et al. 2021; Huang et al. 2021; Hurley et al. 2018; Napper et al. 2021). In studies on flora and fauna, the detection and determination of their effects on organisms have gained attention nowadays (Wu et al. 2019; Sadler et al. 2019; Kukkola et al. 2021). However, studies on aquatic ecosystems are considered special studies that require high effort due to the high cost of research as well as the difficulties arising from the environmental parameters in the area. Each aquatic area has its own unique environmental characteristics and physical, chemical, and biological changes that play a special role in the ecosystem. These processes and environmental factors are frequently encountered in the detection and characterization of MPs studies. For example, since physical changes in aquatic areas will make a difference in the results obtained from sampling, understanding these changes and predicting their behaviors can be obtained by instantaneous data (e.g., satellites, autonomous vehicles). Another example is that excessive algae growth (especially bloom effect), which is observed in the spring months in water with high trophic levels, has a significant effect on the transport of MPs in the water column and their isolation from environment. It is found that the abundance of MPs detected in the water surface or column may not usually be directly proportional to the abundance of MPs that the biota is exposed to and can digest. The heterogeneous distribution of MPs in the water surface and column is also one of the factors affecting this situation. The studies are created based on the factors such as the extent of exposure of the organism to MPs in aquatic environment, the trophic level of the organism (primary consumers and secondary consumers) and its related characteristics, and the size of the MPs that the organism digests. In addition, according to the isolation and identification of MPs from aquatic environments, the methods in the identification of MPs in biota have a more specific road map. For this reason, the field of study should be firstly analyzed and determined, and then research strategies should be developed in accordance with the aim of the MPs-related study.

The selection of the sampling method and equipment, as well as the definition of the study area, is a major factor affecting the property, amount, and characterization of MPs (Barrows et al. 2017; Conkle et al. 2018; Hung et al. 2021; Lares et al. 2019; Miller et al. 2021; Miller et al. 2017). It is possible to determine the vertical and horizontal distribution of MPs with the sampling method that is selected in accordance with the aim of the study. However, for horizontal or vertical sampling, the field properties should be suitable as well as the method and equipment. For example, in coastal sampling, the selection of the sampling site and equipment is as important as defining the point source of pollutants in the region. Sampling results at stations that are close to the undefined source of pollution in the area may be misleading as the distribution of MPs in the water surface and column is not homogeneous. In particular, the analysis results obtained from this study to determine the environmental concentrations of biological pollutants can only be representative for that moment. Therefore, the standard deviation of the results of simultaneous parallel sampling can be found to be high. The result of microplastic studies performed in aquatic environments also represents that moment and is specific to that area. Therefore, the results obtained from a second sampling that was performed even by the same researcher could be different. Expressing the results obtained from parallel sampling of this pollutant, which varies depending on time and location, not with the average, but with the lower and upper values, will increase the accuracy in the interpretation of the results. Thus, the time-dependent microplastic load/profile of the relevant study area can be determined with accurate data.

2.2.2 Sampling for Microplastics

Microplastic sampling in aquatic environments is the main and first step for the studies to be carried out in the determination of size and abundance of MPs and morphological properties and polymer types. Different, incomplete, and incorrect implementation to be performed in this step could result in misinterpreting the data in the next steps.

Sampling methods in all-natural aquatic environments (river, lake, estuary, lagoon, sea, etc.) where MPs studies be performed will be changed depending on many factors including superficial water types, water column or the structure of sediment, the properties of aquatic ecosystems, and the aim of the study. Therefore, the researcher should determine the sampling area and stations in line with the aim of the study. The sampling location, depths, and the distance from the center of human actions are the factors that can change MPs concentration (Yang et al. 2021). Here, this part aims to give the basic issues to be considered in microplastic sampling in surface water, water column, and sediment of seawater and freshwater ecosystems during sampling.

2.2.3 Contamination

Contamination is one of the important factors in MPs studies. Regardless of the method to be preferred and the area to be sampled in MPs sampling, in every step of the studies, care for the contamination of MPs and prevention of contamination should be taken. Considering the data in the literature, contamination of the water samples with synthetic fibers from clothing or atmospheric fallout is a common problem (Wesch et al. 2017; Le Guen et al. 2020). Moreover, polymeric materials used during sampling or transportation can possess potential risks. To minimize contamination during sampling and sample preparation, the use of plastic equipment should be reduced as much as possible. It is recommended to utilize sterile glassware as used in microbiology studies. When the use of plastic equipment is mandatory, blank samples should be prepared and analyzed with this sample to determine whether they have effects on the MP abundance (Klein et al. 2015). This case is found to be more significant in fiber studies. In most of the fiber studies, the reliability of the results may be questioned as contamination factors were not included in the studies. For this reason, determining a standard method plays a significant role in minimizing contamination in all studies which is dependent on its aim. Environmental contamination dynamics, sampling variables, and the processes conducted during sample preparation should be also considered.

The studies performed on sea-based species are one of the examples to prevent contamination based on the working areas and the methods used. The polymer-chemical nets and traps used in these studies are regarded to be possible sources of contamination (Campanale et al. 2020). Therefore, a minimum contact should have been provided, and even nets and traps with polymer-chemical structure should not be preferred. Field studies could be more challenging in terms of preventing airborne contamination compared to laboratory studies. Blind sampling is one of the most preferred methods to overcome the drawbacks caused by polymer structures under sampling conditions. To reduce contamination, regardless of whether the environment is open or closed, all equipment should be washed with ethanol or acetone followed by ultrapure distilled water. Another point to be taken into consideration is that polymer-free clothing, gloves, and equipment should be preferred to be worn.

Although these conditions are provided readily in the laboratory environment compared to field sampling facilities, care should be taken to ensure the sustainability of working conditions. Performing studies with samples containing MPs transported to the laboratory environment in a fume cupboard environment or culture unit with the negative flow, selecting glassware as consumables eminently reduces the risk of contamination, therefore increasing the reliability of the results. The contamination control studies implemented are presented in Table 2.1 when identifying the number of MPs in samples (Pérez-Guevara et al. 2021).

2.2.4 Laboratory Conditions and Main Parameters

The analysis of MPs was carried out in three steps: 1) extraction, 2) purification, and 3) quantification. Laboratory conditions need to be provided suitably at all stages for MPs studies from the identification methods in the studies of the abundance and distribution of MPs to the studies for visual inspection. For example, laboratory conditions are expected to be isolated for analyses that require visual inspection (microscopic) including the identification of microplastic abundance and the definition of colors. Having suitable fume cupboards and ventilation systems plays an important role in oxidation processes. These two conditions are the very least required to determine the polymer types.

The selected methods for analysis may lead to some undesired outcomes. For example, the stainless steel sieve sets used in size analysis usually cause MPs load loss (the loss of MPs). In the case of not determining the size distribution of MPs that are obtained in various sizes as a result of wet/dry sieving, it may give rise to incorrect categorization of MPs based on their sizes. The oxidation process is one of the methods that is used in MPs isolation processes from aquatic environments. Chemical substances are utilized in the oxidation process, and these chemical substances cause color loss in MPs (weak coloration) after treatment while destroying organic materials (Yin et al. 2020).

Table 2.1 Contamination control studies in microplastic sampling (adapted from Pérez-Guevara et al. (2021))

Blanks	Contamination prevention measures	Secondary contamination from blanks	References
Not reported	– Samples were placed in paper bags for storage – Feces samples were lyophilized to prevent contaminating externally	Not reported	Gil-Delgado et al. (2017)
Not reported	– Feces samples were lyophilized to prevent contaminating externally	Not reported	Huerta Lwanga et al. (2017)
Damp filter paper Procedural blank using Milli-Q water	– Sterile centrifuge tubes were used to store samples – Sample collection tubes were rinsed with Milli-Q water – Personal protection equipment such as cotton lab coat and gloves were worn – 70% ethanol was used to clean all work surfaces – All apparatus were rinsed with Milli-Q water – Sieves and filter papers were examined using stereomicroscope	Not reported	Nelms et al. (2018)
One blank filter every ten samples	– The samples were placed in an ash aluminum foil envelope and frozen for storage	Two microfibers were observed and subtracted	Perez-Venegas et al. (2018)
Ten blank samples with tap water	– Cotton lab coat and nitrile gloves were worn – All lab wares including filters, beakers, and petri dishes were thoroughly washed and inspected for microfiber contamination	One or two per sample	Provencher et al. (2018)
Not reported	– Samples were placed in individual vials	Not reported	Reynolds and Ryan (2018)
Procedural blanks (triplicate) performed using Milli-Q water and H_2O_2	– Glass fiber filter Whatman GF/F 0.7 μm was used to filter all the solutions before use – Filters were examined to define any external contamination of MPs – Lab wares were rinsed with Milli-Q water. – Laminar flow hood equipped with 0.2 μm HEPA filter was used for analyzing – Feces samples were placed and stored in glass tube at 4 °C	Not reported	Zhao et al. (2018)

(continued)

Table 2.1 (continued)

Blanks	Contamination prevention measures	Secondary contamination from blanks	References
Three blanks using Milli-Q water	– Sterile bags were used to store samples – Feces samples were lyophilized to prevent contaminating externally – 1.2 µm filters were used to filter all solutions and reagents – All lab wares were washed using Milli-Q water – Laboratory access limited – Cotton lab coat and nitrile gloves were worn	Not reported	Bessa et al. (2019b)
Two control samples from study area Laboratory and procedural blanks	– The samples were placed in a sterile, polyethylene “Whirl-Pak” 7 oz. sample collection bags and frozen for storage – Synthetic fleece clothing was used to stop from contacting collection bags – Personal protection equipment was used such as cotton lab coat and nitrile gloves – The materials after use were explicitly washed and low-lint wipes were used to dry them	No contamination from control samples of study areaProcedural blank: 22 fibersLaboratory control (filter air): 49 fibers	Donohue et al. (2019)
Not reported	– The samples were placed in sterile plastic collection bags or aluminum foil and frozen for storage – To prevent cross-contamination, the surfaces were cleaned prior to collecting each sample – Hot water with soap was used to clean sieves, and the cleaned sieves were then dried and checked visually if no material left	Not reported	Hudak and Sette (2019)
Not reported	– Samples were placed in glass bottler for storage	Not reported	Masiá et al. (2019)
Damp filter paperProcedural blank using Milli-Q water	– Sterile centrifuge tubes were used to store samples – Sample collection tubes were rinsed with Milli-Q water – Personal protection equipment such as cotton lab coat and gloves were worn – 70% ethanol was used to clean all work surfaces – All apparatus were rinsed with Milli-Q water – Sieves and filter papers were examined using stereomicroscope	Not reported	Nelms et al. (2019)

Blanks	Contamination prevention measures	Secondary contamination from blanks	References
Laboratory: One procedural blank (prefiltered Milli-Q water; 0.22 μm) without stool sample	– Lab wares were washed with Milli-Q water – Antibacterial solution Aqua Kem Blue was used – 50 μm metal sieve was used to filter all solutions before use	No contamination in blanks	Schwabl et al. (2019)
Laboratory: Procedural blanks for every 15–20 samples	– Feces samples were lyophilized to prevent contaminating externally – Round-bottom flasks were placed in furnace at 450 °C for 12 h and washed with acetone and methanol before use – Surface layer of feces was removed prior to segregating MPs	Not reported	Zhang et al. (2019b)
One procedural blank for every 10 samples	– The samples were placed in an ash aluminum foil envelope and frozen for storage – Cotton lab coat was worn – All glasswares were rinsed with Milli-Q water	Three blue fibers	Garcia-Garin et al. (2020)
Seventeen procedural blanks using Milli-Q water	– The metal spatula was cleaned with prefiltered ethanol after collecting each sample – Feces samples were placed in 2 mL Eppendorf tubes containing prefiltered 80% ethanol and frozen for storage – Clean glass microfiber was open next to the samples during the whole processes – Cotton lab coat was worn – All lab wares were cleaned with Milli-Q water	59 microfibers were found in 17 procedural blanks and subtracted.	Le Guen et al. (2020)
Damp filter paper and procedural blanks	– Feces samples were lyophilized to prevent contaminating externally – 1.2 μm filters were used to filter all solutions and reagents – To keep airflow under control, HEPA filter was used – Tyvek suits, non-shedding rubber shoes, and nitrile gloves were worn – Lab wares and glasswares were washed three times with prefiltered distilled water	Not reported	Moore et al. (2020)

(continued)

Table 2.1 (continued)

Blanks	Contamination prevention measures	Secondary contamination from blanks	References
One blank filter every ten samples	– The samples were placed in an ash aluminum foil envelope and frozen for storage	Not reported	Perez-Venegas et al. (2020)
Recovery rates detection using PE (150 µm), PS (250 µm), and PVC (75 µm)	– Samples were placed in 1 and 2 L amber glass container for storage – Feces samples were lyophilized to prevent contaminating externally – Cotton lab coat and nitrile gloves were worn – All apparatus were cleaned three times with Milli-Q water – Raman spectra was used to examine filters before use	Not reported	Yan et al. (2020)

2.3 Microplastics Sampling Methods in Aquatic Ecosystems

Microplastic sampling studies in aquatic environments have recently attracted greater attention within the scientific community. These studies were generally performed in surface water, water column, and sediment. The abundance of MPs has been found less in pelagic (surface water and water column) as compared to benthic (sediment). Therefore, the volume of water for sampling should be high in order to obtain enough data for the comparison of surface water with sediment. In sediment sampling, the amount of sample collected often varies in the range of 25–50 mL in volume (Van Cauwenberghe et al. 2013). The results obtained in water sampling can be affected by many variables. For the same sampling point and sampling method, the length of the water column and the current (moving-still water surface) are the most significant criteria. In most studies, it has been determined that the abundance and polymer structures of MPs vary in samples taken from the water surface and water column. In the studies having the same polymer structure, it was mentioned that the various size ranges were determined in different water column depths for the same polymer type (Güven et al. 2017).

In this part of the chapter, MPs sampling in the surface of water, water column, and sediment was presented under separate headings.

2.3.1 Sampling Methods in Water Surface and Water Column

Many of the MPs that are detected in nature have tendency to float due to their density. This physical property allows MPs to be transported to long distances from their source by physical, chemical, or biological vectors (Gago et al. 2019). However, the biofilms that occurred on the surface of MPs or their interactions with suspended solids in the aquatic environment can cause MPs to collapse (Kaiser et al. 2017). Moreover, it is stated that MPs and NaPs, which are regarded as new participants of the water column, can have an adverse effect on photosynthesis by competing with phytoplankton (Aytan et al. 2020). MPs with different densities can potentially be transported through the water column. Giving all these properties, selecting an appropriate sampling method for determining the distribution of MPs on the horizontal or vertical line becomes important. For this purpose, trawl (neuston, manta, plankton, etc.) and bulk water (grab or pump) methods are often chosen for microplastic sampling in seawater (Hung et al. 2021) (Fig. 2.2).

Trawl technology is one of the main methods of water sampling for MPs. Neuston net and manta trawl are selected for surface water sampling, whereas plankton net is preferred for water column sampling (Barrows et al. 2017; Eriksen et al. 2018; Tamminga et al. 2018; Aytan et al. 2020; Tokai et al. 2021). These are expensive equipment, and a marine vehicle (boat, ship, etc.) is also required for these sampling methods. For an accurate MP sampling, mounting this equipment in the most appropriate part of the vehicle by making the necessary calculations is necessary. The

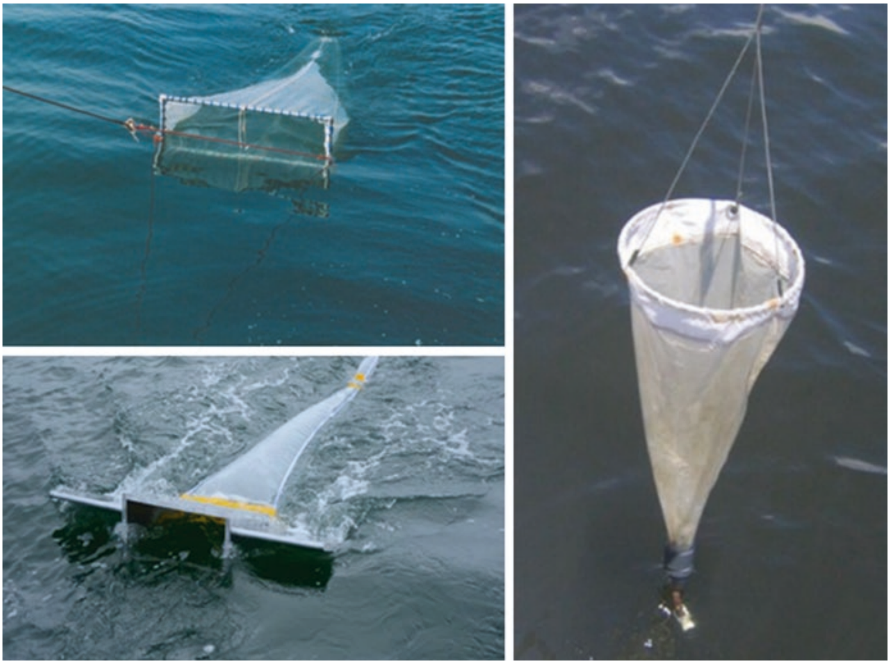

Fig. 2.2 Some photos of the nets used in MPs sampling. (Left top: Neuston net (Neuston 2021), left bottom: manta net (Manta 2021), right: plankton net (Plankton 2021))

location of the trawl on the ship should be positioned correctly in order to not give rise to an incorrect sampling and contamination from the ship (especially from the paint on the exterior of the ships). The maneuvers of the ship should be considered to be compatible with sampling of the trawl and in the opposite direction of the wave. Thus, the samples containing MPs, where they are collected from the surface water, are prevented from being re-discharged to the environment with fluctuating movements and reverse maneuvers.

Sample collection time varies depending on mesh size, sea traffic, weather conditions (wind direction/intensity), and sampling period. The vehicle user is required to be experienced in accounting for the parameters that can affect the success of sampling in determining the station location with GIS and transportation. Parameters such as the measurement of flow velocity with a flowmeter have an effect on the abundance of MPs (Gago et al. 2019). The MPs in the net can be likely to be reintroduced to the environment due to unfavorable atmospheric and hydrographic conditions. Therefore, meteorological parameters such as wind strength and direction should be accounted for on the sampling day. Furthermore, after the sampling is completed, any possible contamination from the ship's deck and the people present in the backwashing of the nets should be accounted for.

High biological activity in the sea related to the trophic situation of the marine environment studied or the increase in primary production corresponding to the

Table 2.2 MPs detection methods in seawater (adapted from Zheng et al. (2021))

Sampling tools	Depth	Volume	Location	Reference
250 μm sieve	3 m	2 m^3	The Northeast Atlantic Ocean	Lusher et al. (2014)
500 μm WP-2 net	50 m	NR	The Greenland Sea	Amélineau et al. (2016)
250 μm stainless steel sieve	11 m	2 m^3	The Atlantic Ocean	Kanhai et al. (2017)
335 μm mesh net	20 cm	NR	The Bay of Brest	Frère et al. (2017)
333 μm nylon plankton net	200 m	110 m^3	The South China Sea	Cai et al. (2018)
30 μm steel sieve	30 cm	0.025 m^3	The North Yellow Sea	Zhu et al. (2018)
335 μm mesh manta trawl	Surface	NR	Kingston Harbor	Rose and Webber (2019)
60 μm stainless steel sieve	30 cm	0.1 m^3	East China Sea	Zhao et al. (2019)
150 μm trawl net	Surface	110–148 m^3	Jiaozhou Bay	Zheng et al. (2021)
20 μm sieve	Surface	0.2 m^3	Jiaozhou Bay	Zheng et al. (2021)

sampling period can cause rapid blockage of the pores of nets. Pore sizes vary between 50 and 3000 μm, and pores having 300 or 333 μm are sourced easily and are preferred for microplastic sampling (Hidalgo-Ruz et al. 2012; Gago et al. 2019). However, it is known that the microplastic abundance distribution is related to the pore size (pore size and above) (Setälä et al. 2016; Lv et al. 2021). Apart from size, there is also a limitation with regard to the morphological properties of MPs. Hung et al. (2021) reported that fewer MPs with fiber characteristics are detected with manta net compared to other sampling methods. However, the abundance of MPs in the range of 125–355 μm with fiber properties is found more in the wastewater treatment plant effluent discharged to the marine environment (Sutton et al. 2016; Estahbanati and Fahrenfeld 2016; Mason et al. 2016). Accordingly, this method is found to be ineffective for microplastic research at the points where the wastewater treatment plant effluent is discharged into the marine environment.

Water column sampling is mostly conducted by filtration after bulk sampling. Bulk water (filtered in situ, i.e., pumped) is one of the most commonly preferred methods which is working in the principle of collecting water with a grab and a pump. Therefore, the filtering system grab sampling method is small enough to be digested by planktonic organisms, and various MPs/NaPs can be detected in a larger abundance in terms of their morphological properties (Barrows et al. 2017; Lv et al. 2021). It can be defined as the sampling method by immersing the sampling container (amber glass 1 L bottles are usually preferred) into the water by hand. By using this method, collecting samples from the water surface becomes possible. However, the biggest disadvantage of this method pertains to the sampling volume. While grab sampling can be performed with 1–2 L volume, this value reaches an average of 10 L with the pump method. In the case of current where the water is moving, the reliability of the results may be improved by increasing the number of sampling points. Furthermore, increasing the number of stations (sampling points) contributes to the determination of the impact of important factors including demographic differences around the water source, treatment facilities, industrial areas, and power plants on microplastic pollution (Alam et al. 2019; Conley et al. 2019; Deng et al. 2020; Khoironi et al. 2020; Murphy et al. 2016; Nel et al. 2017).

MPs samplings studies in the marine ecosystems with the sampling tools discussed are given in Table 2.2.

2.3.2 *Sampling Methods in Sediment*

Aquatic sediments can be divided into two classes in terms of their physical and chemical properties which are sand/sandy sediments and bed sediments. Sand/sandy sediments are composed of large particles that contain a high proportion of inorganic compounds (i.e., silicates) and a low proportion of organic compounds, whereas bed sediments contain high proportions of organic matter having smaller particles (Rivoira et al. 2020). The method to be implemented in microplastic isolation is demonstrated by this significant difference between the organic and inorganic content of both sediment samples.

In contemporary studies, the higher number of MPs in sediments compared to MPs suspended in water columns or floating on water surface could be attributed to the accumulation of MPs in the sediment that results from hydrodynamic movements and other transportation. Therefore, reporting the abundance of MPs in sediments and performing transport models become important.

In many studies, direct sampling from the beach (from the coast) is preferred owing to ease of sampling (Corcoran et al. 2009; Abu-Hilal and Al-Najjar 2009; Van et al. 2012). The abundance of MPs varies based on the depth and distance, as well as the sample collecting point. It has been found that different sampling depths (5 cm, 25 cm from the surface, etc.) have been reported in different studies (Hidalgo-Ruz et al. 2012). Although MSDF recommends sampling depth to be 5 cm, MPs have been found in sediment samples at depths of 25 cm, and 50% of MPs were found to be in the upper layer (5 cm). Also, in some studies, it is stated that the abundance of microplastic in the sample collected at a depth of 1–5 cm from the surface is expected to be higher than the abundance of microplastic at depth of 2–10 cm (Yang et al. 2021). Although some studies have reported the results where samples were collected at a depth of 2 cm, in general, studies reported the results where samples were taken through a depth of 10 cm (Zbyszewski et al. 2014; Castañeda et al. 2014). The manual and device sampling techniques for sediment sampling are given in Table 2.3.

Sediment samples can be collected from various distances to the shoreline by selecting them perpendicular, parallel, or random. Regardless of selecting a sampling method or matrix in accordance with the aim of the study, it is important to acquire information about changes in the area for a long period (at least 2 years

Table 2.3

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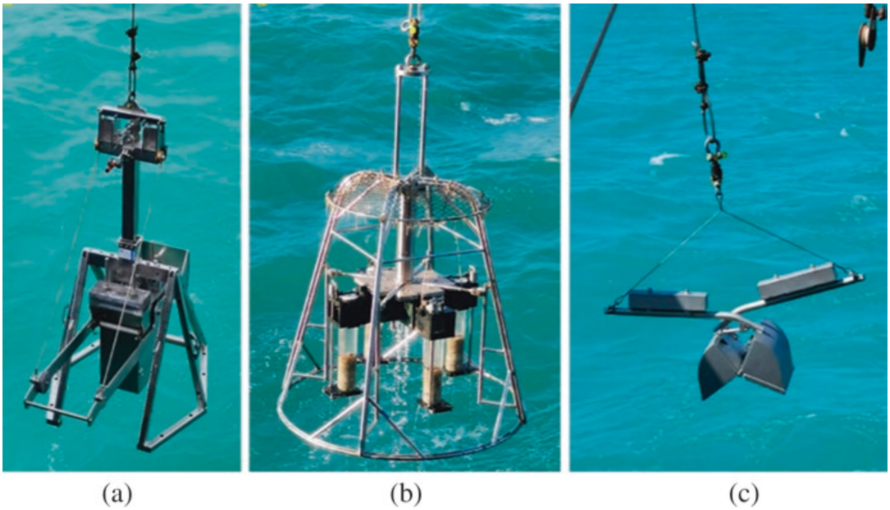


Fig. 2.3 Sediment sampling instrument: (**a**) box corer, (**b**) multi-corer, and (**c**) Van Veen grab. (Reprinted from Campanale et al. (2020)

period/dry-wet seasons) in order to determine the abundance and characterization of MPs. In particular, parameters such as rainy season, possible domestic and industrial discharges, wind and current velocities, and biological activity affect the MPs in the aquatic environment or the transport of MPs to the aquatic environment as well as causing additional microplastic loads to the environment (Horton et al. 2017). However, the use of a standard sample collector has become a great benefit in comparing samples (Castañeda et al. 2014).

The comparability of the obtained results is demonstrated by using the similarity of the sampling tool used in the studies even though a standard method for sediment sampling is not established. The sampling tool directly affects the sample unit. Box corer, multi-corer, and Van Veen grab sampling instruments are frequently used in these samplings. Figure 2.3a–c displays the most commonly used sediment sampling instruments. Almost half of the sediment sampling studies utilize an area sampling unit, while the rest of those use mass and volumetric units. While MSDF recommends volumetric sampling (European Commission 2013), NOAA suggests that the microplastic weight of volumetric sampling should be added to the results. Therefore, it has been reported that the quality and reliability of the data will increase.

2.4 Separation of Microplastics from Samples

2.4.1 Sieving

The size of MPs detected in aquatic environments is demonstrated by the pore size of the sieve (Lv et al. 2021). The use of a stainless steel sieve system facilitates the visual identification process in surface waters having high concentrations of organic/inorganic compounds. Therefore, filter papers having less microplastic load can be easily identified by using the binocular/stereomicroscope while categorizing MPs based on their sizes. This process enables numerical data to be obtained regarding the abundance and distribution of MPs on the filter paper.

Stainless steel sieve sets in different mesh sizes, depending on the aim of the study, are used to determine the minimum size while the maximum size limit is selected as 5 mm in accordance with the definition of MPs. Both dry and wet sieving can be carried out by considering the environmental matrix. However, a long rinsing period with distilled water is required to be able to sieve the particles explicitly in wet sieving. The sieves should be cleaned with distilled water and dried in the oven after each sample and/or at the end of the process.

The sufficiency of backwashing in sieving systems is down to the judgment of the investigator. The MPs that remained in the sieves increase the possibility of having incorrect results when determining the abundance and distribution of MPs. Approximately 20% to 35% of microplastic abundance is estimated to be lost at this stage. As a result of the process, the accuracy and reliability of the dimensional analysis should be investigated.

2.4.2 Elutriation

Elutriation column use is one of the effective methods that can be utilized in MPs separation based on the difference in MPs density in sediment samples of all aquatic systems (Fig. 2.4). Even though this method can be utilized for various purposes, it is nevertheless preferred to be utilized in detecting MPs (Zhu 2015; Bellasi et al. 2021). The column can separate particles depending on their density, size, and shape. This method is considered inexpensive and allows the sediment in high volumes to elutriate. However, some studies showed that the process has been optimized to enhance the microplastic extraction. Generally, parameters that can affect the efficiency of the system are reported as the size of MPs, the size and density of sediment, wash column temperature, and injection speed (Kedzierski et al. 2016).

In this system, a closed-circuit water system is utilized to prevent excessive water consumption for the separation of particles. This system contains storage, filtration, injection and flow, washing column, and temperature control (Claessens et al. 2013; Kedzierski et al. 2016; Bellasi et al. 2021). The mixture of sediment-litter is added to the system along with water from the bottom of the system. During this process, the lightest particles float upward, while the heavier particles deposit at the bottom.

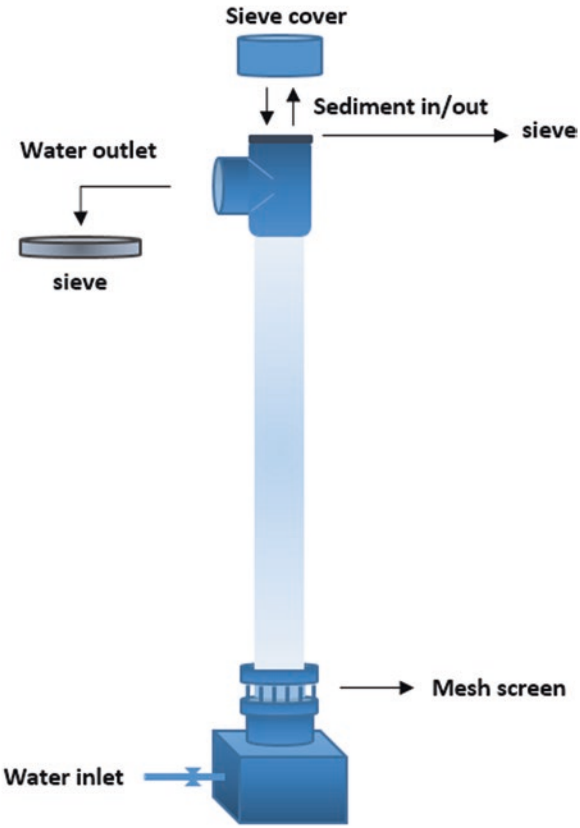


Fig. 2.4 Schematic representation of elutriation column. (Adapted from Solomon and Palanisami (2016))

Table 2.4 The buoyancy of common polymers (adapted from Gago et al. (2019))

Polymer	Abbreviation	Density (g cm^{-3})	Buoyancy
Polystyrene	PS	0.01 – 1.06	Positive (↑)
Polypropylene	PP	0.85 – 0.92	
Low-density Polyethylene	LDPE	0.89 – 0.93	
Ethylene vinyl acetate	EVA	0.93 – 0.95	
High-density polyethylene	HDPE	0.93 – 0.98	
SEAWATER		1.025	
Polyamide	PA	1.12 – 1.15	Negative (↓)
Nylon	PA 6,6	1.13 – 1.15	
Poly methyl methacrylate	PMMA	1.16 – 1.20	
Polycarbonate	PC	1.20 – 1.22	
Polyurethane	PU	1.20 – 1.26	
Polyethylene terephthalate	PET	1.38 – 1.41	
Polyvinyl chloride	PVC	1.38 – 1.41	
Polytetrafluoroethylene	PTFE	2.10 – 2.30	

2.5 Density Separation

2.5.1 Flotation

Polymers can be detected easily in environmental matrices since the density of most polymers is smaller than the density of water (Table 2.4). However, the additives in MPs can change their densities regardless of their polymer structures. Microplastics, particularly in environmental matrices, can absorb different pollutants and/or gain density from their polymers depending on the biofilm layers that are covering them (Tu et al. 2020; Fu et al. 2021). This can be used to facilitate the differentiation of MPs in environmental matrices from organic matter during their isolation.

In studies aiming to analyze MPs having a size of 500 μm and above in aquatic ecosystems, size analysis (with stainless steel sieve sets) is recommended to be performed before the density process. MPs density extraction can be very efficient, particularly in environmental samples since polymers are less dense than soil, sand, silt, etc.

The processes are based on the principle of mixing the sample with a saturated salt solution with a known density and then separating the MPs after a certain retention time. After the retention time, the solution containing the MPs in the supernatant of the separating funnel is separated by first siphoning and then filtration. The precipitation times of samples in saturated salt solutions have been found to vary. Time ranges from shorter times of 5 min (Imhof et al. 2012) to 24/48 h have been reported (Fries et al. 2013).

Sodium chloride (NaCl) is frequently used in the density process since it is inexpensive, readily available, and environmentally friendly (Lusher et al. 2020; Lv et al. 2021). However, the same efficiency may not be acquired from the salt solution prepared with NaCl for each polymer type (Table 2.5). For example, PET and

Table 2.5 Usually used solutions in polymer separation

Solutions	Density (g cm^{-3})	Polymer types	References
NaCl	1.2	PE, PP, PA, PS, PC, PMMA, ABS	Fries et al. (2013), Liu et al. (2018b)
NaI	1.6	PP, PE, PVC, PET, PVA, POM, PS, PU, PMA, alkyd, acrylic, polyester	Claessens et al. (2013)
$ZnBr_2$	1.7	PE, PP, PA, PS, PVC, PET, PU, PVA, PMA, acrylic, polyester, POM, alkyd	Crichton et al. (2017), Hidalgo-Ruz et al. (2012), Quinn et al. (2017)
$ZnCl_2$	1.5–1.7	PE, HDPE, PP, PC, PA, PS, PVC, PET, nylon	Lusher et al. (2020), Liebezeit and Dubaish (2012), Imhof et al. (2012)
$CaCl_2$	1.3–1.5	PE, PP, PS, PET, PVC, PC, PA, PU, ABS	Scheurer and Bigalke (2018)
$Na_6(H_2W_{12}O_{40})$	1.4	PVC, PET, nylon	Corcoran et al. (2020)

PVC are denser than NaCl, and instead of NaCl, sodium iodide (NaI), sodium bromide (NaBr), and zinc chloride ($ZnCl_2$) can be used to separate a greater range of polymers. On the other hand, some special solutions such as $ZnCl_2$ can cause problems in terms of ecotoxicology and occupational health and safety. Sodium polytungstate and NaI are regarded relatively expensive chemicals. The minimum level of waste generation can be supported, and the cost can also be reduced by using the recovery and reuse of these salts. Moreover, when working with various salt solutions, the chemical structure of the salt should be well known, and its reactivity should be investigated. For example, by reacting Na (I) with cellulose, converting cellulose to black color can result in having visual identification hindered (Crichton et al. 2017; Lusher et al. 2020; Lv et al. 2021). Salt solutions used in different studies are represented in Table 2.5.

2.5.2 Removal of Organic Matter

Identification of MPs is often hindered by natural remains accompanying the particles, located at the point where the sample is sourced (Mbachu et al. 2021). Therefore, organic compounds in the environment need to be removed to prevent incorrect or incomplete identification. Chemical or enzymatic methods are generally used in this process. However, organic removal by enzymatic methods is regarded as process that is not yet fully efficient. Moreover, enzymatic methods are not preferred because of the long retention times (approximately 13 days) and the environmental conditions (temperature, pH, etc.) that affect the yield (Courtene-Jones et al. 2017; Liu et al. 2020; Mbachu et al. 2021). In addition, many of these enzymes used in molecular biology are known as expensive materials. In several studies that are using cheaper enzymes, they reported insufficient results obtained for different sample matrices and desired imaging conditions (Löder et al. 2017). Therefore, in this section, information and data about chemical methods that provide faster and higher accuracy in three subheadings are included.

2.5.2.1 Acid and Alkaline Digestion

Various acid/alkaline solutions can be used to isolate MPs from the environment. The most common used solutions and summary of the polymer interactions are presented in Table 2.6. The explanations of A, B, C, and D letters in Table 2.6 are given under the table. The most common solutions are reported to be hydrochloric acid (HCl), hydrogen peroxide (H_2O_2), nitric acid (HNO_3), and potassium hydroxide (KOH) that are effective oxidizers used in chemical pretreatment of organic materials and biological materials in environmental matrices. However, some studies reported issues such as polymeric changes (PA, NY6, NY66 degradation) in MPs after treatment with acid and alkaline solutions, shrinkage in size, and discoloration in MPs (discoloration and/or transparency) (Lv et al. 2021). For example, it

Table 2.6 Chemical resistant chart of polymers (Adapted URL 1 n.d.)

Alphabetical Listing of Materials	Concentration + Weight %	ABS	Acrylic	CAB	CPVC	HDPE	Nylon®, Type 6/6	PET	Polycarbonate	Polypropylene	Polysulfone	PPS	PVC, Type I	PVC, Type II	PVDF	PTFE
Acetaldehyde Aq.	40	D	D	*	D	C	B	A	*	C	*	A	D	D	D	A
Acetic Acid Aq.	10	*	B	C	A	*	C	B	D	*	A	A	A	A	B	A
Acetone		D	D	*	D	A	A	B	C	A	B	A	D	D	D	A
Alcohols, Aliphatic		*	D	*	*	*	B	A	*	*	*	A	*	*	A	A
Aluminum Chloride Aq.	10	*	*	A	A	B	*	A	A	A	*	A	A	A	A	A
Ammonium Chloride Aq.	10	*	*	A	A	A	A	A	C	A	*	A	A	A	A	A
Benzene		D	D	D	D	D	A	A	D	D	D	A	D	D	C	A
Boric Acid Aq.	10	*	*	*	A	A	A	A	*	A	*	A	A	A	A	A
Butanol		*	*	*	A	A	B	B	*	*	C	A	A	D	*	A
Chloroform		D	*	D	D	C	D	D	D	D	D	A	D	D	B	A
Chlorosulphonic Acid Aq.	10	*	D	*	*	D	D	*	*	C	*	D	C	C	D	A
Chromic Acid Aq.	10	*	D	A	A	A	C	A	C	A	D	B	A	D	B	A
Citric Acid Aq.	10	B	C	B	A	A	C	A	A	A	A	A	A	A	A	A
Detergents, Organic		*	*	*	A	*	A	A	*	*	*	A	A	A	*	A
Ether, Diethyl		*	*	*	*	*	A	A	*	*	*	A	*	*	*	A
Ethyl Acetate		D	D	*	*	C	A	*	D	A	*	A	D	D	D	A
Ethylene Glycol Aq.	96	*	A	D	A	A	B	*	C	A	A	A	A	A	A	A
Ferrous Chloride Aq.	10	*	A	*	A	*	C	*	*	*	*	A	A	A	A	A
Formaldehyde Aq.	40	*	A	A	A	A	B	A	C	A	A	A	A	B	A	A
Formic Acid Aq.	3	*	D	*	A	*	B	B	A	*	*	A	A	A	A	A
Hydrobromic Acid Aq.	10	A	*	*	*	*	D	*	*	C	A	A	A	A	A	A
Hydrochloric Acid Aq.	0.4	*	A	B	A	A	*	A	A	A	A	A	A	A	A	A
Hydrofluoric Acid Aq	4	*	C	C	*	B	C	B	B	A	B	D	A	A	A	*
Hydrogen Peroxide Aq.	0.5	*	*	A	*	A	C	A	A	A	A	A	A	A	B	A
Nitric Acid Aq.	0.1	B	A	D	A	A	C	*	A	A	A	*	A	A	A	A
Sulphuric Acid Aq.	2	B	D	*	A	A	C	A	A	A	A	A	A	A	B	A
Trichloroethylene		*	*	D	D	D	B	B	*	D	D	A	D	D	A	A

A = No Attack, possibly slight absorption. Negligible effect on mechanical properties.
B = Slight attack by absorption. Some swelling and a small reduction in mechanical likely.
C = Moderate attack of appreciable absorption. Material will have limited life.
D = Material will decompose or dissolve in a short.

Aq. = Aqueous Solution
* = No data available

has been reported that NaOH solution saponifies the ester bond, impairing the structural properties of PET (Rostami et al. 2020). In contrast to the low acid/base resistance of polymers, it was reported that PE and PVC are resistant to acidic and basic pretreatments (Lusher et al. 2017). It was also stated that if strong acid was used, it might cause the IR spectrum to be incompatible with the standard reference of the MP in the library. For example, in one study, it was stated that chemical degradation was not observed in the polymer structure after pretreatment with HNO_3, but there was a change in the IR spectrum (Ghosal et al. 2018). Therefore, it was reported that the most preferred oxidant is H_2O_2 (10%, 30%, and 35%) (Lee and Chae 2021).

2.5.2.2 Wet Oxidation Peroxide (WPO)

This method has been used successfully to determine polyethylene, polypropylene, polyvinyl chloride, and polystyrene in both surface water and sediment samples (Marine Debris Program 2015; Sutton et al. 2016). The two most important parameters affecting the efficiency of wet peroxide oxidation (WPO) are temperature and oxidation time. At the beginning of the process, the sample is first dried at 60 °C and then placed in a beaker by adding 20 mL H_2O_2 (30%) to be heated for 30 min. When all organic materials cannot be removed, it is recommended to add more 20 ml H_2O_2 (Lares et al. 2018). The oxidation efficiency is determined to be high at 50 °C. When the solution temperature reaches 75 °C, a severe boiling occurs.

2.5.2.3 Treatment with Fenton Reagent (Fe^{2+} with H_2O_2)

This method is used for samples having high organic matter contents. Fenton reagent can remove approximately 90% of organic substances within 2 h under room temperature without losing the chemical and physical properties of MPs (Hurley et al. 2018). If the ambient temperature is increased, the processing time can be reduced to 30 min (Sol et al. 2020). Wet peroxide oxidation (WPO) is a reaction that can be performed on its own using only H_2O_2. As an alternative approach, it has been proposed to use an iron catalyst (Fe^{2+}) to lower the reaction temperature used in oxidation with H_2O_2. Organic removal with Fenton reagent is a frequently used method to separate structurally fewer resistant polymers with minimal damage because it offers higher yields at lower temperatures compared to WPO (Hurley et al. 2018). However, this method has some disadvantages compared to WPO. It was reported that when oxidation starts if oxidation does not perform in a water bath, the temperature can rise above 60 °C instantaneously. This may cause some microplastic types to degrade (Lusher et al. 2020).

2.6 Filtration

Since there is no standard method in the isolation of MPs, the filter paper pore sizes used in the studies are 0.2 μm (alumina oxide), 0.45 μm (GF/C), 1.2 μm (GF/C), and 5 μm (silicone, silver) (Robertson 2018). Unfortunately, these different pore sizes in filter papers lead to have different results (Lusher et al. 2020). Hanvey et al. (2017) reported that glass fiber filters (Whatman GF/A and GF/C or GF/F) were the most used filters. Filter papers used in microplastic analysis and their properties are given in Table 2.7.

The subdimension index in MPs studies is changed due to the lack of a standard filter paper pore diameter. The use of filter paper with different pore diameters hinders the comparability of the data. The filter pore diameters have been found to have an important impact on the abundance of MPs. Researchers are expected to reveal the type of filter and pore diameters that they used when reporting the data. In addition, while categorizing MPs according to their colors, it is observed that blue, green, red, black, and colorless/transparent structures are the most common colors (Young and Elliott 2016). In color categorization with a stereomicroscope, if the background is white or transparent depending on the color of the filter paper, detecting MPs that have the same color as the filter paper becomes difficult. This will result in having different results in the abundance of microplastic detected on the filter paper. The stage of defining MPs isolated from the study area in the laboratory environment is found to be dependent on the aim of each study. MPs can often be identified visually with a binocular/stereomicroscope, but the reliability of the method is considered low for small, transparent, and (or) fiber-type particles (Song et al. 2015; Lenz et al. 2015; Shim et al. 2016). For example, Çullu et al. (2021) evaluated the color scale of MPs in the nutrition of primary and secondary consumers, and transparent films that lost color during oxidation analysis were omitted to avoid errors in the identification of colors.

In the current study (Cai et al. 2020), the retention efficiency of MPs on filter paper was found to be dependent on their morphological properties. Nylon filter (double-layer-hole type) sustained almost 100% of fibers, whereas only 61.7% of fibers were sustained in polycarbonate filter (single-layer-hole type). 80.8% and 54.4% of the fragments were maintained in polycarbonate filter and cotton fiber filter (multilayer-hole type), respectively. It is found that 50 μm pore-size filters have a small fragment of 37.2 μm. Field waters containing MPs were filtered off with filters having various pore size filler to confirm the laboratory results. The correlation between abundance and pore size is revealed to follow the same trend in the synthetic laboratory fiber samples as anticipated. Hence, their results revealed that the structure and the pore size of the filter would have an impact on the abundance of MPs in various shapes. They reported that water samples are recommended to be filtered with 20 μm pore-size filters with a double-layer-hole type of structure. Moreover, they investigated the filter substrates and reported that a gold-coated polycarbonate filter was considered as an optimal material for microplastic particles loading.

Table 2.7 Microplastics, filter types, and analysis solution. (Reprinted from Horiba et al. (2021))

Filter type	Pore size	Optical quality	Handleability	Interference
Borosilicate glass fiber	Lowest 0.6 µm	Rough surface can reduce ability to identify MPs (most significant for small particles, below 10 µm) White membrane low contrast for transparent plastics	No issue	Possible interference signals for Raman and infrared microscopy
Polycarbonate uncoated	Lowest 0.2 µm	Flat surface. White membrane low contrast for transparent plastics	Issue in case of alkali treatment (KOH)	Strong interference with Raman and infrared microscopy. Polycarbonate shows strong bands both in Raman and infrared. Not usable for transmission infrared microscopy
Polycarbonate coated (gold, silver)	From 0.2 to 5 µm	Flat surface and high reflectivity and good contrast Highly textured surface for silver	Issue in case of alkali treatment (KOH)	Less interference than uncoated but still present if metal is thin and for particles below 5 µm. Not useable for transmission infrared microscopy
Alumina	From 0.02 to 0.2 µm	Flat surface. White membrane low contrast for transparent plastics	Highly fragile; careful handling required	Low interference for FT-IR (peak intensity change over the filter) and for Raman (broad spectral feature) Useable for transmission infrared microscopy but no signals below 1250 cm^{-1}
Silicon	From 1 to 18 µm	Flat surface. High reflectivity and good contrast	Easy handling; possible fragility along crystalline direction. Square shaped (dedicated holder needed)	Raman (silicon peaks do not interfere with plastic peaks) FT-IR (possible interference from silicon oxide) Useable in transmission infrared microscopy

2.7 Identification, Chemical Characterization, and Quantification

2.7.1 Microscopy Analysis

Visual description is a method used to determine the abundance of MPs and their morphological properties including their types and colors. However, in general, it is found to be an insufficient analysis method without an additional method. A binocular/stereomicroscope can lead not to detect the small and transparent microplastic fragments. Therefore, in the case of many samples being investigated, combining microscopy analysis with spectroscopic methods is recommended (Song et al. 2015).

Microscopy analysis is a method that tends to give an error caused by human; therefore, it requires experience. MPs can be mixed with biomaterials (e.g., dried algae, seeds, charcoal, and leaves) (Lavers et al. 2016). In the study by Eriksen et al. (2013), about 20% of the particles, that are defined as microplastic visually, were detected as aluminum silicate by Scanning Electron Microscope (SEM). Other particles can be determined as MPs due to similar properties resulting in underestimating or overestimating the abundance of MPs (Hidalgo-Ruz et al. 2012; Lv et al. 2021). However, when analyses are performed by an expertise, MPs may have several distinguishing features that may arise when compared to biological and/or other inorganic particles. For example, they are more noticeable and vibrant colors, irregular physical profiles, or more irregular geometries that are distinguished compared to particles detected in nature (Lusher et al. 2020). As long as the polymer-chemical structures of MPs are supported by further analysis, this method can be used in counting and determining MPs having 100 μm and above (Lusher et al. 2020). MPs that can be determined by eye and detected by binocular microscopy and SEM are represented in Figs. 2.5, 2.6, and 2.7.

Determining MPs under the stereomicroscope has become easier with the selected dyes in recent days. Nile Red, commonly used dye, implemented successfully in the determination of MPs (Shim et al. 2016; Erni-Cassola et al. 2017). Nile Red was used to detect MPs (20 μm–1 mm) in different aquatic samples and is highly effective in identifying PE, PP, PS, and PA particles (Erni-Cassola et al. 2017). However, choosing a suitable dye for all polymer types is not possible. Furthermore, while dyeing is a suitable method for determining the abundance and types of MPs (fiber, fragment, etc.), it is not a suitable method for determining the colors of MPs. Since all MPs have the fluorescence feature under the microscope, their own colors cannot be observed. Transparent MP particles can be counted as colored parts because of containing some polymers structures in them which is regarded to be a threat.

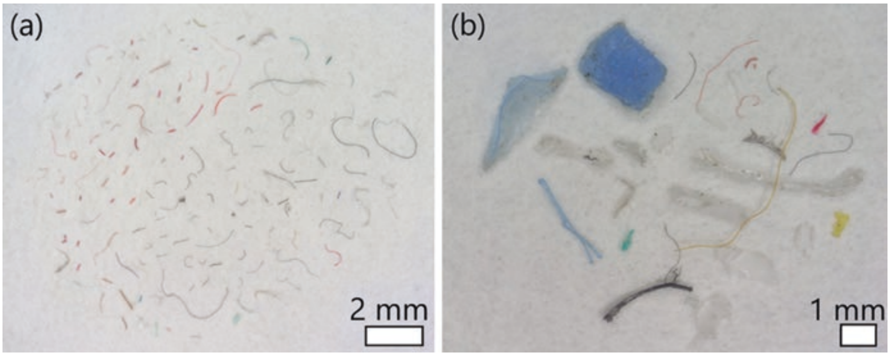


Fig. 2.5 The fibers that can be seen by eyes. (**a**) Microplastics fibers (Akarsu et al. 2020). (**b**) Microplastics fragment, fibers, and films (Akarsu et al. 2020)

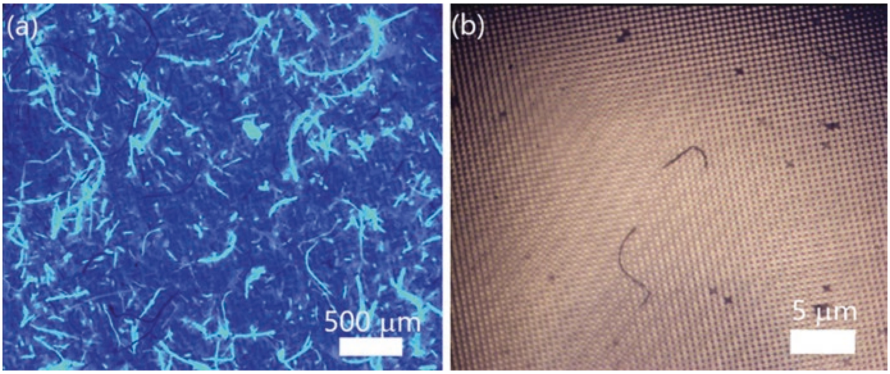


Fig. 2.6 Fibers detected by binocular microscope (500 µ). (**a**) Image taken with UV modification with a binocular microscope (Akarsu and Deniz 2021). (**b**) Image taken with binocular microscope without modification (Akarsu and Deniz 2021)

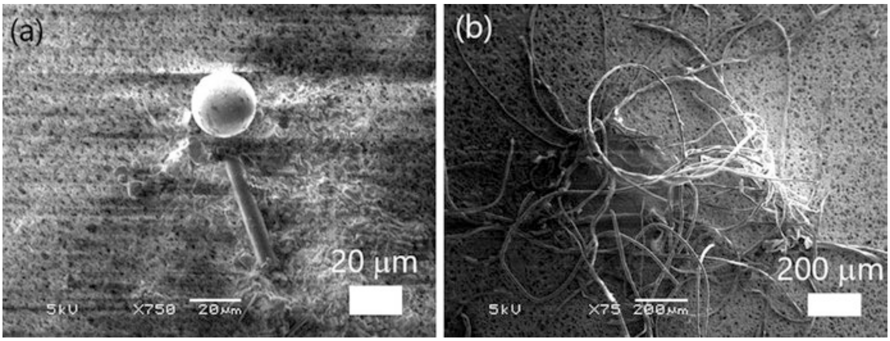


Fig. 2.7 SEM figures of MPs in different sizes: (**a**) microbead fragment (Çullu et al. 2021) and (**b**) fibers (Çullu et al. 2021)

2.7.2 *Instrumental Analysis of Microplastics*

Studies on MPs in aquatic areas are often limited to the morphological properties and density evaluations of MPs. Use of advanced technology is necessary for the determination of the microplastic source and further behaviors. It has been shown that various sampling techniques can lead to different results in the detection and quantification of MPs in the environment. However, it has been known that the spectroscope method is more preferred over the microscope as it increases the accuracy of polymer counting and enables improved characterization of polymers. In this part of the chapter, the most frequently used methods in the analysis of structures of MPs will be discussed.

Identification of microplastic particles using chemical characterization techniques is carried out by infrared microscopy, Raman microscopy, pyrolysis mass spectrometry, and gas chromatography. Combinations of these techniques with the other techniques have also been used in recent studies (Yu et al. 2019).

The current advantages and disadvantages of MP's characterization techniques play a fundamental role in the selection of analysis techniques. The advantages and limitations of each identification and characterization method should be considered for MPs analysis based on micro−/nanoscale scales and similar chemical structures (Yu et al. 2019). For example, the combination of TGA/FT-IR analysis for MPs can be used in the analysis of other types of polymers excluding PE, PP, and PET (Yu et al. 2019). Therefore, it is very important to have an analysis method selection for all the obtained microplastic sample composition.

2.7.2.1 Infrared Microscopy: Transmission/ATR/Micro-Fourier Infrared Transform (μ-FTIR) Spectroscopy

Fourier transform infrared (FT-IR) spectroscopy has been comprehensively used in the identification of MPs pollution present in sediment and water (marine and freshwater) studies (Kovač Viršek et al. 2016). In addition, transparency or white color fragment cannot be detected under the microscope; therefore, it can be analyzed by FT-IR (Yang et al. 2021).

FT-IR is one of the nondestructive distinguishing techniques with polymer database (Yang et al. 2021). The spectrum of a polymer is a plot of measured infrared intensity versus wavelength of light. The wavelength at which the bands emerge depends on the masses of the atoms, the bond strength invariant, and the geometry of the atoms. FT-IR spectroscopy provides information about the chemical bonds found in molecules (Chalmers 2000). FT-IR fundamentally has two operation mode, reflection and transmission (Yang et al. 2021). The schematic approach of FT-IR spectroscopy modes including the Attenuated Total Reflectance (ATR) is given in Fig. 2.8. ATR FT-IR is directly used in the authentication of MPs in reflection mode. The sample surface in ATR FT-IR analysis must be smooth to obtain the spectrum correctly. Therefore, an expert with experience in interpreting spectra is needed for

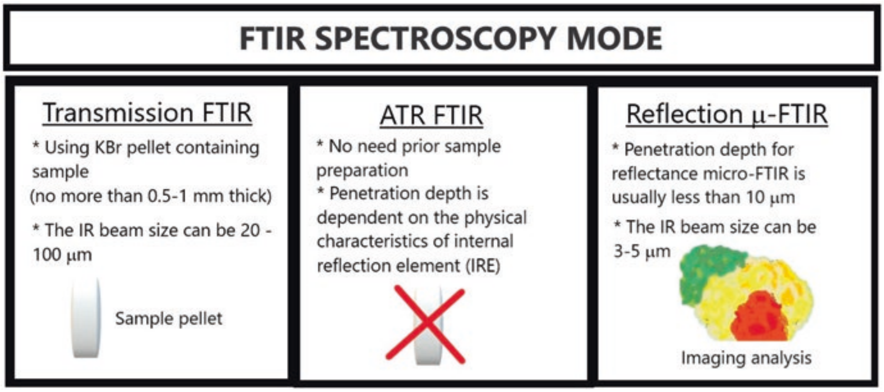


Fig. 2.8 Schematic comparison of FT-IR spectroscopy modes

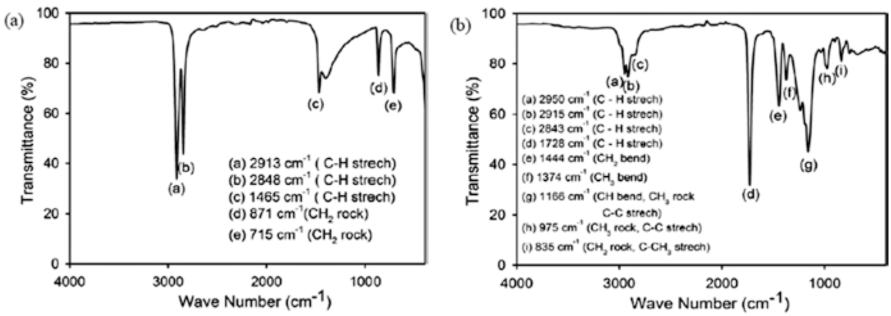


Fig. 2.9 FT-IR analysis of (**a**) polyethylene and (**b**) polypropylene (Altay et al. 2019)

the identification of MPs by FT-IR. The reflection mode provides an advantage at this point for this reason only. For smaller particles, micro-FTIR (μ-FTIR) must be used.

Also, three FT-IR analysis spectrums of polymers are given as an example in Fig. 2.9a, b. As seen from Fig. 2.9a, b, the samples are identified as PE and PP, respectively, from vibration peaks of spectrum (Altay et al. 2019). As can be seen from the spectrum, it represents the fingerprint of a sample (MP). These fingerprint patterns are used to identify microplastic samples composition. The functional groups in the material can be easily identified as the absorption peaks correspond to vibration frequencies between the bonds of the atoms. For this reason, FT-IR is a rapid and reliable method to identify polymer types of different MPs by comparing the resulting FT-IR spectra with known plastic polymers. Some researchers reported that they frequently encountered deterioration of the polymer structure after analysis (Yurtsever 2019; Yap et al. 2020). Before spectral analysis, it should be ensured that the structure of the sample is not deteriorated after the organic matter is removed. Otherwise, the sample analysis results, which are disrupted by strong acids, do not reflect the actual results.

In a study on the contributions of FT-IR spectroscopy to microplastic pollution research in 2020, it was stated that FT-IR analysis technique was used in more than 400 publications which were published between January 2010 and December 2019 (Veerasingam et al. 2020). However, the experience gained over the past years has contributed to improve analytical technologies. The μ-FTIR analysis method which has recently become a trend in microplastic characterization is an example of this development (Chen et al. 2020). It has enabled the analysis of very small amounts of samples by micro (μ)-FTIR spectroscopy method. Table 2.8 shows the μ-FTIR analysis results of freshwater and sediment microplastic samples, especially from recent studies. The μ-FTIR spectroscopy has high analytical sensitivity and spatial resolution. It consents the identification and determination of mixtures of microstructure compounds that constitute samples (La Russa et al. 2009; Chen et al. 2020). On the other hand, the improvement that has contributed to advance in FT-IR imaging is applying focal plane array (FPA)-based detection (allows detection and identification of plastics smaller than 20 μm), where several detectors are placed in a grid pattern (Mintenig et al. 2017; Ivleva et al. 2017). The first study that used FPA-based μ-FTIR imaging to analyze MPs with size down to 20 μm from environmental samples was reported in 2015 (Löder et al. 2015). An infrared map obtained by (FPA)-based μ-FTIR detects MPs by scanning the surface of filters-held MPs. As single particle analysis is not feasible, MPs particles are usually collected on a filter (Hidalgo-Ruz et al. 2012). FPA-FTIR analysis technology is an ideal model to identify MPs due to independence of sample thickness. FPA-FTIR has a high spatial resolution, that is, 5.5 μm in reflection and 1 μm in ATR mode. The acceptable limit is of 5–10 μm (Yang et al. 2021).

2.7.2.2 Raman Spectroscopy

In freshwater sediment, Raman spectroscopy, which is preferred for a nondestructive detection method in micro- and nanoplastic studies, is widely used. The advantage of Raman spectroscopy which enables measurements of vibrational fingerprint spectra is the high spatial resolution which the sample is irradiated with a monochromatic light source, normally a laser (Imhof et al. 2016; Ivleva et al. 2017; Yang et al. 2021). The laser with the single wavelength is operated to excite the molecule; the radiation interaction with the sample is identified (Li et al. 2018). The spatial resolution of the Raman microscope increases with decreasing the excitation wavelength of the laser (Anger et al. 2018). The different spatial resolutions can be obtained depending on sample material properties and the laser wavelength in Raman. The lasers are frequently used in the UV-vis range that allowed spatial resolution in the micrometer range for Raman analysis of MPs. But the limiting factor is the size of the sample for selecting laser wavelengths range. Lasers with shorter wavelengths enable the detection of smaller particles, indicate higher intensities of backscattered light, and at the same time give onto higher interferences via fluorescence (Huppertsberg and Knepper 2018). The weak Raman signal grade is based on fluorescence, and then measurement circumstances (integration time and number of

Table 2.8 Recent microplastic studies of surface water, freshwater, and sediment samples using μ-FTIR analysis method

Location	Sample type	Analysis method	MP's composition	Reference	Year
Ross sea coast (Antarctica)	Surface water	FT-IR equipped with a microscope (reflectance mode) and an FPA detector with high spatial resolution	Predominant abundance is PE and PP	Cincinelli et al. (2017)	2017
Southern Yellow Sea and East China Sea	Sediments	μ-FTIR in transmittance mode	Cellophane 37.2%, PET 21.6%, PE 17.6%, polyester 11.8%, acrylic 9.8%, and cellulose 2.0%	Zhang et al. (2019a)	2019
Western Pacific Ocean	Deep-sea sediments	μ-FTIR	Poly (propylene-ethylene) copolymer (40.0%), PET (27.5%), and others	Zhang et al. (2020)	2020
Lake Guaíba (Porto Alegre, Brazil)	Fresh water	μ-FTIR and μ-Raman spectrometer (laboratory assembled)	PP (54.5%), high- and low-density PE (43.3%), PTFE (0.5%), PA (0.5%), PU (0.5%), and PS (0.5%)	Bertoldi et al. (2021)	2021
Zhejiang Province, (Southeast China)	Sediments	A combined method of μ-FTIR and Nile Red (NR) staining	PE (25.5%), PP (15.7%), PS (including EPS, 11.8%), and PA (9.8%) in first sediment pile. PE (20.8%), PA (16.7%), PS (14.6%), and PVC (12.5%) in second sediment pile	Ji et al. (2021)	2021
Eastern Indian Ocean	Surface water	μ-FTIR	The wide majority of MPs is consisted of PP (51.11%) and PE (20.07%)	Li et al. (2021a)	2021
Guangdong Coastal Areas, South China	Surface water and sediments	μ-FTIR + Raman spectroscopy	Rayon (38.2%), PET (16.4%), EVA copolymer (12.73%), and PAM-11 (PA) (10%) in surface water Rayon (31.3%), PET (23.5%), and PE (20.9%) in sediment samples	Li et al. (2021b)	2021

scans) should be optimized (Lenz et al. 2015). Nevertheless, major source of fluorescence in Raman analysis will be existing of surface altered (due to oxidation, aging, etc.) or inadequate prepared sample (Huppertsberg and Knepper 2018). For example, the PVC degradation by UV in spectrum is revealed by a simultaneous intensity reduction of peaks at 693 and 637 cm^{-1} (Silva et al. 2018). It is necessary to remove disturbing biological components by an effective sample preparation to

Table 2.9 Recent microplastic studies of surface water, freshwater, and sediment samples using Raman and Raman μ-spectroscopy

Location	Sample type	Analysis method	MPs composition	Reference (year)
Atlantic Ocean	Seawater	RμS with a 455 nm excitation laser	PP and PE	Lenz et al. (2015)
Atlantic Ocean	Surface seawater	RμS with 455 nm laser wavelength	PE and PP	Enders et al. (2015)
Bay of Brest, Brittany, France	Surface seawater	RμS	PE, PP, PS, and PUR	Frère et al. (2016)
Lake Garda, Italy	Sediment	Raman spectroscopy with a He-Ne laser (632.8 nm)	PE, PA, PET PS, and PP	Imhof et al. (2016)
Warnemünde, Germany + Gotland Basin, between the Swedish island, Gotland, and the Latvian coast	Sediment	RμS with 532 nm radiation of a Nd:YAG laser	Generally, PE copolymers and oxidized PE, PP, PVC, PC, PS, PET, and PTFE	Käppler et al. (2016)
River Thames basin, UK	Fresh water	Raman spectroscopy using a near infrared laser (785 nm)	PP, PES, and PAS	Horton et al. (2017)
Southern North Sea	Surface water	Raman μ spectroscopy	PE, PP, PS, PMMA, and CA	Cabernard et al. (2018)
Pacific Ocean	Oceanic water	Raman μ-spectroscopy with near-infrared 785 nm diode lasers	PE, PP, and PS	Ghosal et al. (2018)
Laizhou City, Shandong Province, China	Sediments	RμS with a 532 nm laser wavelength	PE, PP, PET, and PVC	Dong et al. (2020)
Tunisian coasts	Sediments	RμS with near-infrared laser (785 nm)	PE, PP, LDPE, HDPE, PA, and PEVA	Missawi et al. (2020)
The Vistula River, Poland	Fresh waters	RμS with a 532 nm laser wavelength	PET, PS, and PU	Kaliszewicz et al. (2020)

refrain from fluorescence during the Raman measurement. Fluorescence due to the existence of a biofilm laminates the Raman signal, which can completely hinder particle identification (Käppler et al. 2015). Furthermore, Raman spectroscopy allows wet samples to analyze (Yang et al. 2021).

The Raman spectroscopy coupled with optical microscope named as micro-Raman spectroscopy (RμS) used are normally in the visible range. (Frère et al. 2016; Ivleva et al. 2017). RμS provides compositional information at the micrometer scale and is suitable for the characterization of small plastic particles in the marine environment (Ghosal et al. 2018; Wu et al. 2020). For Raman microspectroscopy analysis, the right combination between laser wavelength and sample holder should be decided to improve accurate particle detection and appropriate Raman signal (Frère et al. 2016; Huppertsberg and Knepper 2018).

Small microplastic particles (<1 mm) were detected by FT-IR particularly, including those <50 μm. Although FT-IR can determine small MPs down to 10–20 μm, however, plastics smaller than the aperture size are not determined. Raman spectroscopy, using a laser beam, can focus on a smaller area than FT-IR and detect MPs down to 1–2 μm in size (Song et al. 2015; Habib et al. 2021). Another way, Raman spectroscopy has a great advantage compared to FT-IR providing a better resolution and response of nonpolar, symmetric bonds, extensive spectral inclusion. However, it should be noted that FT-IR still allows more clear identification of polar groups (Silva et al. 2018). The results of the Raman and RμS analysis from surface water, freshwater, and sediment samples carried out from 2015 to 2020 are given in Table 2.9. As seen from Table 2.9, the most common types of waste MPs such as PE, PP, PET, etc. have been identified by Raman spectroscopy at different laser wavelengths in worldwide locations.

2.7.2.3 Thermo-analytical Methods: Pyrolysis Mass Spectrometry Gas Chromatography (Pyr-GC/MS) and Thermo-extraction Desorption Gas Chromatography (TED-GC/MS))

Thermo-analytical methods are alternative techniques to complementary of imaging techniques or infrared spectrometry (Primpke et al. 2020). They provide information on pyrolytic decomposition of polymers that occurred at increased temperature and at the same time formed smaller molecules that can be analyzed using spectroscopic techniques such as IR spectrometry and mass spectrometry during analysis (Primpke et al. 2020). Recently, two main techniques are applied for mass-quantitative MPs analysis, pyrolysis-GC/MS (Py-GC/MS) (Fabbri 2001; Fischer and Scholz-Böttcher 2017; Gomiero et al. 2019; Dierkes et al. 2019; Funck et al. 2020; Okoffo et al. 2020; Steinmetz et al. 2020; Sullivan et al. 2020) or thermo-extraction desorption GC/MS (TED-GC/MS) (Dümichen et al. 2017; Eisentraut et al. 2018; Duemichen et al. 2019). Analytical pyrolysis coupled to gas chromatography and mass spectrometry can ensure both qualitative and quantitative data on polymer mixtures. Thermo-analytical methods are destructive techniques in contrast to spectroscopic methods. The sample is thermally decomposed under defined conditions. The analysis device consists of specialized units like pyrolizers or thermogravimetric systems (Primpke et al. 2020). MPs can be analyzed by heating to temperatures above 500 °C, and they are pyrolyzed into many individual fragmentation substances, which can be then separated chromatographically and identified by mass spectrometry (Kusch 2012). It is reported that in the separation of the thermal extraction process from the thermal desorption with the TDS-GC-MS, there is no contamination of a transfer capillary like at Py-GC-MS (Dümichen et al. 2017). But these techniques are almost new; therefore, no standardized protocols are available yet for MPs analysis. Comparing with Pyr-GC-MS, thermo-extraction and desorption coupled to gas chromatography-mass spectroscopy (TED-GC-MS) has been used to measure relatively large numbers of sample masses (100 mg) for analyzing complex and nonhomogeneous samples (Duemichen et al. 2014). Otherwise, pretreatment of standard MPs particles is not required when employing TED-GC-MS to characterize MPs (Elert et al. 2017). On the other hand, Pyr-GC/MS can detect both polymer type of a microplastic particle and contained plastic additives simultaneously (Käppler et al. 2015). Hence, the number of publications dealing with Py-GC-MS analysis of MPs is rapidly increasing (Matsui et al. 2020; La Nasa et al. 2021; Matsueda et al. 2021). So far, only a few studies of fresh water or sediment have used pyrolysis mass spectrometry (e.g., Py/GC/MS or DSC) for analysis (Castañeda et al. 2014; Fries et al. 2013; Wu et al. 2020).

2.8 Results and Accurate Reporting Criteria

In this review study, it is revealed that there is yet no conclusion that has been made by the researchers in the presentation of numerical data obtained from the analysis. Generally, the results of MPs that are given as particles L^{-1} or particle m^{-3} indirect sampling studies (Aytan et al. 2016; Vianello et al. 2018; Uurasjärvi et al. 2020) and particles m^{-} or particles km^{-2} in studies (Gündoğdu and Çevik 2017; Gray et al. 2018; Vianello et al. 2018; Migwi et al. 2020). The differences in the units were explained due to sampling methods and study purposes in aquatic areas (Çullu et al. 2021). Furthermore, the filtration process that was conducted before visual identification is another factor affecting the MPs abundance. MPs below a certain size scale are not considered depending on the filter type and pore diameters used. The numerical result obtained may differ based on the pore diameters of the filters. This situation causes problems in terms of comparability of MPs data by various researchers. However, it can enhance the comparability of data acquired by various researchers working in similar study fields. Moreover, utilizing statistical analysis is one of the significant tools to facilitate the comparison of and correct interpretation of the results. The image of the filter paper obtained using a high-resolution camera after the coloring process can be used for automatic counting, type, and size determination of fluorescent MPs with the Microplastics Visual Analysis Tool (MP-VAT) program (Prata et al. 2019, 2020). This method is worth using in the identification of MPs under the stereomicroscope, minimizing the errors created by human and saving time (Lv et al. 2021).

Both FT-IR and Raman spectroscopy methods are costly; however, they are considered reliable methods in characterization. The researcher needs to consider that infrared spectroscopy methods are the methods that do not damage the sample and thermal analysis methods that damage the sample, and the analysis should not cause sample loss during the research when they choose the analysis. Alternatively, analyzing MPs under fluorescence microscopy applying treatment with hydrophobic fluorescent dyes such as Nile Red can be performed easily. However, since the filter paper will be colored in the treatment with dye, the filter paper should be a filter type that is compatible with dyeing. In the literature, PC filter papers are identified as ideal filter papers for dyeing processes (Zhu et al. 2020).

Using an FT-IR or Raman spectroscopy for chemical identification of MPs, including qualitative verification of polymer types is the most common approach to minimize errors in determining MPs (Song et al. 2015). Quality assurance based on analytical quality control and validated analytical methods plays a significant role in identifying MPs correctly. Instrumental analysis techniques conducted with analytical devices are being preferred rather than visual techniques in the determination of MPs since it is usually impossible to visually demonstrate the composition (SEM/EDS or microscope) of transparent specimens. In this case, information on the material composition can be obtained using only infrared or thermal analysis methods. The results that are obtained from all of these applied analysis methods (apart from SEM/EDS) can be confirmed with the compositions of polymers demonstrated in the polymer library available on the instrument. However, given polymers in long-chain and copolymer form, it is unavoidable that the identification procedure becomes complicated. In the case of determining MPs with an unknown composition, the researcher needs to be an expert in examining and interpreting the material analysis results.

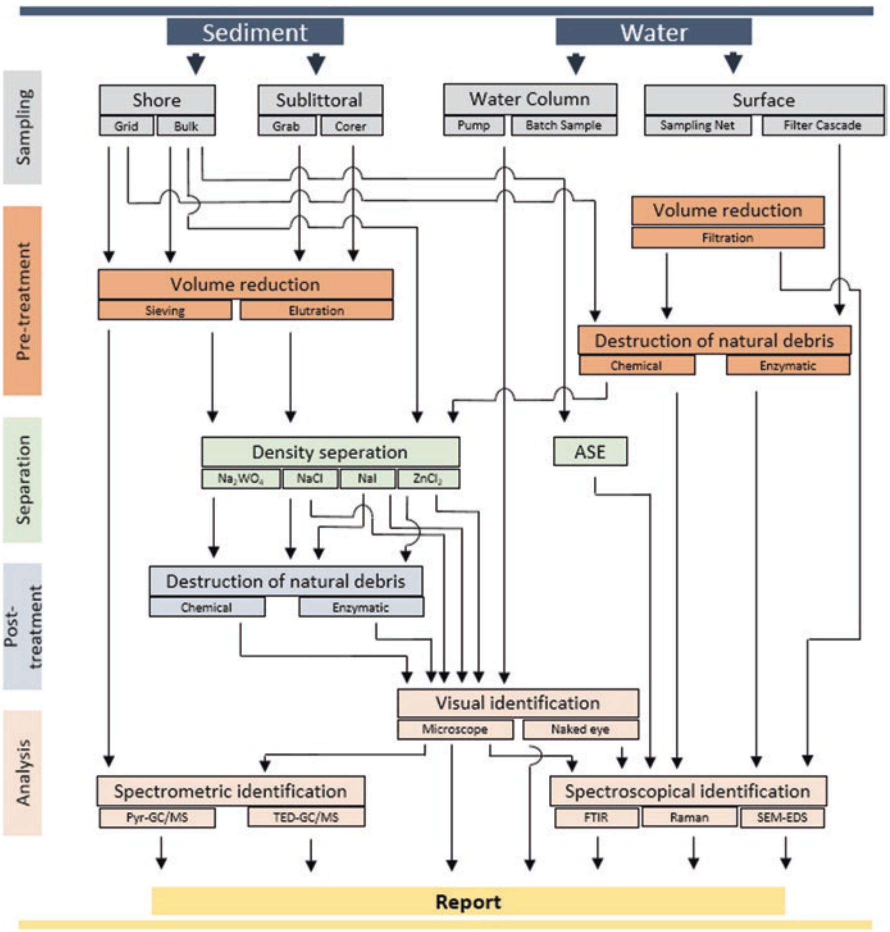
2.9 Recommendations and Future Works

Given the difficulty of removing wastes from the marine environment, the potential effects of plastic waste on marine life are presented (Andrady 2011). The occurrence of all macro-, meso-, or microplastic wastes in the aquatic environment causes a significant harm to aquatic ecosystems (Barnes and Milner 2005). Particularly, the physiological and toxicological effects of the different polymer compounds that are used to make plastics in aquatic environments give rise to irreversible destruction of biodiversity. Detecting the plastic wastes in many ecosystems, especially in coastal areas, proves that the requirement of ensuring controlled production, reduction at the source, and recycling is implemented in every waste management planning.

The ongoing studies have developed some new methods to mitigate the micro- and nanoplastic waste. However, the present situation analyses and projections should be standardized, and this standardized method should be embraced by all researchers in order to implement these methods faultlessly. Moreover, considering that these studies will be conducted throughout worldwide, a cost-effective method requires less equipment, and field specialist should be developed. In many of the studies, not advanced analyses were conducted, and differentiations in the abundance and type of MPs were observed which is attributed to difference in an individual sampling method. The advanced techniques including FT-IR, Raman, and GC/MS are utilized in the analysis of the MPs to enhance the accuracy of the results.

Determining the future status and ecological effects of MPs and having successful and sustainable steps in the reduction of the MPs can be obtained with the studies that will be performed. However, it is unavoidable to apply various control measures in different areas and variants depending on all variables that have been given in this chapter of the book. The researcher needs to have a common method and clear protocol to be able to decide how to select the equipment that will be utilized for the determination of polymer types. The studies on the quantification of the MPs would allow a microplastic threshold level to be suggested and control measures to be demonstrated for the reduction in the use of plastics and consequently its threat to the aquatic environment. The next steps to be taken in this matter will ensure that future studies will be utilized at the desired level.

Graphical Abstract



Adapted from Wagner and Lambert (2018)

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Chapter 3
Monitoring of Microplastic Pollution

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Abstract Microplastic accumulation in marine ecosystem is the potential environmental hazard because of its adverse impacts on the marine life. Bioaccumulation and biomagnification of microplastic particles to the higher level in food chain is an associated serious concern. Monitoring of microplastic in marine ecosystems can be done using a number of methods, i.e., by direct observations; using ships and aerial views, GIS, and trawl surveys; using Remote-Operated Vehicles (ROVs), etc. Few scientific studies have evaluated the temporal trends in plastic accumulation in the marine environment. These include directed efforts of shoreline monitoring through monthly and annual sampling on beaches, seafloor, and surface waters. Various temporal trends that have been observed suggest that there is annual as well as seasonal increase in the marine microplastic pollution. This infers that only yearly monitoring is not sufficient, and there must be a seasonal or, more precisely, monthly sampling in order to have more accurate pattern of the changes occurring in microplastic accumulation. Plastic debris can also be monitored using indicator species. A significant example is the determination of plastic ingestion by *Fulmarus glacialis* or northern fulmars. It began in 1980s, and the plastic levels in the animal are used as a measure of the accumulation of plastic in European Coastlines and North Sea for OSPAR Ecological Quality Objective on marine litter.

Keywords Monitoring · Microplastic · Pollution

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3.1 Introduction

Plastic accumulating in the marine ecosystem is one of the most potential environmental concerns because of its hazardous impacts (Morét-Ferguson et al. 2010; Martins and Sobral 2011; Lee and Sanders 2015; Suaria et al. 2015). Various temporal trends that have been observed suggest that there is annual as well as seasonal increase in the marine microplastic pollution. This infers that only yearly monitoring is not sufficient, and there must be a seasonal or, more precisely, monthly sampling in order to have more accurate pattern of the changes occurring in microplastic accumulation, although the quantification of amount and sources of such ever-increasing amount of plastic in the marine environment still remains difficult. It is because monitoring methods are expensive and time-consuming as well. Moreover, data with respect to time series is essentially required that can be used to address variation in the quantity of litter over time in the marine ecosystems. It means monitoring includes changes in spatial and temporal distribution of the microplastic waste (Lusher et al. 2014).

Microplastic can be induced to the marine environment through a number of different pathways. They can be released into household or industrial drainage systems, can be washed into water as synthetic fibers from clothing, or can flow mixed with the effluent from wastewater treatment work (WWTW). Another way is the movement of MP particles along with wind, storm, sewer, water current, and water runoff. Sewage sludge is another potential source since its MP content is even greater than the effluents being released into the aquatic ecosystem. Being small particles, they can easily flow in the water systems and reach in all areas of the marine habitat (Auta et al. 2017). International Union for Conservation of Nature have quantified microplastic release into the oceans and the results published in its report are represented in the Fig. 3.1 (Boucher 2017).

3.2 Microplastic Sampling

Most of the microplastic in marine environment occurs either in the surface water or sediment bed (Woodall et al. 2014; Enders et al. 2015). However, subsurface water has also been sampled in order to identify any MP particles. Here, we are going to discuss some of the instruments used to sample surface and subsurface water for microplastic identification and quantification. These can either non-discrete or discrete sampling instruments.

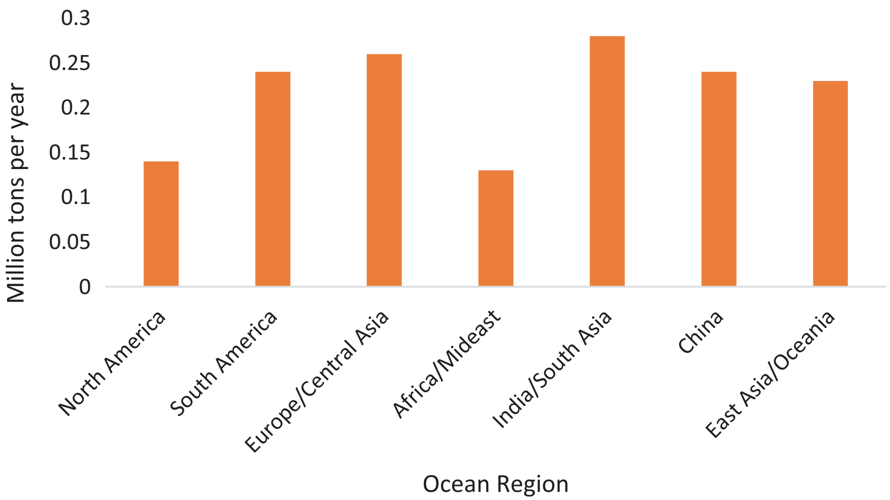

Fig. 3.1 Induction of MP into world oceans (Boucher 2017)

3.2.1 Non-discrete Sampling Instruments

3.2.1.1 Nets

Nets are used when sampling of large volumes of water is required. They can be used for surface, subsurface, and bottom water layer (Kang et al. 2015; Lima et al. 2014). Common net size changes from 20 to 500 μm. These are usually deployed from boats and ships. A flowmeter is also attached that gives accurate measurement about the water area being sampled (Norén 2007). If nets are not provided with a flowmeter, distance between starting and ending points is calculated from the size of net opening and transect length (Eriksen et al. 2018). Most common size for the net is 333–335 μm. A smaller-sized net can result in the coagulation of particles (Sutton and Sedlak 2017; Löder and Gerdts 2015). Sampling time usually may vary from 3 to 240 min., and towing speed may vary from 1 to 5 knots (Green et al. 2018; Eriksen et al. 2018; Syakti et al. 2018).

Manta trawls/nets are useful while sampling surface water microplastic (Masura et al. 2015; Free et al. 2014; Eriksen et al. 2013). A manta trawl is named such because of its resemblance with a manta ray (Moore 2003). It can be described as a metal box with wings and open ends. One end directs water into it where the water passes through a net having fine mesh of about 300 μm in order to trap MP larger than 300 μm (Sutton and Sedlak 2017). ***Neuston net*** is also used for sampling large volumes of surface water. *Plankton net* is used to sample medium volumes of water, and it can also be used for sampling water column. Mesh size for plankton nets varies from 200 μm to 400 μm (Murphy et al. 2016).

Smaller mesh size is usually beneficial in collecting smaller MP particles, but it results in the clogging of the sieve. In order to resolve this, sampling frequency can be increased with smaller volumes of water being sampled (Hidalgo-Ruz et al. 2012). After sampling, net is rinsed with decontaminated water, whereas microplastic particles are collected and sometimes preserved in a glass vessel. Net can be reused for next sampling process (Cutroneo et al. 2020).

3.2.1.2 Pumping Systems

These devices are useful for sampling microplastic from subsurface layers of water. They contain different kinds of pumps and are less commonly used as compared to nets (Zobkov et al. 2019; Setälä et al. 2016; Lusher et al. 2014). These can be used for long-duration sampling, i.e., from minutes to several hours for the same transect (Lenz et al. 2015). Winch can be used to lower the pump toward sides or the end of the boat or ship (Setälä et al. 2016; Ng and Obbard 2006). They are also equipped with filters and/or sieves of different sizes that help in collecting different sizes of MP particles (Cai et al. 2018; Desforges et al. 2014). After sampling, filter can be used for a direct observation and sieves are rinsed with decontaminated water, whereas microplastic particles are collected and preserved in a glass vessel (Picó and Barceló 2020).

3.2.2 Discrete Sampling Instruments

Discrete sampling devices include bottles like common bottles, Niskin bottles, Friedinger bottles, jars, integrated water sampler, bucket, etc. These can be used at different depths, and a precise and well-known volume of water can be sampled by using them. They also limit loss of fibers (Cutroneo et al. 2020; Dris et al. 2018; Crawford and Quinn 2017). In addition, sieves made of stainless steel and rotating drum samplers can be used for the sampling of surface microlayer (Campanale et al. 2020).

3.2.3 Limitations

Although there is a wide variety of sampling methods for microplastic, there exists a lack of proper guidelines and standards. Different mesh sizes make it difficult to compare data from different experiments in particular. Another problem is contamination of MP particles at almost all steps from sample collection to the identification. MP can be contaminated from the atmospheric deposition, laboratory substance, and/or clothes of the personnel handling samples. Different protective measures have been suggested by the experts in this regard which include preparation of sample and laboratory blanks, protection from operators' clothes, and protection from the air particles. Anyhow, no consensus has been developed among different studies (Mendoza and Balcer 2019).

3.3 Extraction of Microplastic

After sampling and pretreatment, microplastic particles are extracted from the samples using different chemical solutions so that organic substances and other impurities are removed, and these samples can further be used for the identification of microplastic. For performing FTIR, 30% hydrogen peroxide (H_2O_2) is added to the solution. Other oxidizing agents can also be used like hydrochloric acid (HCl), nitric acid (HNO_3), potassium hydroxide (KOH), and/or some oxidative enzymes (Cole et al. 2014; Nuelle et al. 2014).

3.4 Identification of Microplastic

Renner et al. (2018) analyzed different studies published from 2015 to 2017 and stated that a vast majority of the researchers had been identifying MP particles through visualization (through naked eye or microscope) in their experiments. Presence of MP particles is evident from their unnatural colors, like bright blue or multicolors and an unnatural shape like sharp ends of fragments or perfect sphere, etc. (Perren et al. 2018). However, there is always a chance of misconception since MP particles can be mistaken for other substances like paint particles, ceramic flakes, fish scales, and even fly ash. Highlighting physicochemical properties through different tests, like staining of nonplastic and natural particles (Ziajahromi et al. 2017) and heating above 100 °C (fiber's hot needle point), can help minimize such mistakes (Zhang et al. 2018; Roch and Brinker 2017). Visual techniques have also been subject to evolution with the passage of time in order to count smaller MP particles. Fluorescence microscope, dissection microscope, stereomicroscope, and scanning electron microscope are in use today (Syakti 2017). Visual counting with the use of stereomicroscope is a very well-established method despite of the fact that it is time-consuming and human error can be induced while using it. Anyhow, visual inspection alone is not enough in order to have clear and reliable information regarding MP characterization. Therefore, new and efficient technology is being used for physically analyzing the particles (Leslie et al. 2017; Mintenig et al. 2017).

Different vibrational techniques have been explained below that offer valid methodologies of MP count.

3.4.1 Fourier Transform Infrared (FTIR) and Raman Spectroscopy

These two techniques are most widely used nowadays because of their nondestructive method of sample identification. After FTIR and Raman spectroscopy analysis, samples can further be used for other analyses (Hermabessiere et al. 2018). Another benefit is that these techniques require small amount of sample. FTIR can identify a polymer at a resolution of 10 μm, whereas Raman spectroscopy at 0.5 μm. This is done through comparing the infrared spectra of polymer of the unknown sample with that of a known one. Data comparison algorithms are used to match the sample spectra with the spectral libraries. Raman spectroscopy and FTIR use vibrational spectra of the polymers which are different for each polymer (Picó and Barceló 2020).

Microscopic and imaging versions of FTIR and Raman spectroscopy are most frequently used techniques nowadays (Picó and Barceló 2020). The Attenuated Total Reflection (ATR) technique is preferable with Fourier Transform Infrared (FTIR) Spectroscopy because it works well for thick samples of microplastics, but it is not so efficient because of its relative insensitivity and inability to detect smaller MP particles (Mendoza and Balcer 2019).

3.4.2 Pyr-GC MS

Nuelle et al. (2014) has demonstrated the use of Pyrolysis-GC/MS. They analyzed the product of thermal degradation after 60s pyrolysis at 700 °C and transferring into GC/MS at 350 °C. The identification is done by comparing degradation product with a common standard. This technique is not only beneficial in indicating the presence of a plastic polymer but is also efficient at detecting organic plastic additives' interference (Fries et al. 2013). However, a disadvantage of using this method is that the analyzed material is completely destroyed (Syakti 2017). Pyr-GC MS coupled with differential scanning calorimetry is also increasingly being used for the characterization of MP (Picó and Barceló 2020).

3.4.3 Remote Sensing Technology

Remote sensing technologies for monitoring MP pollution in oceanic environment have been used in the recent years. The spectrum for the absorbance and reflectance by plastic falls in the near infrared (NIR) region of the light spectrum. So the ***NIR cameras and spectrophotometers*** can do the work of identifying and, hence, sorting plastic items. A limitation, however, is that surface water can absorb spectra from these instruments. So the MP particles present below surface waters will not be identified by these instruments. Another limitation with NIR is the low-resolution images produced from these techniques. This issue is proposed to resolve by reducing the distance between plastic items and the instrumental cameras. This can be done using automatic vehicles which are light and agile as compared to boats, ships, etc. and flow near surface as well as under surface (Mitchell 2015).

3.5 Monitoring Through Marine Animals

Microplastic content in marine water is also determined by determining the amount of MP particles ingested by the marine animals. Fish community, for instance, have been analyzed at different stages (Halstead et al. 2018) from direct capturing in the sea or ocean (Lusher et al. 2014) to the fish being sold in fish market (Rochman et al. 2015). Even fish larvae have been examined for MP content present in them (Steer et al. 2017). Other animals have also been targeted in the recent research experiments. Examples include crab, zooplankton (Cole et al. 2014), and lugworms and mussels (Van Cauwenberghe et al. 2015). Another significant example of MP monitoring through marine animals is the determination of plastic ingestion by *Fulmarus glacialis* or northern fulmars. It began in 1980s, and the plastic levels in the animal are used as a measure of the accumulation of plastic in "European Coastlines and North Sea for OSPAR Ecological Quality Objective on marine litter" (Van Franeker et al. 2011). As a principle for micro-invertebrates and fishes, one should perform data collection regarding animal's sex, body weight, length, girth, etc. prior to the dissection for the sake of isolating gastrointestinal tract for the analysis. Fish liver weight has also been used for the calculation of hepatosomatic index.

Monitoring of MP in marine animals includes removing digestive tract of the subject animal and then flushing the stomach with the shearwater so that the stomach contents including any food and MP particles are rushed out. These contents are then dried and stored prior to microplastic extraction. Sample storage is usually done by using 70% ethanol and 4% formaldehyde. Since some polymers can undergo damage with these solutions, an alternative is simply freezing the samples (Lusher et al. 2017).

3.6 Proposed Solution to Microplastic Pollution

A number of bacterial species are capable of degrading plastic particles. According to Singh et al. (2016), Pseudomonas species, Staphylococcus species, and Bacillus species, which were isolated from soil, showed excellent biodegradation of polyethylene. Asmita et al. (2015) reported that some microbes, viz., *Staphylococcus aureus, Pseudomonas aeruginosa*, some bacillus species, *Aspergillus niger,* and *Streptococcus pyogenes,* from soil were capable of biodegrading polystyrene and polyethylene terephthalate. Mor and Sivan (2008) observed *Rhodococcus ruber* can also degrade polystyrene through the formation of an effective biofilm. Caruso (2015) has reported biodegradation of polyvinyl chloride (PVC) by *Pseudomonas putida*. These and many other microbes can be grown in the laboratories and then applied to the MP particles in a controlled environment in order to decrease pollution level in the environment (Auta et al. 2017).

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Chapter 4
Polymer Types of Microplastic in Coastal Areas

Sedat Gündoğdu

Abstract The tendency for plastic leakage into the environment is increasing, and researchers struggle to detect the increase of plastic particles in marine environments. However, this situation raises a heated debate about the fate and final destination of missing plastics. The main axis of these discussions is whether the polymer types of plastics are also the determinants of the fate of plastics. It is necessary to know the polymer types of microplastics in all marine environments to understand whether this is so. Most of the studies conducted in this context examine microplastics in sea surface water and on the seabed. Although the highest number of microplastics are found in the seabed and the sea surface water, various studies emphasize that microplastic concentration in coastal ecosystems also increases. The major factor that determines the extent of microplastics in coastal environments is their density and polymer types. Therefore, it is possible that different polymer types of microplastics can be found in different marine compartments depending on their density. This chapter evaluates the presence and diversity of some of the produced microplastics in coastal areas. It can be said that the coastal environments are the main accumulation areas of microplastics, especially for types such as polypropylene and polyethylene, which have the highest production rates.

Keywords Coastal ecosystems · Marine plastic debris · Microplastics · Polymer type

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4.1 Introduction

Plastics are lightweight and inexpensive and have a wide range of uses. This has made them a material produced around 400 million tons globally in 2019 (PlasticEurope 2020). As a result of this increased production capacity, there has been an exponential increase in the production of these materials since the 1950s, especially with the increasing trend toward the convenience and cheapness of plastic in producing end-user plastic consumer products. Almost half of all plastics produced have been produced since 2010.

The polymers used to produce end-user plastic products can be grouped as thermoplastics, thermosets, and elastomers. Although there are thousands of plastic polymer types, thermoplastics dominate the market. In general, polyethylene (PE)-, polyethylene terephthalate (PET)-, polypropylene (PP)-, polystyrene (PS)-, and polyvinyl chloride (PVC)-type plastics are the most common thermoplastics in the market. Polyurethane (PUR) is the most widely used thermoset plastic polymer type. These six types of polymers account for ~80% of the total production of plastics (PlasticEurope 2020). Fibers also have an important share in plastic production. Approximately 15% of the total synthetic polymer produced consists of fibers (e.g., polyester, acrylic).

Additionally, most consumer plastics are also produced as a mixture of polymers and various additives. It is necessary to understand their main uses in the global market to understand the most common polymers used to manufacture plastic products. This can also give insight into the exact sources of plastic litter. The industries where the most used plastics are by far have been the packaging and building-construction industry. The third-largest end-use market is the automotive industry (Table 4.1) (PlasticEurope 2020).

Although plastic has taken place in all life and provides many advantages, it has been described as a common and persistent contaminant in aquatic environments in recent years. The increase in plastic use also causes a significant increase in the amount of post-consumption plastic waste. For instance, it is estimated that by 2025, the global urban population will generate more than six million tons of solid waste per day (Lebreton and Andrady 2019). Although the proportion of plastics in solid waste varies according to the regions' socioeconomic characteristics, it is accepted as 10%. It is estimated that the total amount of plastic waste generated in 2060 will exceed 200 million tons per year (Lebreton and Andrady 2019). It was estimated that around 19 to 23 Mt. of plastic waste entered aquatic ecosystems in 2016 globally (Borrelle et al. 2020). Hence, plastic can now be found in all ocean gyres, pristine environments, polar regions, and deep-sea sediments. In the marine environment, plastic is known to break down to much smaller sizes. Multiple groups and researchers have proposed standardized size categories for plastics that have become smaller sizes. These categories are macroplastics (>25 mm), mesoplastics (5–25 mm), microplastics (MPs 0.001–5 mm), and nanoplastics (<0.001 mm), respectively (GESAMP 2019). Cozar et al. (Cozar et al. 2014) state that millimeter-sized plastic pieces are predominantly dominant in floating plastic waste.

Table 4.1 The resin type of common plastics and their applications (adapted from (PlasticEurope 2020) and Wikipedia)

Name of polymer (acronym)-density	Application area
Acrylnitril-butadien-styrol-copolymer (ABS) 1.060–1.080 g/cm^3	Piping systems, musical instruments, automotive components, and household and consumer goods
Ethylene vinyl alcohol (EVOH) 1.17–1.21 g/cm^3	Food packaging and medical applications
Ethylene-vinyl acetate (EVA) 0.92–0.94 g/cm^3	Biomedical engineering, equipment for various sports, packaging, textile, and redispersible powders in plasters and cement renders
Polyamide (PA) 1.08–1.19 g/cm^3	Rope and similar applications, seat belts, parachute fabrics, fishing, welding clothes, sporting goods as composite materials, cargo and marine, car tire manufacturing, transmission belts, and military applications
Polyacrylonitrile (PAN) 1.1–1.15 g/cm^3	Fibers in hot gas filtration, sails, outdoor, and fiber-reinforced concrete
Polybutylene terephthalate (PBT) 1.3–1.4 g/cm^3	Automotive construction as plug connectors, housings in electrical engineering, and showerheads
Polycarbonate (PC) 1.14 g/cm^3	Electronics, construction, automotive and aircraft components, and medical applications
Polychlorotrifluoroethylene (PCTFE) 2.1–2.2 g/cm^3	Water repulsion, chemical stability, and electronic components
High-density polyethylene (PE-HD, HDPE) 0.94–0.96 g/cm^3	Packaging, rope production, disposable suits; nonwoven fabrics, mailing envelopes, flexible pipes, chairs, outdoor use stools, toys and playground applications, bottle crates, trays, plastic bags, and general-purpose containers
Low-density polyethylene (PE-LD, LDPE) 0.91–0.93 g/cm^3	
Polyethylene terephthalate (PET, PETE, polyester) 1.24–2.3 g/cm^3	Textile, packaging, construction, single-use products, and medical application
Polylactic acid (PLA) 1.23–1.25 g/cm^3	Medical implants, packaging, agricultural, and detergent coating
Polymethyl methacrylate (PMMA) 1.17–1.2 g/cm^3	Transparent glass, redirection of daylight, implants, and aesthetic uses
Polyoxymethylene (POM) 1.41–1.61 g/cm^3	Electrical engineering, automotive industry, railway applications, medical applications, food industry, furnitures, constructions, and instrumentations
Polypropylene (PP) 0.83–0.90 g/cm^3	Packaging, clothing, medical, household, piping, food industry, construction, textile, agricultural, and single-use items
Polystyrene (PS) 0.96–1.05 g/cm^3	It is widely used as heat insulation material in thin-walled containers, cooling systems, pipe foam, rubber, various auto parts, panels, and plastic parts of electronic devices. It is frequently used in disposable cups
Polytetrafluoroethylene (Teflon-PTFE)	Aerospace engineering, computer applications, cables, gears, slide plates, seals, gaskets, and bushings

(continued)

Table 4.1 (continued)

Name of polymer (acronym)-density	Application area
Polyurethane (PU, PUR)1.2 g/cm^3	Refrigerators, freezers, automotive seats, bumpers, steering wheels, insulation materials, computer and telecommunication equipment, underwater cables, seat, mattress, carpet pad, sandwich system panels, operating table, hospital bed, wound threads, catheters, packaging, flooring, and adhesives
Polyvinyl chloride (PVC) 1.3–1.7 g/cm^3	Profiles, floor and wall coverings, roofing plates, building products such as swimming tanks, pipes, and fittings for multiple sectors such as water irrigation, sewer transportation, energy and communication, packaging, pharmaceuticals, labeling, cables in automobiles, blood bags, blood transport tubes and surgical gloves, garden hoses, shoes, inflatable pools, and tents
Stiren-akrilonitril (SAN) 1.06–1.1 g/m^3	Electrical/electronics, automotive industry, optical device lenses, medical devices, souvenirs, stationery industry, household items, furniture, water treatment product reservoirs, and hydraulic parts

Coastal zones home to approximately half of the human population. This makes the coastal zone a hot spot for both MPs pollution (Cole et al. 2011). Direct sources of nearshore MPs include land-based sources, marine activities, river discharge, and potentially atmospheric fallout. Researches have shown a strong correlation between nearshore MPs concentrations and coastal populations (Pedrotti et al. 2016; Zhang 2017; Zhao et al. 2015). Numerous studies have shown that the highest concentrations of MPs were found in coastal areas near harbours, cities, and industrial sites (Faruk Çullu et al. 2021; Gündoğdu et al. 2018; Tunçer et al. 2018). It is indicated in recent studies that a high level of MPs has been observed in subtidal and marine sediments (Gewert et al. 2017; Kor et al. 2020), intertidal plains (Blumenröder et al. 2017), mangrove habitats (Maghsodian et al. 2021), and salt marsh habitats (Piarulli et al. 2020). These areas can be considered important sinks for MPs.

The estimated plastic concentration marine environment is based on the data collected using a floating net (e.g., Manta, Neuston) with 333 μm mesh size. Van Sebille et al. (2015) estimated that the concentration of MPs in 2014 ranged from 15 to 51 trillion pieces, weighing 93,000–236,000 tons, with more than 90% of observations collected using a floating surface net with 333 μm mesh (Lindeque et al. 2020). According to Conkle et al. (Conkle et al. 2018), >80% of studies only investigate MPs larger than 300 μm. Naturally, MPs smaller than 300 μm (microbeads, synthetic microfibers, tire wear particles) will not be present in such datasets. Therefore, the number of MPs in marine ecosystems may be higher than expected. Selective sampling nets used in plastic pollution research can affect plastic pollution predictions and underestimating the polymer-type composition of plastics. MPs are made up of polymers, and their distribution depends on the physicochemical composition. Therefore, studying both the morphological and chemical structure allows us to understand the horizontal and vertical distribution patterns. If the sampling methodology cannot fully represent the shape and size distribution, it is impossible to understand the actual distribution of MPs. For instance, tire wears, textile fibers, and microbeads, most of which are smaller than 300 μm, cannot be sampled by sampling with a size-selective manta net. Similarly, if the sediment sampled from sandy beaches is not sieved with smaller mesh size sieves, the sampled particles will be relatively large.

4.2 Polymer Composition of MPs in Coastal Ecosystems

Previously published studies confirm that PE, PP, PET/polyester, and PS are the most common polymer types in marine ecosystems (Table 4.2). These polymers also accounted for around 65% of global plastic production in 2019 and are commonly used as everyday use products. Erni-Casssola et al. (2019) stated that only the sampling zone could be a significant explanatory variable explaining variability in polymer-type prevalence among the multiple moderators considered. This is directly related to the density of the polymers (Table 5.1) and coastal transport, wave direction, vegetation, and tidal regime.

As it can be seen in Table 4.2, various researchers have reported different polymer types for various environment.

PE and PP are the most commonly reported polymer type in almost all coastal ecosystem types. It is clear that this is related to the extensive production of these two types of polymers. Although polymer density determines which plastics can be found in which ecosystem at what level, it cannot be said with certainty that there is such clear evidence. For instance, low-density polymers, such as PE and PP, showed a high presence in sediment and surface water in Australia (Su et al. 2020). Similarly, PS MPs, whose density is higher than seawater and other polymers, can be found widely in surface water and sediment. For instance, according to Jang et al. (2020), PS MPs are more abundant than other polymer types in seawater, sediment, and polychaetes from the aquafarm site located in the southern part of South Korea. Although there is a possible relationship between polymer density and particle suspension in the marine environment, there is an apparent lack of consensus regarding polymer density's influence on the vertical distribution of MPs in seawater.

In conclusion, undoubtedly, it can be said that the concentration and prevalence of MP polymer types in coastal environments vary according to habitat type, oceanographic conditions, geographical location, and sampling methodology. Although all polymer types can be found at any location, it would not be wrong to say that there is a general trend in the relatively higher abundance of the four common polymers (PE, PP, PET/polyester, and PS) in different sampling locations. However, as a result, it can be said that PE and PP are the most common types of plastic polymers in coastal ecosystems.

Table 4.2 Polymer compositions of extracted MPs sampled from different parts of coastal ecosystems

Study area	Sampling environment	Number of MPs	Polymer types and % (if reported)	Reference
Australia (Port Douglas, Busselton Beach), Japan (Kyushu), Oman, United Arab Emirates (Dubai), Chile (Vina Del Mar, Punta Arenas), Philippines (Malapascua Island), Portugal (Faro), Azores (Ponta Delgado), USA (Virginia, California), South Africa (Western Cape), Mozambique (Pemba), the United Kingdom (Sennen Cove)	Sandy beaches	2 (Australia) to 31 (Portugal, UK) fibers per 250 mL of sediment	Polyester 56%, acrylic 23%, PP 7%, PE 6%, and PA fibers 3%	Browne et al. (2011)
Slovenian part of the Northern Adriatic	Surface waters	406 items/m^2	PE >80%, others 6%, and unidentified 14%	Gajšt et al. (2016)
Bohai Sea, China	Surface waters	0.33 items/m^3	PE 51%, PP 29%, PS 16%, PET 3%, and others 1%	W. Zhang et al. (2017)
Persian Gulf, Iran	Littoral sediment	284.4 items/kg	PET 41%, PE 31%, nylon 16%, and others 12%	Naji et al. (2017)
Eastern Mediterranean coasts of Turkey	Surface water and fish samples	Surface waters: 16339 to 520,213 item/km^2 Fish: 2.36 items/individual	LDPE, PP, ABS, and chloroprene	Güven et al. (2017)
Stockholm Archipelago, Baltic Sea	Surface waters	0.42 items/m^2–0047 items/m^2	PP 53%, PE 24%, and PS 5%	Gewert et al. (2017)
Xiamen Coastal Areas, China	Surface seawater and surface sediments	103–2017 particles/m^3 in surface seawater and 76 to 333 particles/kg in sediments	PE 50.4%, PP 28.7%, cellophane 7.8%, PET 5.2%, PU 4.3%, and PS 3.5%	Tang et al. (2018)

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Table 4.2 (continued)

Study area	Sampling environment	Number of MPs	Polymer types and % (if reported)	Reference
Changjiang Estuary, China	Surface waters	23.1 items/100 L	PE 82.4%, PP 9.1%, PVC 6.5%, and others <3%	Xu et al. (2018)
Korean Coastal Waters	Surface waters and water column	871 items/m^3	PP and PE predominated	Song et al. (2018)
Mersin Bay NE Levantine coast of Turkey	Surface water	539,189 items/km^2 and 7,699,716 items/km^2	PE 55.2%, PP 26.9%, styrene/allyl alcohol copolymer 4.7%, and PS 4%	Gündoğdu et al. (2018)
Tuscany (Italy)	Water column and surface waters	0.26 items/m^3 water column; 41.1 g/km^2 and 69,161.3 items/km^2 of surface MP	PE >66%, PP 28%, PS 5%, and EVA and styrene butadiene 1%	Baini et al. (2018)
Kenya's marine environment	Surface water	110 items/m^3	PP and LDPE	Kosore et al. (2018)
Xisha Islands of South China Sea	Seawater, fish, and corals in three atolls	Seawater: 2–452 items/m^3; fish: 0–12 items/individual; coral: 1–44 items/individual	Seawater: Rayon fibers 64.8% and PET fibers 7.3%. In fish: Rayon fibers 31.2%, PET fibers 16.5%, PA fibers 11.9%, and PTFE granules 9.2%. Rayon fibers 32.3%, PET fibers 15.5%, and PVC fibers 14.0% in coral samples	Ding et al. (2019)
Lebanese coast (Eastern Mediterranean Basin)	Surface water, sediments, and biota (*Engraulis encrasicolus, Spondylus spinosus*)	Surface waters: 4.3 items/m^3, sediment 2433 items/kg d.w., *Engraulis encrasicolus* 2.5 items/individual, and *Spondylus spinosus* 7.2 items/individual	PP, PE, PS, PA, PET, PU, PVC, PLA, and ABS	Kazour et al. (2019)
Chabahar Bay, Gulf of Oman	Surface waters	0.49 items/m^3	PE and PP 69%	Aliabad et al. (2019)
Surabaya, Indonesia	Surface waters	0.49 items/L	PS 58.4% and others 41.6%	Cordova et al. (2019)

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Table 4.2 (continued)

Study area	Sampling environment	Number of MPs	Polymer types and % (if reported)	Reference
Northwestern Mediterranean Sea	Beach sediment	66 items/kg for the northern site and 58 items/kg for the southern site	Among fragments PP 17%, PE 15%, and PS 9%; among fibers PET, acrylic and polyacrylamide; among films PP 37%, PE 18%, and PS 10%	Constant et al. (2019)
The Mar Menor lagoon (SE Spain)	Sand and sediment	53.1 items/kg	LDPE 45.7%, HDPE 14.3%, polyvinyl ester 14.2%, and others 35.8%	Bayo et al. (2019)
Guanabara Bay, Rio de Janeiro, Brazil	Surface water	1.40 to 21.3 items/m^3	PE 81.7%, PP 16.20%, and unidentified 2.1%	Olivatto et al. (2019)
Jiaozhou Bay, the Yellow Sea, China	Water column	0.095 items/m^3 (and mesoplastics included)	PP 51.04%, PE 26.04%, polymerized oxidized material 7.29%, PS 5.21%, and others 11%	Liu et al. (2020)
Maozhou River within Guangdong-Hong Kong-Macao Greater Bay Area	Surface water and sediments	Surface water: 3.5–10.5 items/L; sediments: 25 to 360 items/kg	PE (water: 45.0%, sediments: 42.0%), PP (water and sediments: 12.5%), PS (water: 34.5%; sediments 14.5%), and PVC (water: 2.0%; sediments: 15%)	Wu et al. (2020)
Southwest Coast of India	Coastal waters, beach sediments, and marine fishes	1.25 items/m^3 in coastal waters, 40.7 items/m^2 in beach sediments, and 22 particles in GIT of 12 out of 70 species	PE 38.46%, cellulose 23.08%, rayon 15.38%, PET 15.38%, and PP 7.69%	Robin et al. (2020)
Chabahar Bay in the Oman Sea, Iran	Sediment and coastal water	Water 218 items/L; sediment 262 items/kg	PE 38%, PET 29%, and others 43%	Hosseini et al. (2020)
The northern part of the Oman Sea, Iran	Littoral sediment	321.21 item/kg	PE 39.4%, PP 25%, nylon 14.2%, and others 21.4%	Kor et al. (2020)

(continued)

Table 4.2 (continued)

Study area	Sampling environment	Number of MPs	Polymer types and % (if reported)	Reference
Persian Gulf, Iran	Neutonic	18 items/m^2	PE 48%, PP 28%, PS 17%, and others 7%	Kor and Mehdinia (2020)
Mediterranean Sea, Aegean Sea and Marmara Sea, Turkey	Fish (*Chelon saliens, Mullus barbatus mules, Mullus surmuletus, Trachurus mediterraneus, Lithognathus mormyrus*)	1.1 items/ individual	PP 26%, PE 21.9%, PET 8.2%, and cellulose 7.5%	Gündoğdu et al. (2020)
Along the Turkish Coasts	Mussel (*Mytilus galloprovincialis)*	0.69 item/ mussel and 0.23 item/g	PET 32.9%, PP 28.4%, and PE 19.4%	Gedik and Eryaşar (2020)
Bizerte Lagoon, Southern Mediterranean Sea, Tunisia	Surface waters	453.0 items/m^3	PE 51%, PP 25.1%, PET 14.2%, cellophane 3.3%, nylon 5.4%, and PS 1.1%	Wakkaf et al. (2020)
Todos Santos Bay, Mexico	Surface waters and sediments	Surface water: 0.01 to 0.70 items/m^3; sediment: 85 to 2494 items/0.1 m^2	PP, PE, ethylene-propylene-diene, nylon, T-elastomer, PA, PET, PA, and PVC	Ramírez-Álvarez et al. (2020)
Greater Melbourne Area and the Western Port Area, Australia	Surface waters and sediments	0.06 to 2.5 items/L in water and 0.9 to 298.1 items/kg in sediment	Water: PET 26.1%, PP 10.1%, PE 11.6%, PA 10.1%, and rayon 5.6%; sediment: PET 22.7%, PP 16.3%, PE 5.0%, and PA 9.1%	Su et al. (2020)
Island of Okinawa, Japan	Coastal surface water	Not reported	PE 10.94%, PP 0.61%, PVC 0.61%, PA (nylon) 1.52%, and PS 0.91%	Ripken et al. (2021)
Persian Gulf, Iran	Mangroves sediment and *Periophthalmus waltoni* fish	Beach sediment: 162 items/m^2; 8 out of 13 *P. waltoni* contained 15 microplastics	PS, PP, PET, LDPE, and nylon	Maghsodian et al. (2021)

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Chapter 5
Evaluation of Different Metrics to Study Microplastics as an Environmental Forensic Tool

Sidra Farooq, Abdul Qadir, and Nadia Jameel

Abstract Since the plastic frenzy began during the last century, the contamination of air, water, soil, and biota with microplastics, the degraded metabolites of plastics, has become a subject of environmental research. Resolving the issue requires the tracking of microplastics back to their sources. Environmental forensic approaches have the potential to tackle this ubiquitous challenge, but their application faces serious hindrances. The questions concerned with the manifestation of the problem to the transport, eventual fate, and source identification are still to a large extent unanswered. These issues are faced due to the poor understanding of interconnectedness of environmental metrics and lack of standardized data. This review is conducted with the purpose of assessing potential routes of microplastics by understanding the sink-source identification techniques and to determine the effectiveness of different metrics for the study of microplastics and their application as a tool in environmental forensics and other relevant fields.

Keywords Metrics · Evaluation · Microplastic · Environmental forensic

5.1 Introduction

Plastic, the most revolutionary material of the past century, consists of a broad variety of synthetic or semisynthetic materials. They are organic polymers, extracted from monomers of mainly oil and gas (Cole et al. 2011). The use of plastic is expanding rapidly, essentially due to one of its key properties: durability. Plastic is

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undeniably everywhere, and due to its low cost, hydrophobic attributes, and lightweight, it replaced the conventional materials like glass, cloth, and metal (Andrady 2017) and became irrefutably an integral part of an array of inexhaustible applications (Cole 2016). The use of plastic stretches from our domestic household products to varied industrial and agricultural uses, including packaging, textile, consumer products, building, and construction. Owing to the longevity of this synthetic polymer, it is expected that the global plastic production will reach to a staggering 25 billion metric tons by 2050 (Wright et al. 2020).

While undoubtedly the use of plastic has assisted us with numerous social benefits, the environmental impact of such overwhelming use of this commodity is a subject of great concern to the scientific community around the globe, mainly due to the fact that the durability of this material makes it resistant to degradation (Cole et al. 2011). Disposal of plastic waste in environmental systems not only is aesthetically displeasing but contributes to a variety of health problems both for humans and animals. Assessing impacts of plastic debris in different environments is a cumbersome task and requires complete knowledge regarding exposure concentrations, types of polymers, exposed organisms, exposure rate and time, etc. (Rochman et al. 2015). But generally, impacts related to plastic waste can be classified into physical, chemical, and biological. Physical hazards pertain to the ingestion and entanglement posing a serious threat to freshwater and marine species (Frias and Nash 2019; Huang et al. 2020) and were brought into attention in the early 1990s when a number of documentaries were aired showing turtles, seabirds, whales, and dolphins in distress due to entanglement (Vermeiren et al. 2016). Biological impacts include ulcers, starvation (due to ingestion of debris), deterioration of health, and loss of feeding ability.

Hazards pertaining to the plastics visible to the naked eye are well documented, but behind the headlines, the major emerging threat is less visible to the naked eye and has less studied, but perhaps, far-reaching impacts. This less visible threat is the contamination of continental and oceanic environments with microplastics. These microplastics may result from the adding of microbeads in the cosmetics products or from mechanical or chemical degradation or weathering due to the air or UV rays of other plastic products, such as tires or textiles. A consensus on the definition of microplastics was reached recently, and the term was defined as particles <5 mm (Frias and Nash 2019; Hendrickson et al. 2018; Renner et al. 2018). Microplastics can enter the environment in the form of either primary or secondary microplastics (Laskar and Kumar 2019). Primary microplastics are intentionally manufactured microplastics used in hygiene or other domestic products commonly in the form of microbeads (Zhang et al. 2020) or may enter the environment as a byproduct of industrial emissions or as plastic dust from a range of plastic products (Laskar and Kumar 2019), while secondary microplastics are produced by disintegration of relatively larger plastics and include fibers from textile products (Zhang et al. 2020). Understanding the fragmentation process of plastics is vital to estimate the amount of plastic pollution in different environmental compartments. Fragmentation of macroplastics generally occurs due to disintegration by physical or mechanical forces, such as abrasion and wind, and weathering effects, such as by UV sunlight (photooxidation) (Zhang et al. 2020). These microplastics ultimately end up in different environmental sinks by traveling from one environmental compartment to another.

These ubiquitous pollutants (Alimba and Faggio 2019) are spreading globally at an alarming rate, and currently, many researches are focused on determining the extent of damage such tiny particles can cause (Laskar and Kumar 2019). When studying a synthetic molecule, it is crucial to determine its physical and chemical nature because it plays a vital role in determining the toxicity level. Microplastics are categorized into many different polymer types according to the chemical structure, the most common being polypropylene (PP) (the greatest in demand, 19.3%) (Schwarz et al. 2019), low-density polyethylene (LDPE, 17.5%), high-density polyethylene (HDPE, 12.3%), polyvinyl chloride (PVC, 10.2%), polyurethane (PUR, 7.7%), polyethylene terephthalate (PET, 7.4%), polystyrene (PS, 6.6%), and others (Li et al. 2021; Zhang et al. 2020). Plastics ending up in the environment can originate from either one or both ocean- and land-based sources (Andrady 2011). Microplastics are found to be abundant in the air (Dris et al. 2017; Klein and Fischer 2019; Wright and Kelly 2017), soil (Abrusci et al. 2011; Tourinho et al. 2019), and aquatic (Muthukumar et al. 2011; Stanton et al. 2019) systems. Oceanic and freshwater sediments contain highest diversity of polymers, thus confirming that most of the primary and secondary microplastic contaminants are deposited in freshwater sediments or keep floating at the surface of rivers or canals, ultimately adding to the oceanic pollution load (Kumar et al. 2021; Morritt et al. 2014).

5.2 Source, Sinks, and Pathways

Distribution of plastic debris across the environment depends upon a number of factors, such as meteorology, climate, and anthropogenic activities (Dris et al. 2016; Enyoh et al. 2019). In addition to general decomposition and improper disposal through other intentional or unintentional entry modes from domestic sources, agriculture, industry, or transports, microplastics gets deposited at a number of sinks, including biota, soil, and freshwater systems (streams, rivers, lakes, etc.) via various pathways, mainly air and freshwater systems, eventually accumulating in multiple receptors sites, viz., biota, sediments, aquifers, etc. According to Liu et al. (2020), leaves of terrestrial plants, irrespective of the species, serve as important temporal sinks of MPs. Author established 28% deposition of MP alone on the terrestrial leaves along with other natural materials.

Microplastic pollution has become an emerging concern during the last couple of decades globally (Abbasi et al. 2019; Prata 2018; Wright and Kelly 2017). It is known to have a far-reaching effect via accumulating in the fatty tissues of organisms and may result in biomagnification when traveling along the food web, eventually affecting human livelihood and health (Vermeiren et al. 2016). Human exposure to microplastics is mainly through food, drink, and air. Former two pathways have been studied extensively, while the latter is lesser explored (Vianello et al. 2019).

Microplastic pollutant is abundant in different environments. The effect of microplastics in marine environments have been extensively studied, and a lot of studies determine the adverse effects of microplastics on snails, crayfish, etc., but the continental environments, such as freshwater systems, soil, and air (Li et al. 2021), to date remain poorly explored.

Identification and characterization of pollutants and resultantly reaching at the source of pollution is an integral part of any environmental forensic analysis, in this case MPs. Sources of microplastics can be broadly categorized into four main types, viz., macroplastics, cleaning products, pharmaceutical industry, and textile. As discussed earlier, MP are divided into primary and secondary categories based on the mode of production, i.e., commercial production for the former and fragmentation of macroplastics for the latter. In addition, these particles can be released into the environment through point and nonpoint sources. Point sources include domestic wastewater, industrial and livestock activities, other commercial activities, and wastewater treatment plants (WWTPs), whereas nonpoint sources include releases of MP from stormwaters, forests, and agricultural lands. The general trend is noticeable and also supported by various studies that show the positive correlation between increased urbanization and MP levels in different metrics.

5.3 Role of Metrics in Environmental Forensics

Environmental forensics is a discipline that offers the investigators with the necessary tools and techniques for the identification of the source of the contaminants, to develop effective laws and policies for the abatement of any future contamination. An effective forensic analysis is based on available data (Browne et al. 2011). A credible metric provides that data either present already or is gathered through various techniques gives information about the current state of contaminants to be used for further analysis.

Air, water, and soil systems are trackers or metrics of a community's overall health information base and productivity. Ideal metrics are valid, quantifiable (by using available methods and techniques), simple, sensitive, reliable, and contemplative of the representing community (Jakubowski and Frumkin 2010). They can help to identify a certain problem, quantify it, define environmental priorities and goals, aid in policy developments, and serve as a foundation in generating data to compare different sets of environmental characteristics to monitor the developments made over time. Moreover, effective metrics may serve as a tool and aid in tracing back to the source of the pollution to determine environmental forensic analysis. Forensic analysts should have access to all the data to be able to analyze and assess that pathway of contaminants, so that desired actions can be implied, by determining the source of the contaminants.

Microplastics are a new challenge to an environmental forensic expert. However, to meet these challenges, the effectiveness of the aforementioned ecological units (Heink and Kowarik 2010) or metrics, working as an environmental forensic tool,

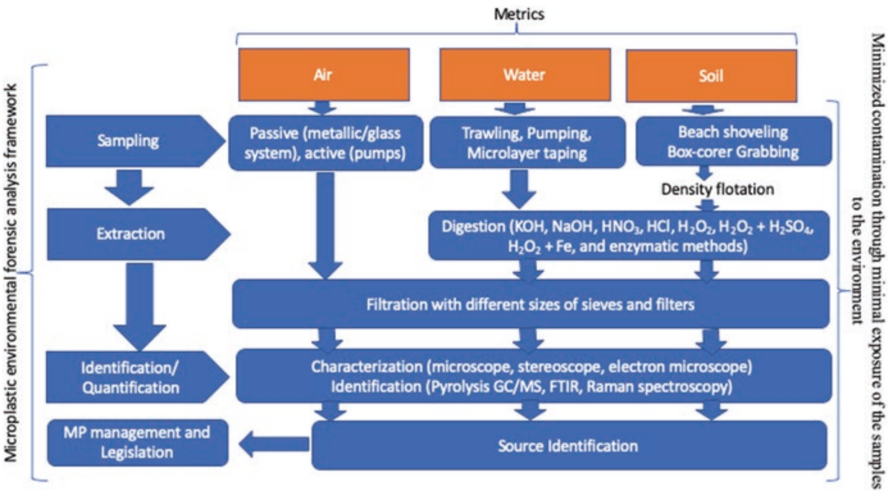

Fig. 5.1 Framework for analysis of microplastics in air, water, and soil

needs to be evaluated by assessing its relevance, quality, and ease of data availability, in all the stages of forensic process, i.e., sample collection and storage, sample analysis, tracking the source, final data evaluation (Thomassen and De Boer 2005), management, and legislation.

In the case of microplastics, the forensic investigation will begin with the evidence of damage. Given the ubiquity of this pollutant, it can have far-flung effects from a single source, both in time and in space. Adapted from Council (2014) and Woodall et al. (2015), Fig. 5.1 conceptualizes a proposed framework to perform environmental forensic investigations linked to microplastics, adapting top-to-bottom-to-top approach, evaluating the metrics along the way.

Methods and techniques for the study of microplastics are continuously evolving with increased research in this field (Miller et al. 2021). Tools and techniques for sampling, transportation, analysis, and further investigation are different for different metrics. To date, a multitude of methodologies have been undertaken, who have their respective advantages and disadvantages, but mostly, the success depends upon the expertise of data analysts, robust sampling protocols, and sensitivity to chemical structure (Huppertsberg and Knepper 2018). Detailed literature review was undertaken to analyze different techniques applied in the forensic process for all the three metrics (water, soil, air). Table 5.1 presents a summary of literature reviewed for this study showing the level of detection of microplastics in different metrics.

Microplastics have been extensively studied in marine waters for the past decades; however, recently, freshwater ecosystems are also being investigated for the extent of contamination of microplastics (Lu et al. 2021).

Table 5.1 A summary of major findings related to microplastics across different metrics in previous studies

Reference	Study area	Metric	Concentration of MPs	MP type	MP composition	Source identified
Vianello et al. (2019)	–	Air	1.7 and 16.2 particles m^{-3}	Synthetic fibers and fragments	PES	Textile clothes
Liu et al. (2019)	Shanghai	Air	4.18 n/m^3 (items per cubic meter of air)	Microfibers and fragments	PET, PE, and PES	Textile clothes
Turner et al. (2019)	North London	Soil	539 particles per kilogram of dried sediment	Fibers	PS, PAN, and PVC	Anthropogenic origin
Lu et al. (2021)	Multiple locations	Soil and water	1.2×10^{-3} to 5.42 × 105 particles/m^3 and 8.1×10^{-1} to 9.5 × 105 particles/kg	–	–	–
Evangeliou et al. (2020)	Multiple	Air	20 ng m^{3-3}	Fibers	–	Tire wear particles
Ashwini and Varghese (2020)	Kerala, India	Beach sand	120.85 items kg^{-1}	Fibers	–	Fishing net mending activities
Tsang et al. (2017)	Hong Kong	Water and soil	51–27,909 particles per 100 m^3 and 49–279 particles per kilogram	Fragments, lines, fibers, and pellets	PP, PE, and PP	–
Luo et al. (2019)	Yangtze Delta, China	Water	1.8–2.4 items/L	Fibers and polyesters	–	Multiple
Lindequeu et al. (2020)	Gulf of Marine (USA)	Water	3700 microplastics m^{-3}	–	BP and PE	Coastal activities
Lin et al. (2018)	China	Water and soil	379–7924 items·m^{-3} and 9597 items·kg^{-1}	–	PE and PP	Multiple
Murphy et al. (2016)	USA	Wastewater	19.67 (±4.51) MP/2.5	Microbeads	–	Personal care product

(continued)

Table 5.1 (continued)

Reference	Study area	Metric	Concentration of MPs	MP type	MP composition	Source identified
Löder et al. (2017)	–	Water	0.7 microplastics m^{-3}	–	PE, PA, and PES	–
Gatson et al. (2020)	–	Air	3.3 ± 2.9 fibers and 12.6 ± 8.0 fragments m^{-3}; mean ± 1 SD	Fibers and fragments	–	–
Prata et al. (2020)	Aveiro, Portugal	Air	6 fibers m^{-3}	Synthetic fibers	–	–
Lenz et al. (2015)	–	Water	$n = 1279$	Fibers and particle	PE and PP	–
Zhou et al. (2020)	China	Soil	571 pieces kg^{-1}	Films and fibers	PE, PP, and PES	Plastic mulching film
Feng et al. (2020)	Tibetan plateau	Soil and water	66.6–733.3 number/m^3, 20–160 items/kg, and 20–110 items/kg	Fibers	PP and PE	Different human activities, facility agriculture, and previous secondary industry
Scheurer and Bigalke (2018)	Swiss nature reserves	Soil	0.20 mg/kg^{-1}	–	–	Diffusion via aeolian transport

PET polyethylene terephthalate, *PE* polyethylene, *PES* polyester, *PP* polypropylene, *PA* polyamide, *PVC* polyvinyl chloride, *BP* biopolymers, *PS* polystyrene, *PAN* polyacrylonitrile

5.4 Water as Metrics for the Study of Microplastics as an Environmental Forensics Tool

A robust and credible forensic analysis necessitates precautionary measures in the sampling process, such as proper labelling and detailed geographical coordinates from where the sample is taken from and also the nearby roads, structures, etc. Such protocols while sampling for water samples in the reviewed studies were absent. While sampling from water samples, control samples from the nearby potential sources of contamination were not identified in any of the studies, which is a standard practice for the environmental forensic sampling, mainly due to the fact that the forensic investigation of micropollutants is still in the infancy stages and the standards are not evolved enough for all the metrics to investigate micropollutants in a methodological form (Gwinnett et al. 2021).

A range of types and sizes of microplastics are contaminating marine water and freshwater sources (Barnes et al. 2009). Where possible, it is mandatory to identify and quantify microplastics in water in order to assess the source. Collection of water samples is mainly being done by trawling, pumping, and microlayer taping and further subjected to filtration with various sizes of filters and sieves (Lindeque et al. 2020; Mai et al. 2018). Different digestion methods have been used in different studies, e.g., digestion with NaOH and KOH (Hurley et al. 2018) and other oxidizing agents or enzymes (Li et al. 2018). Enumeration of microplastics through visual counting is the most common method and can be done through Raman microspectroscopy (Lenz et al. 2015), stereomicroscope, fluorescent microscope, and scanning electron microscope (Qiu et al. 2016). In a study, almost 68% of visually counted samples MP ($n = 1279$) were identified through Raman microspectroscopy (Lenz et al. 2015).

Identification of microplastic source is important to assess the source of the pollution and is carried out by analyzing plastic shapes (fibers, spheres, fragments). Analysis for the identification of other types of microplastics can be done through various spectroscopic methods (Ng and Obbard 2006), such as gas chromatography (Renner et al. 2018), FTIR (Cai et al. 2017), and Pyr-GC-MS (Dümichen et al. 2017), etc. Most of the methods and techniques discussed are applicable and can be followed for attaining significant results. Notably, the applicability of a specific method depends greatly on the scope of the defined analytical question (Huppertsberg and Knepper 2018).

Municipal wastewater discharge is a big contributor to microplastic contamination. According to many studies related to urban water, fibers (Hendrickson et al. 2018) and microbeads (Napper and Thompson 2016) are the dominant pollutant types. Fibers may originate from synthetic textiles, and spherical shapes may suggest originating from personal health-care products (Lin et al. 2018). Effects of health-care products were studied in a first of its kind, conducted at wastewater treatment works, where microbeads were identified to be the most significant contributor to the overall microplastic load of the urban discharge stream. It was calculated to be adding 65 million microplastics in the discharging waters every day despite being treated (Murphy et al. 2016). Such microbeads can be traced back to surfactants, cleansers, makeup products, etc.

Groundwater was found to be the least affected by the microplastic contamination, mainly because the sieving process of soil retains most of the contamination as studied by Mintenig et al. (2019), determining the concentrations of multiple groundwater samples (0.7 microplastics m^{-3}) which helped in highlighting the role of soil in water purification. Additionally, microplastics are found to be more abundant in subtidal zones than in sandy beaches (Browne 2015). A study conducted by Luo et al. (2019) established higher abundance of MPs in freshwaters (2.4 item/L) than coastal systems (0.9 item/L), suggesting the diluting effect of water and the importance of proximity of polluting sources.

While microplastic contamination of water bodies and its effects and sources of the pollution is still an active area of research, a lot more research is needed to understand the pathways and sources of the plastic contamination. Major sources of freshwater pollution identified are industrial zones and urban littering and healthcare products (Feng et al. 2020).

5.5 Soil as a Metric for the Study of Microplastics as an Environmental Forensics Tool

Many studies show the importance of quality soil in determining health of a community (Zahran et al. 2013). A limited number of studies for the analysis of microplastics to determine the importance of soil as an environmental forensic tool were consulted for this review. Generally, there is a high partitioning of the contamination of MPs in soil as compared to the aquatic ecosystems. The processes of soil contamination are affected by a number of interactions, such as sampling sites, plantation, tideline, and depth of sample, and in turn change the environmental fate and toxicity of microplastics.

Sampling protocols of this metric include the collection through flotation, bench shoveling, and box corer grabbing (Mai et al. 2018), separation of MPs through centrifuge tube (Scheurer and Bigalke 2018), density separation (Sruthy and Ramasamy 2017), identification of polymer components through micro-Raman spectroscopy (Sruthy and Ramasamy 2017), and FTIR (Scheurer and Bigalke 2018).

A major contributor of soil MPs is sludge application to agricultural lands from wastewater treatment plants. MPs from textile washing accumulate in the wastewater treatment plants. These plants often are not equipped for the removal of MPs.

The forensic analysis indicated that majority of the microplastic analyzed in the studies determining MPs in the soil is sourced from the site/nearby area. A major contaminant-type polyester and polyamide is examined to be sourced by the degradation of textile fibers. Forensic investigation could draw useful conclusions regarding the pathways of pollution in this matrix. The fibrous microplastic was also found to be sourced from the fishing net mending activities in marine studies, being carried out near the shorelines (Ashwini and Varghese 2020).

5.6 Air as a Metric for the Study of Microplastics as an Environmental Forensics Tool

Humans are possibly exposed to microplastics through food, water, and inhalation. The first two modes have received quite some scientific attention, while the latter is lesser known (Vianello et al. 2019). An indoor microplastic contamination study was conducted by Vianello et al. (2019), determining that all samples analyzed were contaminated by MPs, with concentrations assessed between 1.7 and 16.2 particles m^{-3}.

Spatial and temporal distributions of MPs in air metric are largely poorly understood. In a study conducted in central London, atmospheric microplastic deposition was assessed via 1 L Duran bottles, and the samples taken were vacuum filtered, dried in an oven at 40 °C, and were subjected to the identification of MPs through florescence stereomicroscope, following FTIR analysis. Fibers were assessed to be the most significant MP type deposited at ranges between 510 and 925 microplastics/m^2/d. By setting velocities and assuming wind speed, distance of MP particles traveled was calculated, giving useful information about the source of contamination. Fragments and films were assessed to be the most significant nonfibrous MP type. Fragments likely originate from different recyclable plastic products while films could be driven by plastic bags, packaging, and foams (Wright et al. 2020).

Suspended atmospheric microplastics (SAMPs) can be studied through passive sampling, leading to variations of abundances. A study based in Shanghai used an active sampler to determine SAMP. Samples resulted in determining 67% of microfibers. PET, PE, and PES were determined to be major polymer types through FTIR analysis, paving way to determine the links between sources and sinks (Liu et al. 2019).

Air is one of the major tools to assess the MPs in forensic analysis and is also an important pathway for the deposition in other matrices, i.e., soil and water. To understand the contamination cycle, it is important to understand the concept of "MP cycle" (Enyoh et al. 2019), suggesting the interconnectedness of all the three matrices, but the evidence of interconnectivity is still inadequate. "Microplastic communities" is also an emerging concept which examines the differences (Mbachu et al. 2020), links, and types of MPs among various metrics.

5.7 Forensic Techniques to Investigate Microplastic Origin

While the extent of plastic pollution in the oceans is an example of ubiquity of the problem, where looking for answers as to the source of pollution is nearly impossible, terrestrial environments do provide some answers but are cumbersome and time-consuming and require resources.

To investigate the origin of MP, comparison between the shapes, sizes, and chemical structures of polymers offers valuable understandings. For instance, if the MP is secondarily sourced from a larger plastic product (PP, PS, PE), the debris will be irregularly fragmented. Spheres are more commonly sourced from personal hygiene products, whereas fibers may result from washings of textile products. Age of microplastic can be a determining factor in aquatic environments in relation to the origin. Considering the mobile behavior of MP in aquatic ecosystems, a smooth MP, suggests a local source, while a worn-out MP with a biofilm may suggest a distant source. Therefore, more research is required to collect forensic information regarding the frequency and distribution patterns for different shapes, sizes, and types of polymers and the possible entering pathways (Browne et al. 2011).

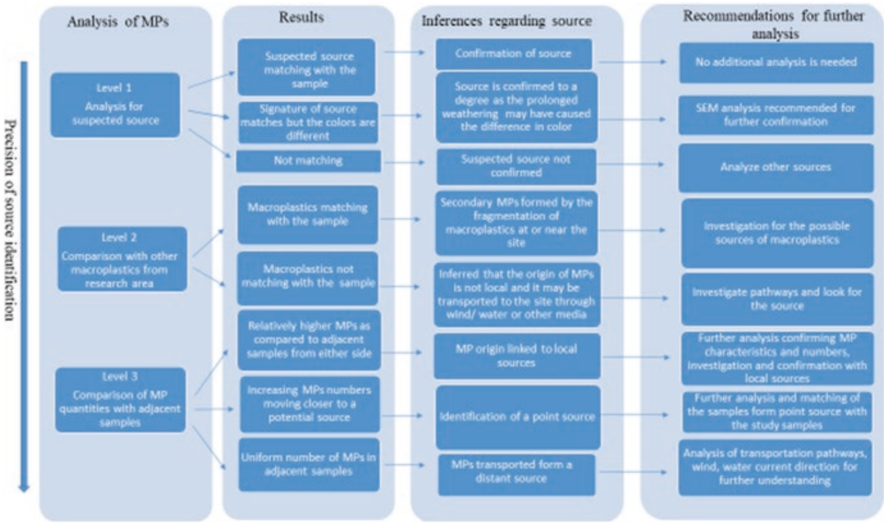

Fig. 5.2 Conceptual framework for forensic analysis of microplastics. (Adapted from Ashwini and Varghese (2020))

A recent study in 2020 at the Nattika Beach, India, proposed a three-level framework (Fig. 5.2) for the forensic investigation of microplastics in marine environments. Level one includes techniques if the source is suspected to be known. After the comparison, if results indicate a matching signature, or the similar physical and chemical characteristics, the source can be confirmed, and no further analysis will be required. Moreover, if the source is not suspected, level two proposes to compare the samples with macroplastics found around the study area. Either the samples will match or they won't, suggesting that the MPs in question were a result of fragmentation from the larger plastics for the former, and perhaps not a local origin for the latter, requiring further analysis. Level three proposes the comparison of the sample in question against the samples collected from the vicinity. If MP levels in the sample being analyzed are higher than the vicinity, it suggests that the source of MP is local, requiring investigation of local sources to confirm the findings. On the contrary, if the MP levels increase toward a river mouth or a drain outlet, then it is the source, requiring further water analysis, while the samples showing same levels of MPs regardless of the position suggest a distant source, requiring further analysis of wind or wave current direction (Ashwini and Varghese 2020).

Another technique used in a study carried out in Japan, to assess the source of MPs in road dust, was to analyze the plastic additives in the road dust samples. Plastics additives, viz., flame retardants and plasticizers, were traced back to the road markings, thus making them an important contributor of MPs in the road dust (Kitahara and Nakata 2020).

Moreover, the advancements in statistical techniques in recent years have aided the researchers in forensic analyses of the polymers. Assigning large sets of data produced by various spectral imaging techniques used by researchers, into groups for analysis and identification, has been carried out by using Principal Component Analysis (PCA), in various studies. It allows differentiation between the spectra of synthetic and natural origin, resulting in the enhanced visual accessibility by creating a two-dimensional image of the MP. Data of similar spectra are grouped and labelled accordingly by comparing it to the reference spectra. A lot of studies state the effectiveness of using PCA for the identification of polymers, albeit the unidentified spectra may hinder the analysis and limit the findings to the automated library search, proving that more research is needed to develop the reference library.

Scientific attention has diverted to the source identification in forensic analysis; however, further management and legal actions for the accused or guilty, whether at individual or organizational level, have still not been prioritized at local or national scales. Forensic framework application of environmental forensic science in the study of microplastics is still in its infancy stages and needs a lot of scientific attention and vigor to develop.

5.8 Microplastic Detection Applications in Different Fields

Detection of microplastics in the metrics of air, water, and soil sediments is an indicator of the quality of the respective environment and ecosystem health as a whole. Analyzing MPs in a given metric shows the level of plastic pollution in that zone and can contribute to determining the level of degradation of the environment. High levels of MPs in any given metrics have been associated with anthropogenic and hydrological dynamics. While the awareness regarding impacts of MPs is growing and in-depth analysis in various metrices is underway, which may help to a certain extent, numerous knowledge gaps are identified which hamper the rate of inclusion of MP analysis applications. Therefore, to understand the relevance of the level of MPs, information about hotspots of plastic uses and releases, identification of polymer types being used in a region, and transportation of MPs between multiple source-sink pathways need to be assessed on a broader and urgent basis. Such data compilation will prove to be valuable for a multiload of applications in various fields.

5.8.1 Environmental Management Systems

Inclusion of MPs analysis in different management systems, such as LCA (life cycle analysis) and others, may help in determining and comparing the overall impacts of various products in a robust and comprehensive manner. This may help in achieving the sustainability objectives of a specific industry by rejecting or accepting its particular products with the help of sustainability assessments including MPs analysis. For instance, LCA of textile apparels should include MP analyses, by calculating MP shredding rate for a number of washes, in order to generate a holistic impact factor (Henry et al. 2019). This may lead to increased awareness about the cradle-to-grave impacts around a particular product and ideally should result in informed decisions involved in eliminating microfibers use in the apparels by the industry and choosing environment-friendly, microfibers-free apparel by the consumers in general.

5.8.2 Agricultural and Geological Applications

In the last 5 years, soil has emerged as an important metric, and more focus has been given to the effects of MP in soil and other environmental compartments, which previously mainly centered around marine MP pollution and terrestrial littering of plastic. The quality of agricultural soil is dependent on many activities which add to the plastic burden of the soil including sludge application, mulching film, atmospheric deposition, quality of pesticides and fertilizers applied, etc. Toxicity determination of MPs is still in the infancy stages; however, movement of MPs across different trophic levels is well established (Kumar et al. 2020). In order to understand the soil and MP dynamics, complete identification and analysis of MPs in this metric will enable to understand the complex effects of MPs on the animals, plants initially, and across generations in the longer run.

Interestingly, MP detection can be used in the field of geology to establish a temporal and sedimentary outlook. The sediments are identified as major sinks for MP deposition, but the area of fluvial systems is yet unexplored. As established already in various studies, MPs have no known geogenic sources; thus, the deposition of these particles can be evaluated to investigate the time marker of Anthropocene. Building on this concept, Lechthaler et al. (2021) investigated the sedimentary layers of the fluvial plains of Inde river in Germany. After analyzing MPs from nine sites from the sample area and the sedimentation rates, the author proposes that MPs levels along a sediment layer not only provides morphological characteristics of human influence but also can be a time marker for the sedimentary deposition as a certain plastic polymer type has a corresponding time reference which can be identified; hence, the sediments were found to have a chronological layering of deposition. Polyethylene terephthalate (PET), a younger material, made in 1973, was detected in the upper layers, while older sediments had deposition of polyethylene (PE), which marks the beginning of plastic frenzy, patented in 1933. MPs detection in floodplains is still a neglected area, and considering they are temporal sinks, more efforts and resources should be allocated to further explore its potential applications.

5.8.3 Epidemiological Studies

Another vital application of MPs detection is its use in epidemiological studies. Even though solid evidence has not been established through exclusive studies on human subjects, initial data suggests that microplastics exposure to humans result in bioaccumulation in several tissues and organs, leading to various health hazards (Akanyange et al. 2021). Exposure routes to humans are analyzed to be, viz., inhalation, ingestion, and dermal contact. Abbasi and Turner (2021) in the first of its kind study, in the settings of Iran, attempted to calculate the human exposure by determining MPs on human hair, skin, and saliva. Author established that human hair accounted for the most MPs (>3.5 MPs per individual/day). Albeit there is not enough evidence of MP toxicity in humans, multiple studies have detected the accumulation of MPs in human stool and lungs. Prata et al. (2020) discussed the possible outcomes of MP toxicity in all biological systems, particularly humans, and acknowledged particle toxicity and chronic inflammations, to be caused due to the defense systems incapacity to eliminate synthetic polymers. Further research is required to identify numerous diseases and allergic reactions that may be a result from the MP exposure to humans. In order to understand the complexities of interconnectedness of MP and human exposure, MP analysis is essential. It will pave the way to help diagnose conditions that may result from MP exposure but are still unidentified.

5.8.4 Urban Development

Various studies indicate that the level of MPs is directly related to the urbanization and population (De Carvalho et al. 2021). Large river mouths have found to be the source of increased levels of MPs (Ashwini and Varghese et al. 2020). Various techniques provide an understanding of the sink-source pathways and help identify the source of the problem. The burden of rectification and legislation is a heavy one and falls on the people with power to make it a priority in order to assume rectification. Urban development strategies need to be formulated with a total consideration of point and nonpoint sources, and necessary measures should be adapted in the light of MP investigations.

5.8.5 Legislation

Various knowledge gaps identified through this review and others on the topic provide insights in this deep-rooted problem. The progress to understand the ubiquitous nature of MPs is underway, albeit slowly hampering the quantification and risk assessment. It is therefore assumed that without the formulation of a comprehensive risk assessment, a risk-based policy for the protection of human and natural ecosystems health cannot be formulated. To fill in the knowledge gaps and bring policy-level changes to eradicate this nuisance, it is imperative to prioritize research efforts for the development of forensic frameworks.

5.9 Conclusion

The field of environmental forensics is evolving in parallel to the realization of disastrous effects of microplastic contamination in natural systems. No peer-reviewed research has been carried out on this topic before. Thus, many important gaps in current knowledge were identified during the formulation of this book chapter.

Microplastics have been identified in different matrices, yet the correlation between different environments is not entirely understood. Major limitation regarding the current research is the lack of harmony and standard practices of methods and techniques used within the scientific community, halting the hominization and comparison between different studies due to differences in the reporting structure. Unavailability of data on the transport of microplastics from different sources, especially in the south Asian region, was a major deterrent in arriving at substantial conclusions in forensic analysis. Further detailed studies should be conducted, at an urgent basis, to assess multiple pathways of microplastic contamination between different sources and sinks, in order to develop a concrete set of databases at the local and regional scale, for future research in this justifiably significant area. Currently, we are bathing in the sea of plastics, and in order to completely eradicate this nuisance from our lives, we need to look into other suitable alternatives such as replacing synthetic plastics with biodegradable options.

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Part II
Environmental Occurrence

Chapter 6
Atmospheric Microplastic Distribution, Fate, and Behavior in Context to Pollution

Iffat Batool and Abdul Qadir

Abstract The use of plastic products is common in our day-to-day life due to its unique properties. But its mishandling and poor waste management lead to its accumulation in the environment. In the environment, it may degrade by different environmental factors such as photo-oxidation and thermal and biological degradation, resulting in particles with a size less than 5 mm called microplastics (MP). These particles are of major concern because they have been detected through different environmental compartments and pose a serious threat to the organism's as well as to human health. MPs are highly persistent and stable in the environment and have long residence time. The environmental pollutants may also adsorb on its surface, that may leach or desorb once into the living body. Furthermore, the presence of these particles in the environment leads to bio-accumulation and bio-magnification in different trophic levels causing adverse human health impacts. The main aim of this study is to highlight the sources, occurrence, and behavior of microplastics in the terrestrial environment. Additionally, this chapter focuses on the impacts of microplastics on human health.

Keywords Plastic pollution · Microplastics · Terrestrial Environment · Human health impacts

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6.1 Introduction

Nowadays, the use of plastic products has been extensively increased due to its unique properties like low cost, excellent resistance to corrosion, durability, lightweight, easy to deform and press, and outstanding insulator properties (Xia et al. 2020). The global plastic industry has begun since the invention of the first synthetic plastic in 1907 (Ritchie and Roser 2018). Plastic products have been massively used since the 1950s (Xu et al. 2020). Plastic encompasses a vast range of synthetic polymeric materials such as polypropylene (PP), polyvinyl chloride (PVC), polycarbonate, polystyrene (PS), nylon, and polyethylene (PE), which provide a diverse range of end products (Leal Filho et al. 2019). Globally the annual production of plastic is approximately 280 million tons; it has extensive application in the packaging industry, agriculture, medical treatment, and other household products (Rocha-Santos and Duarte 2015; Dehghani et al. 2017).

Plastic debris is abundant in the environment as a result of unsustainable use and poor management of industrial and domestic plastic waste (Merga et al. 2020). Polymers that are widely produced as plastic and found as plastic debris are polyethylene terephthalate (PET), PP, PS, PVC, and PE (Rocha-Santos and Duarte 2015). Plastic waste is a serious environmental issue because of its persistence, accumulation, and strong resistance to degradation (Xia et al. 2020). Plastic debris once released into the environment may stay for several years (Merga et al. 2020). Plastics are often resistant to breakdown due to their chemical inertness, and estimates for the full destruction of plastic trash in the environment vary from decades to millennia (Weinstein et al. 2016). Plastic trash continues to build in both the marine and terrestrial ecosystems due to its slow breakdown rate (Eriksen et al. 2014; Klein and Fischer 2019).

The plastic waste problem gets worse when it breaks down into smaller particles. Environmental conditions such as sunlight, temperature, and oxygen levels all play important roles in the breakdown of the plastic. The pace of degradation is mostly determined by its location (Weinstein et al. 2016). Once the degradation started, it may degrade over time and fragment into microscopic particles known as MPs (Wright and Mudway 2019). MPs are defined as plastic particles with a larger dimension smaller than 5 mm (Arthur et al. 2008), and are present in different shapes such as fragments, granules, fibers, foams, or film (Kwak and An 2020). MPs may originate from the fragmentation of large plastic debris or may deliberately or directly manufacture on a millimetric or sub-milimeter size (Cole et al. 2011).

In the recent decade, great attention has been paid to pollution caused by these tiny plastic particles (Dehghani et al. 2017). These particles have been found through almost all environmental matrices such as river and river bed (Hurley et al. 2018; Tien et al. 2020), water and sediments of the freshwater system (Rodrigues et al. 2018), urban river environment (Tibbetts et al. 2018), urban surface water (Wang et al. 2017), municipal wastewater treatment facility (Carr et al. 2016), dry and wet atmospheric deposition (Cai et al. 2017; Zhou et al. 2017; Abbasi et al. 2019), airborne particulate (Dris et al. 2017; Abbasi et al. 2019; Gaston et al. 2020), agriculture land (Piehl et al. 2018), remote area (Allen et al. 2019), and even Polar Regions (Peeken et al. 2018), glaciers (Ambrosini et al. 2019), and the world's deepest ocean (Peng et al. 2018).

MP pollution was named the second most significant scientific problem in the field of environmental and ecological research at the second United Nations Environmental Conference in 2015 (Xie et al. 2020). Several types of research are providing evidence of its presence in food (Barboza et al. 2018), sea salt (Kosuth et al. 2018), drinking water (Mason et al. 2018), honey, sugar, and beer (Liebezeit and Liebezeit 2013, 2014) and particularly seafood (Yang et al. 2019). Ingesting of these microscopic plastic particles can have a negative influence on the organisms that consume them, causing mechanical damage and inflammatory reactions (Wang et al. 2019).

Furthermore, these particles may act as carriers for other compounds adsorbing on their surface, such as environmental contaminants or plastic additives, which may leak out, causing exposure to dangerous substances (Schwabl 2020). There is scientific evidence of the presence of chemicals like polychlorinated biphenyls (PCBs), tetracycline, organochlorine pesticides (OCPs), polycyclic aromatic hydrocarbon (PAHs), dichlorodiphenyltrichloroethane (DDT), and other heavy metals such as Pb, Cd, Zn, Cu, etc. on its surface (Brennecke et al. 2016; Lo et al. 2019; Wang et al. 2021). The waterborne persistent pollutant and plastic additives are endocrine disruptors, which are capable of stimulating the hormone signal transduction pathways in target tissues and changing metabolic and reproductive outcomes; their transfer to animal tissues enhances their potential to damage (Galloway et al. 2017).

The current study aims to highlight the (a) dynamic nature and behavior of MPs in terrestrial environments, (b) sources of MPs, (c) ecological impacts of MPs on terrestrial organisms, and (d) impacts on human health.

6.2 Dynamic Nature of Plastic

Internationally MPs are categorized into two types, namely primary and secondary MPs. Particles, those that are particularly produced in the micrometer size range for various reasons, are primary MPs, such as those used in industrial abrasives for sandblasting, plastic pre-production (nurdles), or microbeads in personal care products such as body washes, face, and toothpaste (Nava and Leoni 2020). While, those virgin plastic pellets that are used as the precursor for manufactured plastic products are also attributed to primary MPs (Ivleva et al. 2017). Besides, the MPs that are derived from the degradation of larger plastic items under environmental conditions are secondary MPs (Nava and Leoni 2020).

In the environment, the degradation of plastics is slow and it may stay for many years (Merga et al. 2020). Plastics can ultimately degrade through different processes. The degradation mechanism is divided into two pathways: non-biodegradation and biodegradation. Photo-degradation, thermal oxidation, and hydrolysis are examples of non-biodegradation of plastics (Shen et al. 2019). The photo-oxidative degradation starts when the plastic is exposed to sunlight, resulting from the absorbance of high energy wavelengths of the ultraviolet (UV) spectrum (Weinstein et al. 2016). Plastic degradation due to UV radiation and microorganisms is highly dependent on the chemical components of the material as well as environmental factors such as temperature (Merga et al. 2020). Mechanical stress and degradation, mostly driven by UV light photo-oxidation, resulted in the release of polymers with less molecular weight such as monomers and oligomers, as well as the formation of polymers of decreasing size (Galloway et al. 2017). Once the degradation process has begun, it can continue further depending on temperature and thermo-oxidation processes, without further exposure to UV radiation as long as oxygen is present (Weinstein et al. 2016).

Plastic debris can disintegrate to smaller fractions via biodegradation, chemical weathering, or physical forces such as wind, wave action, and sandblasting (Hidalgo-Ruz et al. 2012). Soil microbes and other terrestrial organisms may accelerate the biodegradation of plastic (He et al. 2018). The microalgae growth on the surface of the plastic is crucial for the degradation process of plastic, either because it can degrade or because it protects polymers from UV radiation and photo-catalysis (Nava and Leoni 2020).

Furthermore, the degradation rate of plastic largely depends on its location. PE films and thermoplastics materials were shown to degrade at a slower pace when submerged in seawater than when in the air, most likely owing to less solar exposure, lower temperature, and lower oxygen (Weinstein et al. 2016). Depending on size, plastic debris is classified as macroplastic (≥25 mm), mesoplastic (5 to <25 mm), microplastic (1 to <5 mm), and nano plastic (1 nm to 100 nm) (Jambeck et al. 2015; Rezania et al. 2018; Fig. 6.1).

The physico-chemical reactions and interactions with the microbes result in the change of structure and mechanical properties as well as increases the surface area of the particle (Shen et al. 2019). Degradation nanoparticles have an extraordinarily

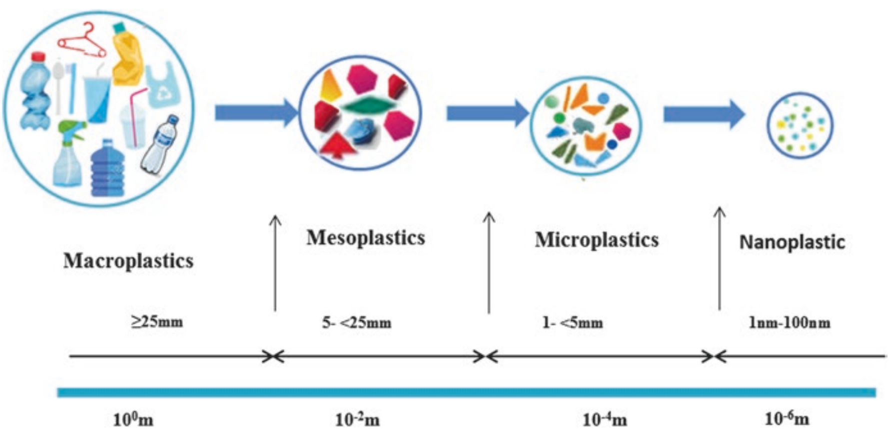

Fig. 6.1 Degradation of macroplastic to nano plastic

high surface area, such as a standard plastic shopping bag converted into 40 nm plastic particles, which have a surface area of 2600 m^2 (Mattsson et al. 2018). Plastic particles are frequently biodegraded outside of bacteria by the secretion of extracellular enzymes secreted by live microorganisms that can break down the polymer chain. This process generates nano-sized plastic particles with varying structures (Shen et al. 2019). The formation of nano-sized particles changes the chemical and physical characteristics of the particle and, consequently, its availability and biological impacts on the organism (Ferreira et al. 2019).

6.3 The Behavior of Microplastics in a Terrestrial Environment

The fate and distribution of indoor, as well as outdoor, MPs are influenced by several factors, which influence human exposure. The behavior and movement of MPs in the atmosphere is similar to those of particulate matter such as vertical pollution concentration gradient, wind speed and direction, precipitation, and temperature (Prata 2018). As cities develop and distances between cities and natural areas shorten, the collection of wind-dispersed plastic garbage and other litter is projected to increase (Rezaei et al. 2019). Other matrices are likely to be contaminated by airborne MPs, with an estimated 0.9–1.4 tons of airborne MPs deposited annually along the coastline of Yantai City, China (Zhou et al. 2017). Another important factor of the transfer of land-based MPs to the water bodies is runoff. Rainfall is the most direct and efficient means of causing surface runoff (Xia et al. 2020). According to Kaya et al.'s (2018) research, there was a substantial decrease in MP count in the samples taken after rainfall.

The inherent properties of MPs such as density shape and size can affect the transportation and dispersion pattern (Li et al. 2018). Each form of MP has a higher density than air, which has a density of roughly 1.225 g/L at sea level at 15 °C and decreases with increasing height (Peng et al. 2017). Lower density particles are lighter and can be easily transported by the air currents, polluting both land and aquatic environments (Rezaei et al. 2019). However, in unfavorable meteorological conditions such as low wind speed, MP clearance may be decreased, resulting in greater concentration exposure (Prata et al. 2020).

According to Allen et al. (2020), MPs may resuspend from the ocean's surface via sea spray, generate aerosols up to a few micrometers size and transported back to the terrestrial environment. Under normal conditions, micro- and nano-sized salt particles are expelled from the sea surface when breaking waves cause trapped air bubbles to rise to the surface and explode. When the unsupported surface of the bubble bursts, nano-sized particles are ejected and suspend in the air, ready for transport by wind (Allen et al. 2020). According to the model results, the distribution of MPs in the ocean may be anticipated using LaGrange tracking models that take into account current, wind-driven current, and horizontal diffusion (Li et al. 2020c). Bank and Hansson (2019) proposed a novel cycle as the "Plastic cycle"like other famous carbon or nitrogen cycle. The plastic cycle is the process through which plastics transfer between soil, air, and water via various paths; for example, light plastics are more easily transported in diverse systems, with the action of different factors leading to the spread of plastics to certain remote areas (Li et al. 2020b).

6.4 Terrestrial Sources of Microplastics

Plastic debris is frequently found in the environment as a result of unsustainable usage and improper management of industrial and residential plastic trash (Merga et al. 2020). According to estimates, about 4.8–12.7Mt of plastic garbage from land-based sources were discarded into the ocean each year (Xu et al. 2020). The terrestrial ecosystem is a greater "sink" of MPs than the ocean. Annual plastics discharged to land are estimated to be 4–23 times more than those released to the ocean (Choi et al. 2021). The principal sources of MPs in terrestrial environment are car, bus, and lorry tires, synthetic textile, building material, road marking paints, footwear, furniture, uncovered landfill sites, city dust, personnel care products, wastewater treatment facility, wind-blown debris, and rubbish discarded by anthropogenic activities comprising plastic bags, bottles, and boxes (Dalla Fontana et al. 2020).

6.4.1 Indoor Sources of Microplastic

With the increased global manufacturing of synthetic fibers, there is the worry that MPs produced from synthetic textiles will continue to pollute the environment (Cai et al. 2020). Microfibers are microscopic fibers of nylon, polyester (PES), acrylic, and other synthetic textile nature (Mishra et al. 2020). Only synthetic fibers account for more than 73% of global fiber usage, with polyamide (PA), polyolefin, acrylic, and PES being the most popular (Xu et al. 2018). Natural or artificial processed fibers are dominant in our daily lives and are used for a variety of household purposes such as bed linens, curtains and carpets, clothing, mattresses, chair covers, old interior paints, and upholstery (Dris et al. 2018; Mishra et al. 2019).

When compared to the outdoors, the concentration of MPs in the interior environment is considerably high (1–60 particles/m^3 and 1600–11,000 particles/m^2/day), which may be attributed to particle release from indoor sources and lesser clearance rates via dispersal processes (Zhang et al. 2020). Similarly, Dris et al. (2017) found a high concentration of microfibers in an indoor environment (1 and 60 fibers/$m^{3)}$ than outdoor environment (0.3 and 1.5 fibers/m^3). The deposition rate of microfibers in an indoor environment was between 1586 and 11,130 fibers/day/m^2. Vianello et al. (2019) found MPs with the concentration of 1.7 and 16.2 particles/m^3 in an indoor environment from which PES was the predominant synthetic polymer. Likewise, Gaston et al. (2020) found MPs (3.3 ± 2.9 fibers and 12.6 ± 8 fragments /$m^{3)}$ in the indoor environment which is high as compared to the MPs

(0.6 ± 0.6 fibers and 5.6 ± 3.2 fragments/m^3) found in the outdoor environment. Liu et al. (2019a) found high concentration of PET and polycarbonate MPs at higher concentration in indoor environment ranging 1550–120,000 mg/kg and 4.6 mg/kg, respectively, than outdoor environment 212–9020 mg/kg and 2 mg/kg, respectively.

The textile industry is responsible for the majority of the microfibers discovered in the ocean, with additional major sources being indoor and outdoor washing, household drainage, and direct dumping of waste clothing into rivers (Almroth et al. 2018). According to Browne et al.'s (2011) study, both natural fibers as cotton and wool and synthetic as PES and nylon are the textile fibers that are released into the environment resulting from domestic washing. It was discovered that 6 kg of synthetic fabric (PES, acrylic textiles, and PES-cotton blend) may release about 140,000–700,000 fibers every wash in a commercial home laundry washing machine (Napper and Thompson 2016). Another study found that a single garment may discharge over 1900 fibers every wash (Browne et al. 2011).

Moreover, the usage of detergent during household laundry has the greatest influence on microfiber production. When detergent was used to wash clothes, it produced around 75% more microfibers than when water alone was used (Hernandez et al. 2017). Similarly, De Falco et al. (2018) suggested that the use of detergents increases the number of microfibers released from fabrics during washing; under these conditions, powder detergents produce more microfiber shedding than the liquid detergents in domestic washing, and that, in general, the industrial washing detergents have a greater influence than domestic washing detergents. According to research, one pair of worn jeans can emit 56,000 ± 4100 microfibers per wash (Athey et al. 2020). Likewise, a single fleece jacket may produce around 2000 microfibers in a single wash (Pirc et al. 2016).

6.4.2 Personal Care Products

Plastic micro-beads (MBs) are commonly used as an abrasive component in a wide range of personal care products (PCP), comprising facial scrubs and exfoliating soaps, skin cream, shampoos, showering gel, and liquid cosmetics (Xu et al. 2020). The abrasive scrub cleanser was produced when people comprehended that mechanical exfoliation—the process of removing the outermost skin layer with an abrasive material—cause smoother skin (Chang 2013). It is predicted that an exfoliant may discharge between 4594 and 94,500 MBs in a single application (Napper et al. 2015).

Exfoliate agents are MBs, which come in a range of sizes and forms. In 9 face scrubs from mainland China, the mean density of MBs was 20,860particles/g, with sizes ranging from 85 to 186 μm. Every year, up to 209.7 trillion MBs (306.9 t) are emitted, accounting for 0.03% of all plastic debris that ends up in the ocean (Cheung and Fok 2017). Polyethylene MBs are often used in personal care products, with research estimating that the average individual in the United States consumes 2.4 mg of MBs each day (Gouin et al. 2015).

In toothpaste, MBs remove plaque and stains due to their abrasive action (Prata 2018a, b). In toothpaste and face cleansers, MBs were discovered between 3 and 178 m, with the bulk having granular forms (Praveena et al. 2018). Toothpaste alone can contain up to 1.8% PE (by weight), which is similar to the PE found in WWTP effluent (Carr et al. 2016). Moreover, about 4000 MBs can release through toothpaste (Carr et al. 2016). The risk assessment shows that about 871 million grams of MPs were released by toothpaste practice in Istanbul (Ustabasi and Baysal 2020).

6.4.3 Paints

Non-aqueous paints are considered plastic since their primary ingredients have a polymer backbone and often contain co-polymers such as alkyls, PES, and epoxies (Zhou 2014). MPs from paints can be released into the environment through normal wear and tear (weathering), removal of old paint layers (sanding, abrasion), and washing brushes and rollers (Verschoor et al. 2016). Building coatings, a type of paint used to prevent corrosion and fouling while still serving an aesthetic purpose, are a source of paint MPs. MPs are released into the environment through these surfaces as a result of UV irradiation, sanding of old paint layers, or meteorological conditions, which are subsequently washed by rainfall into surrounding water bodies (Verschoor et al. 2016; Gaylarde et al. 2021). A total of 490 tons of emissions were estimated from the building sector (Verschoor et al. 2016).

Road marking paints, which may include PE, polyurethane (PUR), and other PES, as well as PA, is another urban source of paint particles (Gaylarde et al. 2021). The global demand for road marking paints is projected to be 588Ktons annually, with global release accounting for 7% of entire consumption (Boucher and Friot 2017). It is estimated that Sweden emits 504 tons of MPs each year as a result of road marking (Magnusson et al. 2016). The quantity of road marking paint MPs on roadside snow-banks in two cities in Sweden, Lule and Ume, was studied. In the 16 samples taken, they discovered 417,600 particles/m^2 (Vijayan et al. 2019).

6.4.4 Landfill

Municipal solid waste, packaging, littering, landfills, and the building sector have all been linked to the release of MPs into the environment (Xu et al. 2020). Landfilling, a commonly used waste disposal method across the world, is projected to hold between 21 and 42% of worldwide plastic waste output (Nizzetto et al. 2016). The widespread use of open, unlined landfill sites in developing nations means plastic waste will readily release secondary MPs to the environment (Dehghani et al. 2017). Wind may also blow plastic trash out of landfills (Cai et al. 2017). MPs were found in landfill leachate from both active and closed landfills at the 4 locations in southern China, with abundances ranging from 0.42 to 24.58 particles/L with PE and PP being the main polymers (He et al. 2019). According to

Alimi et al. (2018), landfill leachate may introduce MPs into the soil. The generation of MPs from landfill is a long process that might last to 100 years in 15 Chinese cities, with an average leachate generation of 1300–3200 L/t waste (Yang et al. 2015b). Sun et al. (2021) discovered 235.417.2 MPs/L in untreated leachate.

6.4.5 Tire Wear

At the moment, vehicle tires are made from a variety of synthetic polymers, including butadiene, PES fiber, styrene, and halo-butyl rubber (Mishra et al. 2020). Tire wear particles are produced by abrasion on road surfaces and are generally concentrated in the 50–350 μm fraction (Vogelsang et al. 2019). Abrasion causes by the shear and heat in the tire, which results in the development of wear particles. Because of the shear stresses, rather large tire particles are emitted (Kole et al. 2017). It has been projected that roughly 503,600 tons/year of microfibers are emitted from tire wears in the European Union, with a total of about 52,000 and 136,000 tons/year reaching water sources (Hann et al. 2018).

Tire and road wear particles have a density of around 1.2–1.3 g/cm^3 (Verschoor et al. 2016). It has been reported that the tires of about 1500 million cars emit around 4 Mt. of man-made fibers into the environment globally, with a per capita emission of 5 kg annually (Mishra et al. 2020). According to data from 13 countries, tire-derived MPs emissions per capita per year ranged from 0.23 kg (in India) to 4.7 kg (in the United States), with a global average of 0.81 kg (Kole et al. 2017).

6.4.6 City Dust

City dust in the urban runoff is a major cause of pollution in rivers. A significant percentage of the components of city dust are polymer-based materials, such as tires, which are considered MPs (Verschoor et al. 2016). According to Dehghani et al. (2017), street dust contains between 88 and 605 MPs per 30 g of dry dust, with particle sizes ranging from 250 to 500 μm. Similarly, Abbasi et al. (2017) found MPs and MRs commonly as fibers and fragments through street dust ranging from 21 to 166 particles/g and 4.4 to 78 particles/g, respectively. According to them, the concentration of microfibers in the industrial area was high than in urban areas as 0.76 particles m^{-3} and 0.63 particles m^{-3}, respectively. Another research discovered non-fibrous and fibrous particles ranging from 175 to 313 particles/m^3/day through atmospheric fallout (Cai et al. 2017). Allen et al. (2019) discovered a relative day-to-day count of 249 fragments, 73 films, and 44 fibers/m^2 in a remote mountain catchment. Furthermore, Dris et al. (2017) found microfibers in the atmospheric fallout ranged from 2 to 355 particles/m^2/day. According to them, the fluxes are greater at the urban site than at the suburban location. ZHOU et al. (2017) discovered MPs ranging from 130 to 624 particles/m^2/day^1 in dry and wet atmospheric fallout.

6.4.7 Agricultural

Soils, particularly agricultural soil, have been identified as important MP sinks (Xu et al. 2020). MPs can enter into the environment through multiple sources. Plastic mulch films and greenhouse materials and soil conditioners such as PUR foam and PS flakes are direct sources in agriculture, but general littering and the use of treated wastewater and bio-solids are indirect sources (Ng et al. 2018). Organic fertilizers derived from bio-waste fermentation and composting can potentially act as a vehicle for MP penetration into soil (Weithmann et al. 2018). MPs in compost could reach a concentration of up to 1200 mg/kg (Bläsing and Amelung 2018). It has been predicted that bio-solid applications may provide up to 300,000 tons of MPs to farmed soils in North America each year (Nizzetto et al. 2016).

The plastic sheets have also been frequently used as greenhouse shade or mulching film. Plastic mulch film has been pushed as a strategy to improve resource efficiency and food security during the last few decades (Zhang and Liu 2018). Low-density PE foils are also widely used in agriculture to protect crops, reduce weeds, regulate temperature, and retain irrigation water in the soil. These polymers grow brittle with time and break down into micro-sized pieces (Hüffer et al. 2019). Plastic mulching is commonly utilized in agricultural activities for fertilizer holding, heat retention, water retention, and soil enhancement (Liu et al. 2018). In 2017, China used more than 1.47 million tons of agricultural plastic mulching film (Gao et al. 2019). While there were 4 Mt. of agricultural plastic sheets in the international market in 2016, the value is anticipated to grow at 5.6% annually by 2030 (Von Moos et al. 2012). The low recovery rate of plastic film residues adds significantly to the growing buildup of MPs in agricultural soil (Kasirajan and Ngouajio 2012).

MPs, once released into the soil, may be subjected to a complex and diverse system of environmental processes, posing a variety of ecological hazards (Li et al. 2020a). MPs are likely to be carried vertically through the soil via biopores, cracking, or plowing, and horizontally by soil biota or agricultural processes such as harvesting and plowing (Möller et al. 2020). According to Previous studies, MPs can affect soil physical properties like bulk density and water dynamics, lowering overall soil bulk density while boosting rhizosphere density (Helmberger et al. 2020).

6.4.8 Industrial

The plastic resin pellets and powder overflow from an air blasting machine, MBs used in PCP, and ingredients used to make plastic items are all industrial sources of plastics (Wang et al. 2020). MPs can be released into the environment by direct leakages during transfer and transport to/from the industries as a primary source like pellets and MBs (Antunes et al. 2018). Meanwhile, pellets could also originate as industrial abrasives (Mao et al. 2020). Nurdles, the precursors for bigger plastic objects, are a major contaminant around areas of heavy industrial activity, such as harbors, as a result of unintentional loss and leakage during transit. Furthermore, harbors may contain a high concentration of MBs utilized in industrial operations such as air blasting (Nel et al. 2017).

In the packaging industry, PE is widely used (Kor and Mehdinia 2020). Mason et al. (2018) evaluated MPs from water bottles of 11 different brands through 19 different locations in 9 different nations. Approximately 93% of the bottled water tested positive for MPs impurity out of a total of 259 bottles tested. Another study (Schymanski et al. 2018) discovered 14 particles/L through the single-use plastic bottles and 118 particles/L through the re-turnable plastic bottles that were linked back to the bottle itself. Furthermore, cap-bottleneck friction in plastic bottles has been shown to increase the number of MPs in a bottle (Winkler et al. 2019).

Industries are considered as the source of MPs pollution in the area. According to Fuller and Gautam (2016), the plastic concentrations on the surface soil along the road through the industrial zone varied from 0.03 to 6.7%. MPs were also discovered in the tidal flat soil samples in concentrations of 317 items/500 g (Zhou et al. 2016). This means that the industrial manufacturing process can introduce a wider range of polymers into the soil.

6.4.9 Wastewater Treatment Facility

Wastewater treatment plant (WWTP) discharges have been recognized as potentially major release routes for tiny plastic particles such as meso-, micro-, and nano-size to the terrestrial environment (Hurley et al. 2020). Talvitie et al. (2015) hypothesized that wastewater effluent may function as a route for MPs to reach the environment since they found comparable kinds of MPs, mainly fibers and man-made particles, in both tertiary effluents from a WWTP in Finland and seawater from the Gulf of Finland. Mason et al. (2016) proposed that municipal WWTP produce up to 15 million particles per day after analyzing 90 samples from 17 sites across the United States (U.S.). They estimated that 3–23 billion MPs particles were discharged into the U.S. aquatic bodies each year via urban wastewater.

Several kinds of research on the removal efficiency of MPs in WWTPs have been conducted. Murphy et al. (2016) estimated that 6.5 × 107 MPs/day could enter the aquatic environment through the secondary WWTP in Scotland, instead of removal efficiency of 98%. The effectiveness of WWTPs in Sweden was measured, with a total capacity of 1,502,000 people equivalent in Stockholm, Göteborg, and Lysekil (Magnusson and Wahlberg 2014). The influent and effluent were filtered through 20 μm and 300 μm filters and the number of MPs were counted using a microscope. On average, 19.8% of the MPs >20 μm and 0.6% >300 μm passed through WWTP. Leslie et al. (2017) measured the efficiency of WWTPs in the Netherlands. They compared the number of MPs in the influent and effluent of 5 WWTP. The data showed that on average 28% of the MPs between 10 μm and 5000 μm pass the WWTP.

According to Carr et al. (2016), the tertiary treatment facility was shown to be efficient in eliminating MPs, with no MP fibers or particles identified in the final tertiary effluent of the WWTPs examined. Furthermore, they discovered that the secondary treatment process was as effective, as higher as 99.9% elimination, resulting in an average discharge of one MP per 1400 L of treated wastewater.

6.5 Ecological Impact of Microplastics

The MPs can easily be transported by wind, because of their small size and lesser density, and can linger in the atmosphere for a long period (Liu et al. 2019b). The ecological risk caused by suspended airborne MPs in Shanghai was assessed using single ecological hazardous indices, with values ranging from 0.23 to 0.64 indicating a minor threat to the research area (Liu et al. 2019b). MPs may be transported by air to other places, polluting the environment via re-concentration (Enyoh et al. 2019).

Airborne MPs may contain a variety of hazardous compounds, including untreated monomers, plastic additives, and other harmful contaminants absorbed from the environment, such as PAH, persistent organic pollutants (POP), heavy metals, and microbes, increasing their toxicity to organisms (Huang et al. 2020). Chemical damage to ecological components induced by MP deposition may be more severe than physical stress, resulting in chronic carcinogenic and endocrine abnormalities (Sarker et al. 2020).

MPs are easily eaten by tiny soil creatures such as insects, nematodes, and snails because of their small size, which results in a variety of health consequences (Wang et al. 2020). According to Zhu et al. (2018), exposure to 0.1% PVC MPs for 56 days severely reduced the development and reproduction of *Folsomia candida* in soil by 16.8% and 28.8%, respectively, and significantly altered the metabolic cycle of this species. According to Huerta Lwanga et al. (2016), earthworms are inhibited in their development and eventually die when exposed to polyethylene MPs at concentrations ranging from 0.2 to 1.2% (w/w in dry soil).

6.6 Human Health Impact of Microplastic

Plastic pollution may affect organisms physically, chemically, and biologically (Li et al. 2020b). The major exposure routes in humans are considered to be ingestion, inhalation, and dermal contact (Thompson et al. 2009). After MPs/NPs become airborne, humans may be exposed to them by inhalation (Revel et al. 2018). The proportion below 2.5 μm is mostly retained in the lungs and can pass the respiratory barriers. As a result, the major mechanism of particle toxicity is the development of oxidative stress, leading to inflammation (Feng et al. 2016).

Furthermore, microfiber flying fleeces breathed by people deposit in the lungs tissue and can cause malignancies (Mishra et al. 2019). The previous studies have shown that respiratory inflammation, asthma, diffuse interstitial fibrosis and granulomas with fiber inclusions (extrinsic allergic alveolitis, chronic pneumonia), inflammatory and fibrotic changes in the bronchial and peri-bronchial tissues (chronic bronchitis), inter-alveolar septa lesions can also be caused due to regular and prolonged exposure but the effects depending on the difference on individual metabolism and vulnerability (Prata 2018a, 2018b). Other biological reactions that might occur include genotoxicity, apoptosis, and necrosis (Wright and Kelly 2017).

According to a study, parental exposure to air pollution may be related to increased respiratory demand and airway inflammation in newborns (Latzin et al. 2009). More recently, (Wick et al. 2010) research discovered that fluorescent PS particles with diameters up to 240 nm were taken up by the placenta by breaching the placental barrier and are linked to parental exposure to air pollution during pregnancy.

Ingestion is the other primary route of human exposure to MPs. According to the study, the estimated amount of MPs is 39,000 to 52,000 particles/person/year that is ingested by the consumption of contaminated foodstuff (Cox et al. 2019). However, Catarino et al. (2018) anticipated that the accumulation of dust on the plate's surface during meals may be more significant than the MPs previously present in food. In Scotland, human fiber intake by eating dust fall accidentally during meal varied from 13,731 to 68,415 particles/year/person, which was much greater than the values obtained from mussel consumption (Catarino et al. 2018).

Recently, MPs have been detected through Mussels and fish (Digka et al. 2018), shrimps (Nan et al. 2020), bivalve (Baechler et al. 2020), as well as from commercial fish (Karbalaei et al. 2019). Many factors affect particle uptake and consequent translocation to secondary target organs, including surface charge, hydrophobicity, surface functionalization, and the associated protein corona (Rist et al. 2018). PE particles of up to 50 μm have been shown to translocate to lymph nodes and could in some cases be found in the spleen and liver (Doorn et al. 1996; Urban et al. 2000). While, PS particles of 2 μm size only demonstrated a low degree of translocation along with the gut layer (Doyle-McCullough et al. 2007). MPs are often resilient to destruction in-vivo and remain until they are removed. This might pass through the gastrointestinal (GI) system if they are concentrated in the mucus layer released by the gut wall cells, or by urine, pulmonary alveoli, cerebrospinal fluid, bile, or milk in nursing females (Waring et al. 2018). Recent research by Schwabl et al. (2019) detected 9 different types of MPs with the size range of 50-500 μm per 10 g through the human stool.

Skin contact with MPs is seen as a fewer relevant route of exposure; however, it has been hypothesized that nano-plastics (<100 nm) may be able to penetrate the dermal layer (Revel et al. 2018.). Particles may come into touch with the skin via PCP, fabrics, or indoor dust (Rahman et al. 2020). MPs and MBs from toothpaste and personal care products can potentially enter the body through the skin, mucosal membranes, and gastrointestinal mucosa (Lassen et al. 2012). Furthermore, this pathway is more frequently connected with exposure to plastic monomers and compounds such as phthalates, endocrine disruptors, and bisphenol A (BPA) through everyday usage of common appliances (Prata et al. 2020).

There are two basic routes for substance absorption in the skin: via the skin appendages or through the stratum corneum and underlying layers (Schneider et al. 2009). Because of the size of the MPs and the fact that particle uptake through skin needs stratum corneum penetration, which is limited to particles smaller than <100 nm, absorption via the skin is unlikely (Revel et al. 2018). According to Van Tienhoven et al. (2006) research, and in-vivo subcutaneous introduction of <10 mm plastic disks into mice discovered that after 98 days, PE disks induced encapsulation with minimal inflammation, whereas, PVC containing organo-tin or plasticizers prompted encapsulation with inflammatory in-filtrate and moderate degeneration and necrosis, possibly due to leachate toxicity. In human epithelial cells causes oxidative stress from exposure to micro- and nano-plastics as well (Schirinzi et al. 2017; Fig. 6.2).

Apart from particle toxicity, MPs may also pose chemical and biological risks. Many compounds, including anti-ultraviolet radiation stabilizers, phthalates, and BPA, are often employed in the manufacture of plastics (Hirai et al. 2011). According to research, phthalates in microfibers can harm the human body by causing reduced male reproductive system development, testosterone levels, impaired hormone system function, early puberty, reproductive and genital defects, and reduced sperm count level, whereas, BPA can damage female reproductive hormones. Likewise, the other dangerous compounds found in microfibers reach our bodies and may cause DNA and protein disruption (Meeker et al. 2009). Furthermore, flame retardant chemicals used in plastic goods, such as polybrominated diphenyl ethers (PBDE) and tetrabromobisphenol A (TBBPA), have been predicted to alter thyroid hormone homeostasis, although PBDEs alone have anti-androgen activity (Sjödin et al. 2003). Because of their hydrophobic surface, they can adsorb and concentrate hydrophobic organic pollutants like OCP, PAHs, PCBs, and DDT which can have a negative influence on human health (Frias et al. 2010; Campanale et al. 2020).

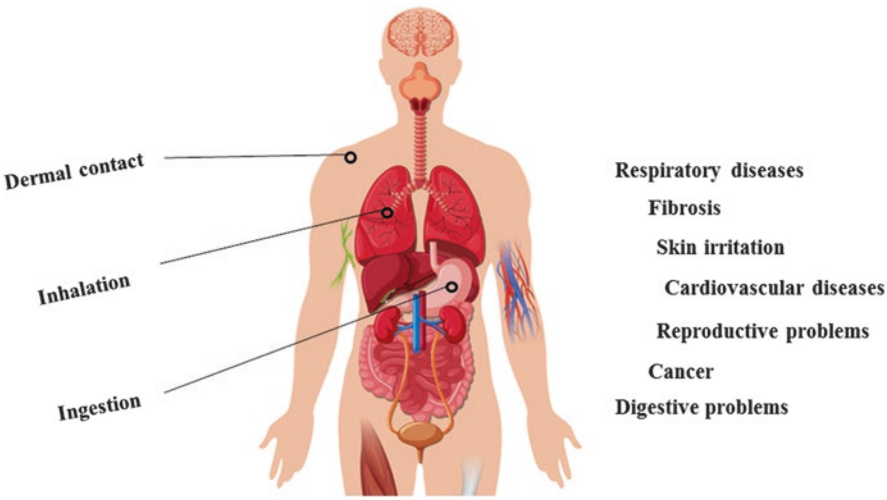

Fig. 6.2 Impacts of microplastics on human health

6.7 Conclusion

MPs have become the global environmental problem as these particles are persistent for a long time due to their low degradation properties and had been found in an aquatic and terrestrial environment. It has already been detected through different environmental compartments. The terrestrial environment is the major source of MPs and its main sink is the ocean where it bio-accumulates and bio-magnify, posing different health impacts to the organisms. These particles may be resulting from the environmental degradation of larger debris or may purposefully manufacture to the small size fraction. These particles also cause adverse health effects to the human when inhaled, ingested, and dermal contact. Furthermore, these particles have chemical additives and adsorbed organic pollutants that may enter organisms posing different health issues.

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Chapter 7
Microplastic (MP) Pollution in the Context of Occurrence, Distribution, Composition and Concentration in Surface Waters and Sediments: A Global Overview

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Abstract Microplastic (MP) pollutants are widespread and have been detected in both surface waters and sediments across the globe. At some places, MP abundance reached 90–100% of the samples sampled in surface waters. They (MP) have even reached the remote and pristine parts of the world. Plastic litters have outnumbered fish larvae and plankton at several places in the world. The increased abundance of MP was reported in surface waters close to larger cities and cities with higher population density, enclosed basins, gyres, dams/reservoirs and coastal areas. The Coastal Soya Island, South Korea (having 46,334 MP particles/m^2) and the Pearl River Estuary, Hong Kong (having 5595 MP particles/m^2), are the two MP pollution 'hot spots' in surface waters. MP pollutants are also widespread and detected in sediments of a wide range of aquatic environments including archipelagos, bays, channels, coasts, beaches, deep seas, estuaries, lagoons, rivers, shellfish farms and ship-breaking yards. At some places, MP abundance in sediments reached around 64% to 100% of the samples sampled. The Kachelotplate and Spiekeroog islands, Germany (having 38,000 MP particles/kg dw sediments) and Jakarta Bay, Indonesia (having 30,006 MP particles/kg dw sediments), are the two MP pollution 'hot spots' in sediments. Based on polymer shapes, the most commonly detected MPs in both surface waters and sediments are fibres, fragments, foams and films. On the other hand, based on polymer chemistry, the most commonly detected MPs in surface

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waters and sediments are polyethylene (PE), polypropylene (PP) and polystyrene (PS). To reduce impacts from plastic wastes and plastic pollution, a number of measures have been suggested.

Keywords Microplastic · Surface waters, sediments · Rivers, Lakes, Estuaries, Seas, Oceans · Global

7.1 Introduction

Plastic wastes are ubiquitous and are reported from the Arctic to the Antarctic and from the surface to sediments (Kibria 2017) (Fig. 7.1). They are most commonly derived from petrochemicals (natural gas, oil or coal). Plastics are synthetic **polymers** and one of the most significant pollutants of recent times in the aquatic environment. Polymers are composed of many repeated subunits of monomers. In fact, plastics are simply a chain of molecules linked together. These chains are called polymers. Ethylene and propylene are the most important sources of plastic products. Since plastic is made from nonrenewable petrochemicals, so, consequently, they are principally based on the carbon (up to 87.6% C) and hydrogen (up to 14.5% H) atom. It usually consists of additives, fillers and colours (Lackner 2015).

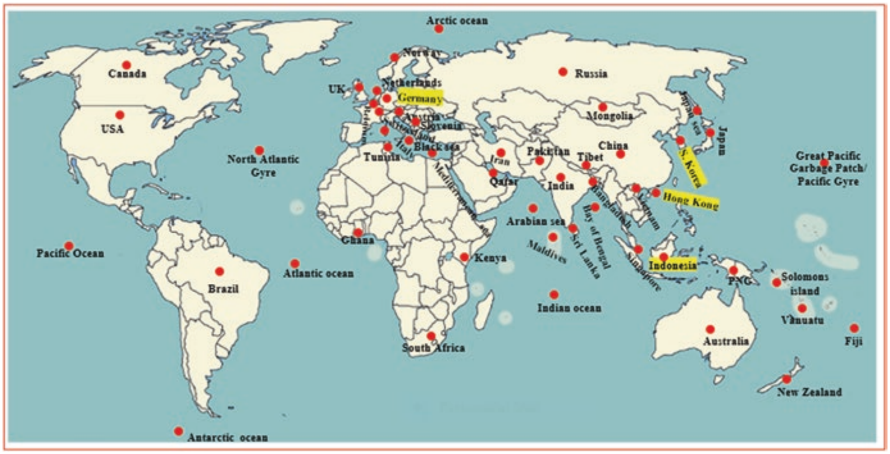

Fig. 7.1 Examples of microplastic (MP) pollutants detected/reported in surface waters and sediments in the world waterways (shown in red dots). The four pollution **'hot spots'** are highlighted in yellow in the figure. The four 'hot spots' are (i) South Korea (Soya Island, 46,334 MP particles/m^2) (*surface waters*), (ii) Hong Kong (Pearl Estuary, 5595 MP particles/m^2) (*surface waters*), (iii) Germany (Kachelotplate and Spiekeroog islands, 38,000 MP particles/kg dw) (*sediments*) and (iv) Indonesia (Jakarta Bay, 30,006 MP particles/kg dw) (*sediments*). Figure 7.1 depicts that MP pollution is ubiquitous and reported from the Arctic to the Antarctic and to every continent (Asia, Africa, Australia, Europe, North America, South America). Figure 7.1 was prepared based on information compiled in Tables 7.1 and 7.2 for this chapter

The common types of plastic are polyethylene (PE), polypropylene (PP), polystyrene (PS), polyethylene terephthalate (PET), polyamide (PA), polyester (PES) and polyvinyl chloride (PVC) (Kibria 2018; Kibria et al. 2021). Based on thickness/size, plastics are classified as nanoplastic, microplastic, mesoplastic and macroplastic. Microplastics are generally defined as plastic particles smaller than 5 mm (particle size 1–5000 μm). Microplastics are further classified into primary microplastics and secondary microplastics (based on morphology/appearance) (Almi et al. 2018; Crawford and Quinn 2017; Germanov et al. 2018). *Primary microplastics* include microbeads and pellets. Microbeads are commonly made from PE, PP, PET, PA and polymethyl methacrylate (PMA) and found in cosmetics, personal care products (soap, toothpaste, facial scrub), nail polish, lipsticks, microfibres (used in textiles), virgin pellets and cleaning products (Potter 2017; Germanov et al. 2018). Pellets or nurdles are raw material/ building blocks for the manufacture of plastic products. *Secondary microplastics* (resulting from environmental degradation of larger plastic items as a result of weathering, wave action, wind abrasion, biodegradation and ultraviolet photodegradation) include synthetic fibres (fishing gears, textiles), fragments (plastic bags, bottles, car tyres) and films (Kibria et al. 2021).

Plastic waste is an emerging contaminant and does not readily biodegrade but persists in the aquatic environment for long periods. Plastic pollution in freshwater and marine environments has been identified as a global problem. Plastic pollution and its negative impacts in the freshwater and marine environments are getting more attention on the international agenda lately. Governments, nongovernmental organisations, industry and international organisations are taking new initiatives to reduce plastic waste in marine, riverine and terrestrial environments. Plastics are highly durable materials, and its persistence coupled with increasing emissions to the environment has resulted in a wide-scale accumulation from shallow waters to the deep sea. It is estimated that plastic debris accounts for 60–80% of marine litter, reaching 90–95% in some areas (Xanthos and Walker 2017). About 80% of plastic pollution originates from land-based sources with the remainder (20%) coming from ocean-based sources (fishing nets, fishing ropes) (Sebille et al. 2016). The major land-based sources are illegal dumping (mismanaged plastic wastes) and inadequate waste management (Sebille et al. 2016; Kibria 2017). Mismanaged plastics (plastic dumped openly) enter the environment via inland waterways, wastewater outflows and transport by wind or tides (Jambeck et al. 2016). Small plastic particles (microplastics/microbeads) cannot all be filtered out, making wastewater treatment plants a significant source of microplastic pollution into aquatic ecosystems (Sebille et al. 2016; Xanthos and Walker 2017). Microplastics that once enter the aquatic environment can travel vast distances floating in seawater or sediments to the seabed (Xanthos and Walker 2017). The objective of this chapter is to collect, collate, synthesise, analyse, interpret and document microplastic pollution (occurrence, distribution, composition and concentrations) in the global surface waters and sediments.

7.2 Microplastic (MP) Pollution in Surface Waters

7.2.1 MP Occurrence and Distribution in Surface Waters

MP pollutants are widespread and occur at all types of surface waters (creeks, lakes, channels, rivers, estuaries, coasts, oceans and gyres). They (MP) have been detected across the globe from the Arctic to the Antarctic (including Australia, Austria, Bay of Bengal, Canada, China, European coasts, Germany, Great Pacific Garbage Patch (GPGB), Hong Kong, India, Japan, Kenya, Mediterranean Sea, Mongolia, New Zealand, North Atlantic Subtropical Gyre, North Pacific Central Gyre, North Western Pacific, Oceania, Papua New Guinea (PNG), Russia, Qatar, Sri Lanka, South Korea, South Pacific Ocean, Sub-Antarctic, Switzerland, Tibet, the USA and Vanuatu) (Table 7.1 and Fig. 7.1). At some places, MP abundance reached 90–100% of the samples sampled as listed below:

- MP detected in 100% of the samples*: Canada* (North Eastern Pacific Ocean and coastal British Columbia) (Desforges et al. 2014), *Japan Sea* (Day et al. 1990), *Mediterranean Sea* (basin) (Cózar et al. 2015), *Mediterranean Sea* (Gulf of Lion) (Schmidt et al. 2018), North Pacific Central Gyre (Moore et al. 2001) and *Russia* (Baltic beaches, Kaliningrad region) (Esiukova 2017).
- MP detected in 98% of the samples*: The USA* (estuaries in the Chesapeake Bay) (Yonkos et al. 2014; Fok et al. 2017).
- MPs detected in 96% of the samples: *South Pacific Ocean* (Pacific subtropical Gyre) (Eriksen et al. 2013).
- MPs detected in 95% of the samples: *North Pacific Gyre* (Law et al. 2014; Faure et al. 2015a).
- MPs detected in 94% of the samples: *Great Pacific Garbage Patch* (GPGB) (between California and Hawaii) (Leberton et al. 2018).
- MPs detected in 91% of the samples: *Hong Kong* (Pearl River Estuary) (Fok and Cheung 2015).
- MPs detected in 90% of the samples*: Norway* (Arctic Ocean, Svalbard) (Lusher et al. 2015) and *Mediterranean Sea* (northwestern) (Collignon et al. 2012; Bakir et al. 2020).

Nevertheless, the increased abundance of MPs was reported in waterways close to (i) larger cities (Reisser et al. 2013; Schmidt et al. 2018), (ii) cities with higher population density [(e.g. North Sea (Claessens et al. 2011; Liebezeit and Dubaish 2012; Thompson et al. 2004; Van Cauwenberghe et al. 2013), the Mediterranean Sea (Kaberi et al. 2013; Vianello et al. 2013), Asia (Ismail et al. 2009; Ng and Obbard 2006; Nor and Obbard 2014; Reddy et al. 2006), the highly populated coast of Brazil (Costa et al. 2010)] and (iii) Lakes adjacent to horticultural, agricultural, fishing and tourism activities (Migwi et al. 2020).

MPs pollutant have reached even the **remotest** and **pristine parts** of the world such as in the Arctic Ocean (Svalbard, Norway) (Obbard et al. 2014; Lusher et al. 2015), the Antarctic Ocean (Gregory and Ryan 1997), the Atlantic Ocean (South

Table 7.1 Microplastics composition, concentrations, and occurrences in global surface waters (bays, coasts, estuaries, gyres, lakes, oceans, reservoirs, rivers, seas, wastewater treatment plants)

Country/region and references	Polymer composition (na = data is not available)	Polymer concentration (na = data is not available)	Remarks (MPs = microplastics)
Arctic Ocean Svalbard, Norway (Lusher et al. 2015)	***Shapes:*** Fibres (95%), fragments (4.9%), and films (<0.1%) ***Types:*** Rayon (30%), PES (15%), PA (15%), PE (5%), acrylic (10%) and PVC (5%)	***Range:*** na; ***mean:*** 28,000 MP particles/km^2 (0.028 MP particles/m^2)	1. Plastics detected in **90%** of the samples 2. Rayon, PES and PA were most abundant polymers
Atlantic South Cape Basin (Morris 1980)	***Shapes***: na ***Types***: PE and PP	***Range*****:** 0–3600 MP particles/km^2; ***mean:*** 1874.3 MP particles/km^2 (0.0019 MP particles/m^2)	Plastic pollution reached the most **remote** areas of the world's oceans
Australia Marine, Tasmania (Gregory and Ryan 1997)	na	***Range*****:** na; ***mean*****:** 213 MP particles/km^2	Plastics detected in 71% of the samples (40% fisheries related)
Australia Marine, Victoria (Gregory and Ryan 1997)	na	***Range:*** na; ***mean:*** 47.12 MP particles/km^2	Plastics detected in 62% of the samples
Australia Coastal Waters (Reisser et al. 2013)	***Shapes:*** Hard plastic, soft plastic, plastic line, expanded polystyrene, and pellet ***Types:*** PE (65%), PP (31%), EPS (1%), and ethylene vinyl acetate (0.5%)	***Range:*** 0–48,896 MP particles/km^2; ***mean*****:** 4256.4 MP particles/km^2 (0.000426 MP particles/m^2)	1. Higher amounts of plastic were found close to **cities** 2. PE and PP found associated with packaging and fishing items
Austria Danube River (Lechner et al. 2014)	***Shapes*****:** Pellets, flakes, and spherules (79.4%) of the plastic debris ***Types*****:** na	***Plastic:*** 316.8 MP particles/1000 m^3 ***Fish larvae***: 275.3 individuals/1000 m^3	Plastic litter **outnumbered fish** larvae in Danube River

(continued)

Table 7.1 (continued)

Country/region and references	Polymer composition (na = data is not available)	Polymer concentration (na = data is not available)	Remarks (MPs = microplastics)
Bay of Bengal (BoB) Southwest (Eriksen et al. 2017)	***Shapes:*** Fragments (41%), films (40%), lines (13%), foams (4.5%) and pellets (1%) ***Types:*** na	***Range:*** Few hundreds to 20,000 MP particles/km^2; ***mean***: na	The Ganges River is the largest emitter of plastics to the BoB
Canada Northeastern Pacific Ocean and Coastal BC (Desforges et al. 2014)	***Shapes***: Fibres (75%) ***Types***: na	***Range:*** 8–9200 MP particles/km^2; ***mean***: 2080 MP particles/km^2 (0.02 MP particles/m^2)	1. MPs identified in **100%** of the samples 2. Elevated MPs were found in coastal stations compared to offshore
Canada Lake Winnipeg, (Anderson et al. 2017)	***Shapes***: Fibres (90%), films, foam ***Types:*** na	***Range:*** 53,000–748,000 MP particles/km^2; ***mean:*** 193,420 MP particles/km^2 (0.193 MP particles/m^2)	Plastics detected in 77% of the samples
Canada St. Lawrence River (Castañeda et al. 2014)	***Shapes***: Microbeads ***Types:*** Polyethylene	***Range*****:** na; ***mean***:13,832 microbeads per m^2	Freshwater sediments can act as a sink for plastic pollutants
China Three Gorges Reservoir, *Yangtze River* (Zhang et al. 2015)	***Shapes:*** na ***Types:*** PE (36.79%–57.12%), PP (42.14%–63.21%) and PS (0.0%–12.7%)	***Range:*** 340,8000–1,361,8000 MP particles/km^2; ***mean:*** 846,6000 MP particles/km^2(8.47 MP particles/m^2)	1. Plastic debris accumulated behind the **dam** 2. MPs abundance related to the human activities/population
China Guangdong Province, Beach (Fok et al. 2017)	***Shapes***: Foams (96%), resin pellets (2%) and fragments (2%) ***Types***: na	***Range***: na; ***mean*****:** 6675 MP particles/m^2)	Plastics detected in **98%** of the samples
European Coasts (Galgani et al. 2000)	na	***Range***: 0–101,000 MP particles/km^2; ***mean***: na	Plastics detected in 70% of the samples
Germany The Rhine River (Mani et al. 2015)	***Shapes*****:** Opaque spherules (45%), fragments (37%), transparent spherules (13%), and fibres (2.5%) ***Types:*** PS (70%), PE (15%) and others (PA, PP)	***Range***: na; ***mean*****:** 892,777 MP particles/km^2 (0.89277 MP particles/m^2)	MPs particle in Rhine River are much higher (compared to other rivers and lakes)

Country/region and references	Polymer composition (na = data is not available)	Polymer concentration (na = data is not available)	Remarks (MPs = microplastics)
Great Pacific Garbage Patch (GPGP) (California and Hawaii) (Leberton et al. 2018)	***Shapes:*** Hard plastics, sheets and films (47%) and nets and ropes lines (52%) ***Types***: PE and PP	***Range:*** 216–643,930 MP particles/km^2; ***mean***: 169,460 MP particles/km^2 (0.17 MP particles/m^2)	1. Plastics detected in **94%** of the samples 2. GPGP is a large-scale accumulation zone of marine debris
Hong Kong Pearl River Estuary (Fok and Cheung 2015)	***Shapes:*** Fragments (5%) and pellets (3%) ***Types***: EPS (92%)	***Range***: na; ***mean:*** 5595 MP particles/m^2	1. Plastics detected in 91% of the samples 2. Hong Kong is a hot spot of marine plastic pollution (5595 MP particles/m^2)
India (Arabian Sea) (Jayasiri et al. 2013)	na	***Range*****:** 12–960 MP particles/m^2; ***mean***: 68.33MP particles/m^2	Plastics detected in 41.85% of the samples
Japan Japan Sea (Day et al. 1990)	***Shapes***: Fragments (21.7%), Styrofoam (12.8%), PP line fragments (7.4%), unidentified plastic (7.4%) and raw pellets (5.9%) ***Types:*** PP	***Range***: na; ***mean:*** 74,700 MP pieces/km^2 (0.075 MP particles/m^2)	Plastics detected in **100%** of the samples
Japan (Kuroshio Current area) (Yamashita and Tanimura 2007)	***Shapes***: Fragments (56%), Styrofoam (21%) and resin pellets (1%) ***Types:*** na	***Range:*** 0–352,0000 MP pieces/km^2; ***mean***: 174,000 MP particles/km^2 (0.174 MP particles/m^2)	Plastics detected in 72% of the samples
Japan East Asia Seas (Isobe et al. 2015)	***Shapes***: na ***Types:*** PE (68%) and PP (19%)	***Range***: na; ***mean:*** 1,720,000 MP particles/km^2 (1.72 MP particles/m^2)	The East Asian seas can be regarded as a 'hot spot' since MPs concentration is 27 times greater than in the world oceans
Kenya Lake Naivasha (Migwi et al. 2020)	***Shapes:*** Fragments, fibres and films ***Types:*** PES (35%), PP (25%), PE (20%), PET (10%) and nonplastics (10%)	***Range***: 0.183–0.633 MP particles/m^2; ***mean:*** 0.407 MP particles/m^2	High MPs abundance was found adjacent to settlements, horticultural, agricultural, fishing and tourism activities

(continued)

Table 7.1 (continued)

Country/region and references	Polymer composition (na = data is not available)	Polymer concentration (na = data is not available)	Remarks (MPs = microplastics)
Mediterranean Sea (Faure et al. 2015a)	***Shapes:*** Fragments (73%), films (14%), foams (5%) and lines (2%) ***Types***: PS	***Range:*** 10,000–420,000 MP particles/km^2; ***mean:*** 129,682 MP particles/km^2 (0.13 MP particles/m^2)	Plastics detected in 62% of the samples
Mediterranean Sea Basin (Cózar et al. 2015)	***Shapes***: Fragments (87.7%), films (5.9%), foams (2.3%), fishing thread (2.3%) and pellets/granules (1.8%) ***Types***: na	***Range***: na; ***mean:*** 243,853 MP particles/km^2 (0.244 MP particles/m^2)	Plastics detected in **100%** of the samples
Mediterranean Sea NW, Gulf of Lion (Schmidt et al. 2018)	***Shapes***: Fragments ***Types***: na	***Range***: 6000–1000,000 MP particles/km^2; ***mean:*** 112,000 MP particles/km^2 (0.112 MP particles/km^2)	1. Plastics detected in **100%** of the samples 2. Large cities are sources of microplastics
Mediterranean Sea Northwestern (Collignon et al. 2012; Bakir et al. 2020)	***Shapes:*** Filaments ,> polystyrene > or thin plastic films ***Types***: na	***Range***: 0–892,000 MP particles/km^2; ***mean***: 116,000 MP particles/km^2 (0.116 MP particles/m^2)	1. Plastics detected in **90%** of the samples 2. MPs can be a **carrier of POPs/** pollutants (Teuten et al. 2007), and MPs can act as substrates to transport of alien species/microorganisms (Barnes 2002)
Mongolia Lake Hovsgol (Free et al. 2014)	***Shapes:*** Fragments and films, plastic bottles (37%), fishing gear (25%), plastic bags (16%) and plastic fragments (18%) ***Types***: na	***Range:*** 997 to 44,435 MP particles km^2; ***mean***: 20,264 MP particles/km^2 (0.022 MP particles/m^2)	Microplastic density decreased with distance from the shore; MPs detected in a remote lake
New Zealand Marine (Gregory and Ryan 1997)	na	***Range:*** na; ***mean:*** 748 MP particles/km^2	Plastic debris accounted for 75% of samples (of which, 40% fisheries related)

Country/region and references	Polymer composition (na = data is not available)	Polymer concentration (na = data is not available)	Remarks (MPs = microplastics)
North Atlantic Subtropical Gyre (Law et al. 2010; Faure et al. 2015a)	***Shapes:*** na ***Types***: PE and PP (buoyant plastics materials)	***Range:*** 0–580,000 MP particles/km^2; ***mean:*** 100,000 MP particles/km^2 (0.1 MP particles/m^2)	1. Plastics were detected in 62% of the samples 2. Wind-driven Ekman currents and geostrophic circulation transported floating plastic debris
North Pacific Central Gyre (Moore et al. 2001)	***Shapes***: Fragments > thin plastic films ***Types***: PP (98%)	***Range:*** 31,982–969,777 MP particles/km^2; ***mean:*** 3,34,271 MP particles/km^2 (0.334 MP particles/m^2)	1. Plastics detected in **100%** of the samples 2. The ratio of plastic to plankton (mass) was 6.1:1
North Pacific gyre (Law et al. 2014; Faure et al. 2015a)	na	***Range***: na; ***mean:*** 339,800 MP particles/km^2 (0.34 MP particles/m^2)	Plastics detected in **95%** of the samples
Northwestern Pacific Kuroshio current area (Yamashita and Tanimura 2007)	***Shapes:*** Fragments (56%), Styrofoam (21%) and resin pellets (1%) ***Types***: na	***Range***: 0–352,0000 MP particles/km^2; ***mean:*** 174,000 MP particles/km^2 (0.174 MP particles/m^2)	1. Plastics detected in 72% of the samples 2. Kuroshio current aided in the transport of plastics
Northwestern Pacific (Pan et al. 2019)	***Shape***s: na ***Types:*** PE (~58%), PP (36%), PA/nylon (3.4%), PVC (1.1%) and PS (0.6%)	***Range:*** 640–42,000 MP pieces/km^2; ***mean:*** 10,000 MP particles/km^2 (0.01 MP particles/m^2)	Combined ocean circulation adjacent eddies; the Kuroshio and Kuroshio current were responsible for MPs transport and distribution
Oceania Ducie Atoll, Pitcairn Islands (Gregory and Ryan 1997)	na	***Range:*** na; ***mean:*** 184.7 MP particles/km^2	Plastics detected in 77% of the samples (23% of plastics were fisheries related)
Oceania Raoul Island (Gregory and Ryan 1997)	na	***Range***: na; ***mean*:** 360 MP particles/km^2	Plastics detected in 80% of the samples (25% plastics were fisheries related)

(continued)

Table 7.1 (continued)

Country/region and references	Polymer composition (na = data is not available)	Polymer concentration (na = data is not available)	Remarks (MPs = microplastics)
Papua New Guinea Motupore Island (Smith 2012)	na	***Range***: na; ***mean***: 13.74 MP particles m^2	Plastics detected in 89.7% of the samples
Russia Baltic Beaches, Kaliningrad Region (Esiukova 2017)	***Shapes:*** Foams > fragments > films > pellets ***Types:*** PP, foamed PE, foamed PS, PVC and foamed polyurethane	***Range***: 7–5560 MP particles/m^2; ***mean:*** 42–1150 MP particles/m^2	1. Plastics detected in **100%** of beach samples 2. The most prevalent type of plastic pollution was foam plastic (polystyrene foam and extruded polystyrene)
Qatar (Arabian Bay; Sandy Beaches) (Abayomi et al. 2017)	***Shapes***: Fibres (93.8%), films (4.7%) and fragments (1.5%) ***Types***: PE and PP	***Range***: 44,000–1,500,000 MP particles/km^2; ***mean:*** 78,000 MP particles/km^2 (0.078 MP particles/m^2)	**Fishing lines, nets and ropes** were the main sources of fibres, and low-density polymers (PE and PP) were the dominant polymers
South Korea Coastal Soya Island (Kim et al. 2015)	***Shapes***: na ***Types***: EPS (>99% of MP), PP (0.13%) and PE (0.10%)	***Range:*** 56–285,673 MP particles/m^2; ***mean:*** 46,334 MP particles/m^2	46,334 MP particles/m^2 is the **highest level** globally recorded
South Pacific Ocean Pacific Gyre (Eriksen et al. 2013)	***Shapes:*** Fragments > pellets > lines > films ***Types***: na	***Range***: 0–396,342 MP particles/km^2; ***mean:*** 26,898 MP particles/km^2 (0.027 MP particles/m^2)	Plastics detected in **96%** of the samples (25% fisheries related)
Sri Lanka Coastal Waters (Koongolla et al. 2018)	***Shapes***: Fragments ***Types***: PE, PP and PS	***Range:*** 0.000–0.021 g/m^3; ***mean:*** 0.0182 g/m^3	1. MP particles found in 70% water samples 2. PE and PP were the major polymer types identified, which could have been originated from plastic bags, bottles, beverage container caps and drinking straws, and fishing (nets, ropes)

Country/region and references	Polymer composition (na = data is not available)	Polymer concentration (na = data is not available)	Remarks (MPs = microplastics)
Sub-Antarctic Atlantic Ocean (Gregory and Ryan 1997)	na	***Range:*** na; ***mean:*** 677.6 MP particles/km (0.0007 MP particles/m^2)	Plastics detected in 77% of the samples (25% fisheries related)
Switzerland Swiss Lakes (Faure et al. 2015b)	***Shapes:*** na ***Types***: PE (62%), PP (15%), PS (12%) and PVC (4%)	***Range***: 11,000–220,000 MP particles/km^2; ***mean:*** 91,000 MP particles/km^2 (0.091 MP particles/m^2)	Hydrophobic pollutants (see Kibria et al. 2010) (PCBs, PAHs, OC pesticides, PBDEs, phthalates, nonylphenols, bisphenol-A adsorbed in MPs
Tibet Lakes in Siling Co Basin (Zhang et al. 2016)	***Shapes:*** na ***Types:*** PE > PP > PET, > PVC > PS	***Range:*** 4–563 MP particles/m^2; ***mean***: 113.8 MP particles/m^2	1. MPs even reached remote 'Tibet' with a low population 2. Atmospheric fallout is a possible transport pathway
USA WWTP (McCormick et al. 2014)	***Shapes***: Fragments > pellets > Styrofoam > fibres	***Mean*****:** Upstream: 730,341 MP particles/km^2; downstream: 669,8264 MP particles/km^2 (6.7 MP particles/m^2)	1. The downstream of WWTP had higher MPs. 2. MPs particles found colonised by dense bacterial biofilms
USA Lake Erie (Eriksen et al. 2013)	***Shapes:*** Pellets > fragments (most abundant) > others (line, film, foam) ***Types***: na	***Range:*** 4686–466,305 MP particles/km^2; ***mean***: 105,503 MP particles/km^2 (0.105 MP particles/m^2)	MPs detected in **95%** of the samples of Lake Erie
USA Four Estuaries in the Chesapeake Bay (Yonkos et al. 2014)	***Shapes***: na ***Types***: PE	***Range:*** 5.534–259,803 MP particles/km^2; ***mean:*** 93,883.7 MP particles/km^2 (0.094 MP particles/m^2)	1. Microplastics detected in **98%** of the samples 2. Highest microplastics concentrations occurred shortly after major rain events

(continued)

Table 7.1 (continued)

Country/region and references	Polymer composition (na = data is not available)	Polymer concentration (na = data is not available)	Remarks (MPs = microplastics)
Vanuatu South Pacific Ocean (Bakir et al. 2020)	***Shapes***: na ***Types:*** PE (41%), PP (24%) and PS (23%)	***Range:*** 9779–101,700 MP particles/km^2; ***mean:*** 51,144 MP particles/km^2 (0.051 MP particles/m^2)	Enclosed regions are likely the **accumulation zones** of plastics

Note: *Gyre* is a large system of rotating ocean currents, *bay* is a broad inlet of the sea where the land curves inward. *ABS* acrylonitrile butadiene styrene, *ALKD* alkyd resin, *AN* acrylonitrile, *CP* cellophane, *dw* dry weight, *EPP* ethylene propylene, *EPR* ethylene propylene rubber, *EPS* expanded polystyrene, *fragments* broken down minute pieces of larger plastic particles, *HDPE* high-density polyethylene, *LDPE* low-density polyethylene, *MPs* microplastics (<5 mm), *na* data/information is not available, *nylon* polyamide, *PA* polyamide/nylon, *PAN* polyacrylonitrile, *PE* polyethylene, *PEP* poly(ethylene propylene), *PES* polyester, *PET* polyethylene terephthalate, *PEVA* polyethylene vinyl acetate, *PMA* polymethylacrylate, *PP* polypropylene, *PS* polystyrene, *PSS* polystyrene sulfonate, *PTFE* polytetrafluoroethylene, *PU* polyurethane, *PVC* poly vinyl chloride

Cape Basin) (Morris 1980), mid-ocean islands (beaches of the Archipelago of Fernando de Noronha, Atlantic) (Ivar do Sul et al. 2009), Hawaiian Islands (the remote beaches of Hawaiian sands) (McDermid and McMullen 2004), Tibet (lakes in Siling Co basin) (Zhang et al. 2016) and Mongolia (Lake Hovsgol) (Free et al. 2014). Moreover, there have been **more plastics than fish/plankton** in various waterways: For example, (i) plastic litters outnumbered fish larvae in the Danube River, Austria (Lechner et al. 2014); (ii) MPs were present in 61% of zooplankton samples in Portuguese coastal waters (Frias et al. 2014); and (iii) The mass of plastic was approximately six times that of plankton in the North Pacific Central Gyre (Moore et al. 2001).

In fact, several list of MP **accumulations zones** have been identified worldwide, for example, (i) the **semi-enclosed basin** in the Mediterranean Sea (surrounded/enclosed by states/lands with high coastal population and connected to the sea or the ocean by a narrow outlet/inlet) (*http://scienceline.ucsb.edu/getkey.php?key=995*) (where outflow mainly occurs through a deepwater layer). A study carried out by Cózar et al. 2015 found plastics in 100% of the sites sampled in the Mediterranean Sea, and the concentration estimated at the sampling site was 243,853 MP particles/km^2 (Cózar et al. 2015). (ii) The **Great Pacific Garbage Patch** (GPGB) (1.6 million km^2 and located between Hawaii and California; GPGB collects marine debris/plastics in the North Pacific Ocean via the action of vortex/spinning). Currently, GPGB contains 1.8 trillion pieces of floating ocean plastic, of which 94% was MPs; the concentration estimated at GPGB was 169,460 MP particles/km^2. The plastic objects identified at GPGB are containers, bottles, lids, bottle caps, packaging straps, eel trap cones, oyster spacers, ropes and fishing nets. Fishing nets alone represented more than 46% of the plastic load at GPGB (Leberton et al. 2018). (iii) Accumulation of floating microplastics in **dam/reservoirs** (where plastics got accumulated behind the dam as floating microplastics cannot pass the dam). Zhang et al. 2015 investigated the occurrence and distribution of microplastics in surface waters from the Three Gorges dam (TGR) in China. A high abundance of MPs was observed in samples collected from the TGR with an estimated concentration of 846,6000 MP particles/km^2 (Zhang et al. 2015).

The **sources of plastic pollution** in waterways can be both land- and sea-based. Around 80% of plastic pollution in the marine environment originates from land-based sources, while the remainder comes from ocean-based sources (fishing nets, fishing ropes) (Sebille et al. 2016). MPs can enter the freshwater and marine environment (see Kibria et al. 2021) via the following pathways: (i) *synthetic textile fibres* (rayon, polyester and acrylic) released during the washing of clothes (Andrady 2011; Browne et al. 2011; Cole et al. 2014; Anderson et al. 2017; Peng et al. 2017). A single garment can produce >1900 fibres per wash (Browne et al. 2011); (ii) *fishing and aquaculture activities*: fishing gears, handlines and ropes (Andrady 2011; Browne et al. 2011; Cole et al. 2014; Frias et al. 2014; Esiukova 2017). Foams used in fishing boats and seafood markets (Fok et al. 2017) and Styrofoam buoys for aquaculture (oyster, mussel) (Kim et al. 2015); (iii) *WWTP (waste water treatment plant) influents/effluents* (Magnusson and Noren 2014; Leslie et al. 2017) and wastewater discharge (Dris et al. 2015); (iv) *biosolids* containing plastics applied in agriculture farming and forestry (Leslie et al. 2017); (v) *atmospheric fallout* of microfibres (Dris et al. 2015); (vi) *cosmetics and personal care products* containing microbeads (released during showering/bathing) (Liebezeit and Dubaish 2012; Zhang et al. 2015); (vii) *Ship-breaking yards* (insulating, fabrics, packaging materials used in ships) (Reddy et al. 2006); (viii) *Urban and industrial* outflows containing plastic fibres (Naji et al. 2017); (ix) *seasonal outputs*: wet season/high runoff can have high plastic inputs in waterways compared to the dry season (Cheung et al. 2016); and (x) *other sources* (beach visitors and recreational activities causing plastic pollution in beaches (Jayasiri et al. 2013; Esiukova 2017).

Plastics can enter or be transported to the aquatic environment through many **pathways** (Ryan 1988; Galgani et al. 2000; Derraik 2002; Andrady 2011; Eriksen et al. 2013; Lambert et al. 2014; IUCN 2018; USEPA 2018; Kibria et al. 2021) including the following:

- Intentional littering/open dumping of bags, bottles and other plastic items (during large public gatherings/events or from coastal tourism).
- Urban and stormwater runoff.
- Urban drains.
- Extreme weather events (floods and cyclones).
- Wind-blown debris.
- Wastewater/sewage treatment discharges (as most wastewater treatment facilities don't filter out microplastics/microbeads).
- Sewage overflows.
- Landfill wastes (mismanaged plastics).
- Accidental spillage during transport and handling of plastic pellets/virgin pellets.
- Release of microfibres during washing/laundering of synthetic clothes/textiles.
- Discharge of microbeads from the use of personal care products.
- Fibres from fishing nets and lines (plastic debris from fishing activities may be a key source in some areas).
- Improper or ineffective solid waste management.
- Plastic pollution enters the marine environment via rivers, beaches, maritime activities and illegal dumping at sea.

Some specific examples related to the **transport of MPs** are provided below: (i) The wind-driven ocean circulation had an effect on microplastic accumulation at beaches, with higher concentrations reported in sheltered areas than exposed ones (Vianello et al. 2013); (ii) the wind-driven Ekman currents and geostrophic circulation transported floating plastic debris in North Atlantic Subtropical Gyre (Law et al. 2010; Faure et al. 2015a); (iii) The combined effects of ocean circulation pattern, adjacent eddies, the Kuroshio and Kuroshio current were responsible for MPs distribution in the North Pacific Ocean (Yamashita and Tanimura 2007; Pan et al. 2019); (iv) the atmospheric fallout was most likely a transport pathway of MPs to remote Tibet (Zhang et al. 2015); and (v) rain, wind, hurricanes, and flash flooding (Yonkos et al. 2014) and urban rivers (McCormick et al. 2014) caused a major transfer of plastics from land to sea.

7.2.2 MP Composition in Surface Waters

Based on **polymer shapes**, the most commonly detected MPs are fibres (lines, thread, net and ropes), fragments, foams and films. The less commonly detected are pellet, resin and spherules (Table 7.1). Some examples of fibres, fragments, foams and films detected in different surface waterways are highlighted below:

Fibres including lines, nets and ropes of large plastic particles from shipping activity, fishing equipment, recreation and offshore industries (oil, gas) could be broken down into microparticles (Day et al. 1990; Lusher et al. 2015; Abayomi et al. 2017) and reach sewage treatment plant (Cincinelli et al. 2017) and released as atmospheric fallout (Dris et al. 2015) on waterways. Fibres accounted for more than 90% of MPs in various waterways, for example, 97.2% in Greenland (Arctic) (Amelineau et al. 2016), 95% in the Arctic Ocean (Norway) (Lusher et al. 2015) and 94% in the Northeast Atlantic Ocean (Lusher et al. 2014).

Fragments are small pieces of plastic broken from larger pieces. This category includes primarily chips and pieces of sheets (Day et al. 1990). About 92% of MPs found were fragments in Southern California's coastal waters, USA (Moore et al. 2002). In the Kuroshio Current area (Japan), fragments were accounted for 56% of the MPs (Yamashita and Tanimura 2007).

Foams include all pieces of foamed plastic (polystyrene/extruded polystyrene) (Day et al. 1990). Foams are heavily used by fishing boats, seafood markets (Fok et al. 2017) and as Styrofoam buoys for aquaculture (oyster, mussel) (Kim et al. 2015). In the coasts of Guangdong Province in southern China, 96% of MPs found were foams (Fok et al. 2017).

Films are a thin or membrane-like pieces of plastic (Almi et al. 2018). At the southwest of the Bay of Bengal, about 40% of the MPs found were films (Eriksen et al. 2017).

Pellets are spherical pieces of plastic (Almi et al. 2018). Pellets are probably directly released from plants or indirectly via runoff (Hoellein et al. 2014).

Based on **polymer chemistry**, the most commonly detected MPs in surface waters are polyethylene (PE), polypropylene (PP) and polystyrene (PS). The less commonly detected polymers are acrylic, ethylene vinyl acetate, PA (polyamide)/nylon, PET (polyethylene terephthalate), polyurethane, PVC (polyvinyl chloride) and rayon (Table 7.1). PE, PP and PS (which are lightweight, buoyant, float and travel long distances) are used in packaging, office equipment (Reisser et al. 2013; Mani et al. 2015) and vehicle construction (Mani et al. 2015). PE and PP are the main component of fishing nets and lines (Wang and Wang 2018). PP is widely used in food packaging, folders, car bumpers (Plastics Europe 2015), carpets and ropes (Gregory 1996; Zitko and Hanlon 1991). PS is most commonly used in packaging and industrial insulation (Browne et al. 2008, 2011). Some examples of PE, PP and PS detected in different surface waterways are highlighted below:

PE: East Asia Seas, Japan (68% of the polymer) (Isobe et al. 2015); coastal waters, Australia (65% of the polymer) (Reisser et al. 2013); East Asia Seas, Japan (68%

of the polymer) (Isobe et al. 2015); Swiss lakes, Switzerland (62% of the polymer) (Faure et al. 2015b); Northwestern Pacific (~58% of the polymer) (Pan et al. 2019); Three Gorges Reservoir, Yangtze River, China (36.79–57.12% of the polymer) (Zhang et al. 2015); the South Pacific Ocean, Vanuatu (41% of the polymer) (Bakir et al. 2020); Lake Naivasha, Kenya (20% of the polymer) (Migwi et al. 2020); and the Rhine River, Germany (15% of the polymer) (Mani et al. 2015).

PP: Coastal waters, Australia (31% of the polymer) (Reisser et al. 2013); South Pacific Ocean, Vanuatu (24% of the polymer) (Bakir et al. 2020); Northwestern Pacific (36% of the polymer) (Pan et al. 2019); North Pacific Central Gyre (98% of the polymer) (Moore et al. 2001); Lake Naivasha, Kenya (25% of the polymer) (Migwi et al. 2020); East Asia Seas, Japan (19% of the polymer) (Isobe et al. 2015); and Three Gorges Reservoir, Yangtze River, China (42.14–63.21% of the polymer) (Zhang et al. 2015).

PS: Coastal Soya Island, South Korea (99% of the polymer) (Kim et al. 2015); Pearl River Estuary, Hong Kong (92% of the polymer) (Fok and Cheung 2015); the Rhine River, Germany (70% of the polymer) (Mani et al. 2015); and the South Pacific Ocean, Vanuatu (23% of the polymer) (Bakir et al. 2020).

7.2.3 MP Concentration in Surface Waters

MP concentration varied among and between waterways (lakes, rivers, bays, coasts, seas, oceans and gyres) (Table 7.1). The Coastal Soya Island, South Korea (46,334 MP particles/m^2) and the Pearl River Estuary, Hong Kong (5595 MP particles/m^2), are the two MP pollution **'hot spots'** in the world (Fig. 7.1, Table 7.1). Based on various research results, we found that there is a relationship between elevated MPs concentration and closeness of cities/towns (with high population) to lakes, rivers, estuaries and coastal towns, for instance, coasts of Australia (Reisser et al. 2013), the Yangtze River in China (Zhang et al. 2015), Lake Ontario in Canada (Ballent et al. 2016) and Gulf of Lion (Mediterranean Sea) in France (Schmidt et al. 2018). In addition, higher MP concentrations were also found in semi-enclosed basin such as the Mediterranean basin (Cózar et al. 2015), dam/reservoirs (where plastics got accumulated behind the dam) (Zhang et al. 2015) and areas of circular/rotating ocean current called 'gyres' (e.g. North Pacific Central Gyre, Great Pacific Garbage Patch) (Moore et al. 2001; Leberton et al. 2018). In fact, the elevated MP concentrations were found in several lakes, rivers, bays, coastal waters, seas, oceans and gyres across the globe as highlighted below:

Lake/reservoir: Lakes in Siling Co basin, *Tibet* (113.8 MP particles/m^2) (Zhang et al. 2016); Three Gorges Reservoir, the Yangtze River, *China* (8.47 MP particles/m^2) (Zhang et al. 2015); Lake Naivasha, *Kenya* (0.407 MP particles/m^2) (Migwi et al. 2020); and Lake Winnipeg, *Canada* (0.193 MP particles/m^2) (Anderson et al. 2017) (see Table 7.1 for details).

River: Pearl River Estuary, *Hong Kong* (5595 MP particles/m^2) (Fok and Cheung 2015); the Rhine River, *Germany* (0.89277 MP particles/m^2) (Mani et al. 2015); Kaliningrad region, Baltic, *Russia* (42–1150 MP particles/m^2) (Esiukova 2017); and Bootless Bay, *Papua New Guinea* (13.74 MP particles m^2) (Smith 2012) (see Table 7.1 for details).

Bay, coast, sea and ocean: Coastal Soya Island, *South Korea (*46,334 MP particles/m^2*)* (Kim et al. 2015); Coast, Guangdong Province, *China* (6675 MP particles/m^2) (Fok et al. 2017); East Asia Seas, *Japan* (1.72 MP particles/m^2) (Isobe et al. 2015); *Mediterranean* Sea basin (0.244 MP particles/m^2) (Cózar et al. 2015); and Northwestern *Pacific* (0.174 MP particles/m^2) (Yamashita and Tanimura 2007) (see Table 7.1 for details).

Gyre: North *Pacific Central Gyre* (0.334 MP particles/m^2) (Moore et al. 2001) and Great Pacific Garbage Patch/*Pacific Gyre* (0.17 MP particles/m^2) (Leberton et al. 2018) (see Table 7.1 for details).

7.3 Microplastic Pollution in the World Sediments

7.3.1 MP Occurrence and Distribution in Sediments

Microplastic pollutants are widespread and detected in sediments of a wide range of aquatic environment including archipelagos, bays, channel, coasts, beaches, deep seas, estuaries, lagoons, rivers, shellfish farms and ship-breaking yards. They (MP) have been detected in sediments across the globe including the Arctic Ocean, Bangladesh, Belgium, Black sea, Brazil, Canada, China, Fiji, Germany, Ghana, India, Indian Ocean, Indonesia, Iran, Italy, Japan, Maldives, Netherlands, Norway, Pakistan, Qatar, Russia, Singapore, Slovenia, Solomon Islands, South Africa, Tunisia, UK, Vanuatu and Vietnam (Table 7.2 and Fig. 7.1). At some places, MP abundance in sediments reached 64% to 100% of the samples sampled. Those are listed below:

- 100% of the samples: *China* (Changjiang estuary) (Peng et al. 2017); *Germany* (the Rhine and Main Rivers) (Klein et al. 2015); *Russia* (Baltic beaches, Kaliningrad region) (Esiukova 2017); and *Vietnam* (Da Nang beach, Vietnam) (Nguyena et al. 2020)
- 89% of the samples: *UK* (North Sea and English channel) (Maes et al. 2017)
- 81.8% of the samples*:* Fiji (South Pacific Ocean, Suva coastal), Fiji (Ferreira et al. 2020)
- 67% of the samples: *Canada* (creek, lake and beach) (Ballent et al. 2016)
- 64% of the samples: *Slovenia* (beaches along the Slovenian coast) (Laglbauer et al. 2014; Bakir et al. 2020).

The increased abundance of MPs has been detected in sediments of deep seas and aquatic habitats adjacent to harbours, high-population areas, industries, lagoons,

Table 7.2 Microplastics composition, concentrations and occurrences in sediments (archipelagos, bays, channels, coasts, beaches, deep seas, estuaries, lagoons, rivers, shellfish farms, ship-breaking yards)

Country/region and references	Polymer composition (na = data is not available)	Polymer concentration (dw = dry weight)	Remarks (MPs = microplastics)
Arctic Ocean Deep Sea; 2340–5570 m depth (Bergmann et al. 2017)	***Shapes:*** na ***Types:*** Chlorinated polyethylene (38%) > PA (22%) > PP (16%)	***Range***: 42–6595 MP particles/kg dw; ***mean:*** 4356 MP particles/kg dw	The microplastic quantities are among the **highest recorded** from benthic sediments
Bangladesh Cox's Bazar Beach (Rahman et al. 2020)	***Shapes:*** Fragments (64%), foams (15%), fibres (9%), beads (6%) and films (6%) ***Types***: PP (47%), PE (23%), PS (9%), PVC (6%) and PET (3%)	***Range***: 3–12 MP particles/kg dw; ***mean:*** 8.1 MP particles/kg dw	**Tourist** activity is a potential and significant source of MPs in the beach sediments
Belgium Belgian Coast (Claessens et al. 2011)	***Shapes***: Fibres (59%), granules (25%), spherules (12%) and films/ nylon (4%) ***Types:*** PP, PS, PA/ nylon and polyvinyl alcohol	***Range:*** 52.8–390 MP particles/kg dw; ***mean:*** 121.4 MP particles/kg dw	The highest MPs concentrations were found in the **harbour areas** (partially enclosed having reduced flushing rate)
Black Sea (Beach) Romania (Popa et al. 2014; Bakir et al. 2020)	***Shapes***: Fibres ***Types***: na	***Range:*** 100–5000 MP particles/kg dw; ***mean***: na	The most popular **tourist spots** showed the highest MPs pollution
Brazil Marine Beach (Costa et al. 2010; Ballent et al. 2016)	***Shapes***: Fragments (96.7%) and virgin plastic pellets (3.3%) ***Types***: na	***Range***: na; ***mean:*** 310 MP particles/kg dw	The presence of virgin plastic pellets implies long-range marine transport
Canada Shellfish (Scallops, Oysters) Growing Region, British Columbia) (Kazmiruk et al. 2018)	***Shapes:*** Microfibres (major), microbeads and micro fragments ***Types:*** HDPE	***Range***: Up to 25,000 MP particles/kg dw; ***mean:*** na	The shellfish industry is the likely source of the MPs contamination (HDPE used for netting, oyster bags, trays, cages, fences, etc.)

(continued)

Table 7.2 (continued)

Country/region and references	Polymer composition (na = data is not available)	Polymer concentration (dw = dry weight)	Remarks (MPs = microplastics)
Canada Nova Scotia, Beach and Mussel Farms (Mathalon and Hill 2014; Ballent et al. 2016)	***Shapes***: Fibres ***Types***: PP	***Range:*** 2000–8000 MP particles/kg dw; ***mean:*** 4189 MP particles/kg dw	Microfibres (PP) in farmed mussels were found higher compared to wild mussels
Canada Creek, Lake and Beach (Ballent et al. 2016)	***Shapes:*** Fibres and fragments ***Types***: PE (31%), PS (10%), PU (4%), PP (3%), PVC (3%) and polystyrene sulfonate (PSS) (3%)	***Range:*** 20–27,830 MP particles/kg dw; ***mean:*** 870 MP particles/kg dw	1. 67% of samples were plastics 2. **High population density** and industrial activity were responsible for high MPs contamination
China Changjiang Estuary (Peng et al. 2017)	***Shapes:*** Fibres (93%), fragments (6%) and pellets (1%) ***Types:*** Rayon (RY) (63.1%), PES (18.5%), acrylic (AC) (13.9%), PET (1.5%), PS (1.5%) and poly (ethylene-Propylene-Diene) (1.5%)	***Range:*** 20–340 MP particles/kg dw; ***mean***: 121 MP particles/kg dw	1. Plastics were present in **100%** of the samples 2. Detection of synthetic fibres (rayon, polyester and acrylic) in the estuary may indicate MPs were from **laundering** clothes
China Coastal five beaches: Shapawan, Haikou, Wanning, Sanya and Beiha (Qiu et al. 2015)	***Shapes:*** *na* ***Types:*** HDPE, PET, PE and PS	***Range:*** 5020 to 8720 MP pieces/kg dw; ***mean***: 6923 MP particles/kg dw	The high MPs were found at sites near the **tourist areas** and fisherman's wharves (fishing gear and nets, which are usually made of PET)

(continued)

Table 7.2 (continued)

Country/region and references	Polymer composition (na = data is not available)	Polymer concentration (dw = dry weight)	Remarks (MPs = microplastics)
Fiji South Pacific Ocean (Suva Coastal) (Ferreira et al. 2020)	***Shapes:*** Fibres (60.2%), fragments (26.9%), films (9.4%) and microbeads (3.5%) ***Types***: PE (23%), latex (12%), PP (11%), nylon (10%), PETE (9%), PS (8%) and ethylene vinyl acetate (EVA) (8%).	***Range***: na: ***Mean:*** 19.8 MP particles kg dw	1. 81.8% of the sediments samples were MPs 2. Nylon and latex are used in fisheries activities (nets) and EVA (a rubber used) in fisheries gears, shoes and packaging (Emblem 2012; Fabris and Knauss 1989)
Germany Kachelotplate and Spiekeroog Islands Beach Transects (Liebezeit and Dubaish 2012)	***Shapes:*** Fibres ***Types:*** na	***Kachelotplate***: 0–62,100 MP particles/kg dw; ***Spiekeroog:*** 15,000–58,000 MP particles/kg dw; **mean**: 38,000 MP particles/kg dw	1. **Atmospheric transport** and the deposition could be the route of transport of MPs in the two islands 2. MP fibres can be a source from **sewage treatment** plants' discharge
Germany The Rhine and Main Rivers (Klein et al. 2015)	***Shapes:*** Spheres (50%) and fibres (13%) ***Types:*** PE (49%), PP (26%) and PS (10%)	***Rhine River***: 228–3763 MP particles/kg; ***Main River*****:** 786–1368 MP particles/kg	1. MPs were detected in **100%** of the sediments samples 2. The occurrence of PE, PP and PS may indicate the high level of industrial usage of these polymers
Ghana Coastal Lagoons (Chico-Ortiz et al. 2020)	na	***Range:*** na**;** ***mean:*** 17.85 MPs/10 cm^3 (1785 MP particles/kg)	1. MPs abundance and distribution varied with depth (highest at the upper layer) 2. **Mangrove** (with extensive root system) and lagoons (with sand barriers) can trap plastics (Yona et al. 2019; Chico-Ortiz et al. 2020)
India Ship-Breaking Yards (Reddy et al. 2006)	***Shapes***: na ***Types***: Polyurethane, nylon, polystyrene, polyester and glass wool	***Range***: na; ***maximum***: 81 mg MP particles kg of sediments	1. The plastic fragments resulted from the **ship-breaking** activities 2. Polyurethane, nylon, polystyrene, polyester and glass wool are used in constructing ships (insulating materials, fabrics, packaging)

(continued)

Table 7.2 (continued)

Country/region and references	Polymer composition (na = data is not available)	Polymer concentration (dw = dry weight)	Remarks (MPs = microplastics)
Indian Ocean The Chagos Archipelago, Beach (Readman et al. 2013)	***Shapes:*** na ***Types:*** Nylon, PE, PES, PP and rayon	***Range****:* na; ***mean****:* 91 MP particles/kg dw	The study indicates that microplastic can accumulate in remote locations
Indonesia Java Sea (Yona et al. 2019)	***Shapes:*** Fragments (54%), fibres (41.5%) and films (4.21%) ***Types:*** na	***Range*****:** 206–897 MP particle/kg dw; ***mean***: na	1. MPs were found at the highest levels in the **mangrove** area 2. Domestic wastes and fisheries activities were the main causes of the high microplastic particles
Indonesia Jakarta Bay (Manalu et al. 2017)	***Shapes:*** Fragments (major), fibres and pellets ***Types***: PP (major)	***Range:*** 18,405–38,790 MP particles/kg dw; ***mean:*** 30,006 MP particles/kg dw	1. Fragments possibly released from broken plastic bags 2. This finding is much higher than the previous study in the mangrove area of Jakarta (217–2218 MP particles/kg) (Hastuti et al. 2014)
Iran Beaches of the Strait of Hormuz (Naji et al. 2017)	***Shapes***: Fibres (83%), films (11%), fragments (6%) and granules (1%) ***Types***: PE, PA/nylon and PET	***Urban site:*** 122 MP particles/kg dw) (Gorsozan); ***industrial site:*** 1258 MP particles/kg dw (Bostanu)	The industrial site (Bostanu) had 10 times more MPs per kg sediments than the urban site of Gorsozan. This could be related to manufacturing, oil refineries, sewage, parks, ports and fishing ground at Bostanu
Italy Venice Lagoon (Vianello et al. 2013)	***Shapes*****:** Fragments (86%), fibres (11%), films (2%) and pellets (1%) ***Types****:* PE (48.4%), PP (34.1%), polyethylene PP (5.2%), polyester (3.6%), PS (3.5%) and PVC (0.5%)	***Range:*** 672–2175 MP particles/kg dw; ***mean****:* 1445.2 MP particles/kg dw	1. A higher concentration of PE was found near the **industrial zone** (close by the lagoon) and PP fibres near a fishing fleet harbour 2. PE can be from packaging or breakdown of rigid plastics, while PP from plastic tools, furnishings, water and gas pipes
Japan Tokyo Bay (Matsuguma et al. 2017)	***Shapes:*** Fragments (75%), fibres (15%) and beads (4%) ***Types:*** PE, PP (light polymers), PVC and PET (heavy polymers)	**Range**: 1845–5385 MP particles/kg dw; **mean**: 1800 MP particles/kg dw	Inputs from rivers and sewage outfalls increased the abundance of MPs in the sediments

(continued)

Table 7.2 (continued)

Country/region and references	Polymer composition (na = data is not available)	Polymer concentration (dw = dry weight)	Remarks (MPs = microplastics)
Maldives Coral Island (Patti et al. 2020)	***Shapes:*** Fragments (51%) *and* filaments (49%) **Types**: na	**Range:** 226–333 MP particles/kg dw; **mean:** 277 MP particles /kg dw	The inadequate treatment of sewage and discharge of raw sewage could be the main source of MPs pollution in the Maldives
Netherlands Seacoast (Leslie et al. 2017)	***Shapes:*** Fibres (major), foils and spheres ***Types:*** na	***Range:*** 100–3600 MP particle/kg dw; ***mean:*** 2078 MP particle/kg dw	The study showed that marine sediments act as sinks for microplastics
Norway Barents Sea, Central North Sea, Northern North Sea; 100–500 m depths (Norwegian Environment Agency 2018)	***Shapes:*** na ***Types:*** Chlorinated polyethylene, PA, PET, phenoxy resin and rubber materials	***Range*** (all areas): 0–29,020 MP particles/kg dw; ***mean****:* 4408 MP particles/kg dw	1. MPs can be found even in **deep-sea** sediments transported by ocean currents and marine activities (oil and gas, mining, fishing and shipping) 2. Sediments are the ultimate environmental sink. MPs can be a threat to benthic ecosystems and organisms
Pakistan Ravi River, Lahore (Irfan et al. 2020)	***Shapes:*** Fragments (83.1%), fibres (11.8), sheets (1.3%), foams (3.4%) and beads (0.4%) ***Types:*** PE, PP and PS	***Range*****:** na; ***mean:*** 3726 MP particles/m^2	Effluents from industries and municipal sewerage may have caused MP pollution in the Ravi River
Qatar Arabian Bay; Beaches (Abayomi et al. 2017)	***Shapes****:* Fibres (43.8%), films (40.7%) and fragments (14.2%) ***Types****:* PE (LDPE), PP and PET	***Range:*** 6–38 MP particles/kg dw; ***mean:*** 13.5 MP particles/kg dw	1. Low-density PE and PP were the dominant polymers 2. **Fishing lines**, nets and ropes were the main sources of fibres
Russia Baltic Beaches, Kaliningrad Region (Esiukova 2017)	***Shapes:*** Foam, fragments, films and pellets ***Types:*** PP, foamed PE, foamed PS, PVC and foamed polyurethane	***Range****:* 0.2–175.3 MP particles/kg dw; ***mean:*** 1.3–36.3 MP particles/kg dw	1. Plastics were found in **100%** of beach sediments sampled 2. MPs pollution on the beaches was both sea- and land-based (construction, tourism, recreational, waste dumps, shipping and fishing) 3. The most prevalent type of plastic pollution was foam plastic

(continued)

Table 7.2 (continued)

Country/region and references	Polymer composition (na = data is not available)	Polymer concentration (dw = dry weight)	Remarks (MPs = microplastics)
Singapore Coast (Nor and Obbard 2014)	***Shapes:*** Fibres (72.0%), films (23.3%) and granules (4.7%) ***Types:*** PE, PP, PVC and PA/nylon	***Range:*** 12–62.7 MP particles/kg dw; ***mean*:** 36.8 MP particles/kg dw	PP, PVC and nylon were the major polymers identified: PP fibres (ropes, nonwoven fabrics, air filters, diapers, fishing nets), PVC fibres (bonding agents for nonwoven fabrics and products, clothing), nylon fibres (clothes, ropes, fishing lines and fishing nets), films (PE, PP) (plastic wrapping and bags) and granule particles (PP, PE) (personal care products) (Cole et al. 2014)
Slovenia Beaches along the Slovenian Coast (Laglbauer et al. 2014; Bakir et al. 2020)	***Shapes:*** Fibres, fragments and films ***Types*:** na	***Range:*** 170.4–177.8 MP particles/kg dw; ***mean:*** 174.1 MP particles/kg dw	64% of the samples were plastics, and land-based inputs were an important source of micro-debris
Solomon Islands South Pacific Ocean (Bakir et al. 2020)	na	***Range*:** 450–15,167 MP particles/kg dw; ***mean:*** na	Results have indicated the widespread occurrence of microplastics
Sri Lanka Coastal Beaches and Waters (Koongolla et al. 2018)	***Shapes***: Fragments ***Types***: PE, PP and PS	***Range:*** 0–57 MP particles/m^2; ***mean:*** 70 MP particles/m^2 (20 m distance)	1. MP particles found in 60% of sand (sediment) samples 2. PE and PP were the major polymer types identified, which could have been originated from plastic bags, bottles, beverage container caps and drinking straws, and fishing (nets, ropes)
South Africa Five Urban Estuaries, KwaZulu-Natal (Naidoo et al. 2015; Ballent et al. 2016)	***Shapes*:** Fragments (59%) and fibres (38%) ***Types:*** PE and PP (80%)	***Range***: na; ***mean:*** 1165 MP particles/kg dw	Sediments at the harbour had the highest average plastic concentrations

(continued)

Table 7.2 (continued)

Country/region and references	Polymer composition (na = data is not available)	Polymer concentration (dw = dry weight)	Remarks (MPs = microplastics)
Tunisia Lagoon-Channel of Bizerte (Abidli et al. 2017)	***Shapes:*** Fibres (88.8%) and fragments ***Types:*** na	***Range:***3000–18,000 MP particles/kg dw; ***mean:*** 7960 MP particles/kg	The abundance of MPs found in the sediments (7960 particles/kg) may signify they can enter the food chain through ingestion by bivalves and fish and finally to humans via consumption of seafood
UK North Sea and English Channel (Maes et al. 2017)	na	***Range:*** 0–3146 MP particles/kg dw; ***mean:*** 421 MP particles/kg	MPs particles were found in 89% of the sediments
Vanuatu South Pacific Ocean (Bakir et al. 2020)	na	***Range:*** 333–33,000 MP particles/kg dw; ***mean:*** na	The highest concentration of 33,000 MP particles/kg dw is higher than Europe and America (Manalu et al. 2017; Bakir et al. 2020)
Vietnam Da Nang Beach (Nguyena et al. 2020)	***Shapes***: Fibres (99.2%) and fragments (0.8%) ***Types***: na	***Range:*** 9000–11,000 MP particles/kg dw; ***mean:*** 9238 MP particles/kg dw	1. MPs were detected in **100%** of the sediments samples 2. Possible sources of microfibres are discharge of domestic wastewaters and effluents from the textile and garment industry

mangroves, tourist places, shellfish farms and ship-breaking yards as detailed below: It reveals that sediments act as sinks for microplastics (Leslie et al. 2017).

Coral island: *Maldives* (Indian Ocean) (mean: 277 MP particles/kg dw) (Patti et al. 2020).

Deep sea: *Arctic Ocean* (2340–5570 m depth) (mean: 4356 MP particles/kg dw) (Bergmann et al. 2017); and *Norway* (Barents Sea, Central North Sea, Northern North Sea; 100–500 m depth) (mean: 4408 MP particles/kg dw) (Norwegian Environment Agency 2018).

Harbours: *Belgium* (Belgian coast) (mean: 121.4 MP particles/kg dw) (Claessens et al. 2011) and South Africa (urban estuaries, KwaZulu-Natal) (mean: 1165 MP particles/kg dw) (Naidoo et al. 2015; Ballent et al. 2016).

High-population density areas: *Canada* (creek, lake and beach) (mean: 870 MP particles/kg dw) (Ballent et al. 2016).

Industries: *Iran* (beaches of the Strait of Hormuz) (mean: 1258 MP particles/kg dw) (Naji et al. 2017); *Italy* (Venice lagoon) (mean: 1445.2 MP particles/kg dw) (Vianello et al. 2013); and *Pakistan* (Ravi River, Lahore) (mean: 3726 MP particles/m^2) (Irfan et al. 2020).

Lagoons: *Ghana* (coastal lagoons) (mean: 1785 MP particles/kg dw) (Chico-Ortiz et al. 2020) and *Tunisia* (Lagoon-Channel of Bizerte) (mean: 7960 MP particles/kg) (Abidli et al. 2017).

Mangroves: *Ghana* (coastal lagoons) (mean: 1785 MP particles/kg dw) (Chico-Ortiz et al. 2020) and *Indonesia* (Java Sea) (range: 206–897 MP particle/kg dw) (Yona et al. 2019).

Remote locations: *Indian Ocean* (the Chagos Archipelago, beach) (mean: 91 MP particles/kg dw) (Readman et al. 2013).

Tourism: *Bangladesh* (Cox's Bazar beach) (mean: 8.1 MP particles/kg dw) (Rahman et al. 2020); *Black sea* (beach), Romania (range: 100–5000 MP particles/kg dw) (Popa et al. 2014; Bakir et al. 2020); and *China* (coastal five beaches) (mean: 6923 MP particles/kg dw) (Qiu et al. 2015).

Shellfish farms: *Canada* (scallops, oyster growing region) (range: up to 25,000 MP particles/kg dw) (Kazmiruk et al. 2018) and *Canada* (beach and mussel farms) (mean: 4189 MP particles/kg dw) (Mathalon and Hill 2014; Ballent et al. 2016).

Ship-breaking yards: *India* (Alang-Sosiya Ship-breaking yards) (mean: 81 mg MP particles/kg dw) (Reddy et al. 2006).

7.3.2 MP Composition in Sediments

Based on **polymer shapes**, the most commonly detected MPs in sediments are fibres, fragments, and films. The less commonly detected MPs are foams, microbeads, granules, spherules, nylon, rayon, PES and acrylic (Table 7.2). Some examples of fibres, fragments and films detected in sediments are described below:

Fibres: 99.2% in Da Nang beach, Vietnam (Nguyena et al. 2020); 93% in Changjiang estuary, China (Peng et al. 2017); 88.8% in Lagoon-Channel of Bizerte, Tunisia (Abidli et al. 2017); 83% in beaches of the Strait of Hormuz, Iran (Naji et al. 2017); 72% in coastal areas of Singapore (Nor and Obbard 2014); 60.2% in South Pacific Ocean, Fiji (Ferreira et al. 2020); 59% in Belgian coast, Belgium (Claessens et al. 2011); 43.8% in Arabian Bay, Qatar (Abayomi et al. 2017); 41.5% in Java Sea, Indonesia (Yona et al. 2019); 38% in urban estuaries, KwaZulu-Natal, South Africa (Naidoo et al. 2015; Ballent et al. 2016); and 9% in Cox's Bazar beach, Bangladesh (Rahman et al. 2020).

Fragments: 96.7% in Marine beach, Brazil (Costa et al. 2010; Ballent et al. 2016); 86% in Venice lagoon, Italy (Vianello et al. 2013); 83.1% in Ravi River, Lahore, Pakistan (Irfan et al. 2020); 75% in Tokyo Bay, Japan (Matsuguma et al. 2017); 64% in Cox's Bazar beach, Bangladesh (Rahman et al. 2020); 59% in urban estuaries, KwaZulu-Natal, South Africa (Naidoo et al. 2015; Ballent et al. 2016); 54% in the Java Sea, Indonesia (Yona et al. 2019); and 26.9% in South Pacific Ocean, Fiji (Ferreira et al. 2020).

Films: 40.7% in the Arabian Bay, Qatar (Abayomi et al. 2017); 23.3% in coastal areas of Singapore (Nor and Obbard 2014); 11% in beaches of the Strait of Hormuz, Iran (Naji et al. 2017); and 6% in Cox's Bazar beach, Bangladesh (Rahman et al. 2020).

Based on **polymer chemistry**, the most commonly detected MPs in sediments are polyethylene (PE), polypropylene (PP) and polystyrene (PS). The less commonly detected polymers are acrylic, PA (polyamide)/nylon, PET (polyethylene terephthalate), PU (polyurethane) and PVC (polyvinyl chloride). Some examples of PE, PP and PS detected in sediments are described below:

PE: 49% in the Rhine and Main Rivers, Germany (Klein et al. 2015); 48.4% in Venice lagoon, Italy (Vianello et al. 2013); 38% in deep sea, 2340–5570 m depth, Arctic Ocean (Bergmann et al. 2017); 31% in the creek, lake and beach, Canada (Ballent et al. 2016); 23% in Cox's Bazar beach, Bangladesh (Rahman et al. 2020); 23% in South Pacific Ocean, Fiji (Ferreira et al. 2020); and 3% in the creek, lake and beach, Canada (Ballent et al. 2016).

PP: 47% in Cox's Bazar beach, Bangladesh (Rahman et al. 2020); 34.1% in Venice lagoon, Italy (Vianello et al. 2013); 26% in the Rhine and Main Rivers, Germany (Klein et al. 2015); 16% in deep sea, 2340–5570 m depth, Arctic Ocean (Bergmann et al. 2017); and 11% in South Pacific Ocean, Fiji (Ferreira et al. 2020).

PS: 18.5% in Changjiang estuary, China (Peng et al. 2017); 10% in the Rhine and Main Rivers, Germany (Klein et al. 2015); 9% in Cox's Bazar beach, Bangladesh (Rahman et al. 2020); 10% in the creek, lake and beach, Canada (Ballent et al. 2016); and 3.5% in Venice lagoon, Italy (Vianello et al. 2013).

7.3.3 MP Concentration in Sediments

MP concentration varied among and between archipelagos, bays, channel, coasts, beaches, deep seas, estuaries, lagoons, rivers, shellfish farms and ship-breaking yards (Table 7.2). The Kachelotplate and Spiekeroog islands, Germany (38,000 MP particles/kg dw sediments) (Liebezeit and Dubaish 2012), and Jakarta Bay, Indonesia (30,006 MP particles/kg dw sediments), are the two MP pollution **'hot spots'** in the world (Fig. 7.1 and Table 7.2). Elevated MP concentrations were also found in sediments of deep seas, harbours, populated cities, industrial areas, lagoons, mangroves, tourist areas, remote oceans, shellfish farms and ship-breaking yards (see Table 7.2 for the concentrations detected in sediments).

7.4 Conclusion

Microplastic (MP) pollutants are widespread in both surface waters and sediments across the globe. MPs have been detected in surface waters (creeks, lakes, channels, rivers, estuaries, coasts, oceans and gyres). Based on *polymer shapes*, the most commonly detected MPs in both surface waters and sediments are fibres (lines, thread, net and ropes), fragments, foams and films. Based on *polymer chemistry*, the most commonly detected MPs in surface waters and sediments are polyethylene (PE), polypropylene (PP) and polystyrene (PS) (see Sects. 7.2 and 7.3 for MPs composition in surface waters and sediments, respectively).

At some places, MP abundance reached 90% to 100% of the samples sampled in surface waters. The increased abundance of MPs was reported in surface waters close to *larger cities* and cities with higher population density (Asia, Brazil), *enclosed basins* (with narrow outlet/inlet, e.g. the Mediterranean Sea), *gyres* (with circular/rotating ocean current*;* e.g. North Pacific Central Gyre, Great Pacific Garbage Patch), *dams/reservoirs* (where plastics got accumulated behind the dam; e.g. Three Gorges dam in China), *rivers* (Pearl River Estuary, Hong Kong, Yangtze River, China), *coastal areas* (Soya Island, South Korea, and Guangdong Province, China) and adjacent to horticultural, agricultural, fishing and tourism activities. MPs concentration varied among and between waterways (lakes, rivers, bays, coasts, seas, oceans and gyres) (Table 7.1). The Coastal Soya Island, South Korea (46,334 MP particles/m^2), and the Pearl River Estuary, Hong Kong (5595 particles/m^2), are the two MP pollution **'hot spots'** in surface waters. Nonetheless, MPs pollutants have reached even the remote and pristine parts of the world (the Arctic, the Antarctic, Hawaiian Islands, Tibet). Moreover, there have been more plastics than fish or plankton in various waterways (e.g. plastic litters outnumbered fish larvae in the Danube River, Austria; MPs were present in 61% of zooplankton samples in Portuguese coastal waters; and the mass of plastic was approximately six times that of plankton in the North Pacific Central Gyre).

MP pollutants are also widespread and detected in sediments of a wide range of aquatic environments including archipelagos, bays, channel, coasts, beaches, deep seas, estuaries, lagoons, rivers, shellfish farms and ship-breaking yards (Table 7.2). At some places, MP abundance in sediments reached 64% to 100% of the samples sampled. The increased abundance of MPs has been detected in sediments of deep seas and aquatic habitats adjacent to harbours, high-population areas, industries, lagoons, mangroves, tourism, shellfish farms and ship-breaking yards. The Kachelotplate and Spiekeroog islands, Germany (38,000 MP particles/kg dw sediments) (Liebezeit and Dubaish 2012), and Jakarta Bay, Indonesia (30,006 MP particles/kg dw sediments), are the two MP pollution **'hot spots'** in sediments.

To reduce impacts from plastic wastes and plastic pollution, a number of measures can be undertaken including (i) 3Rs (reduce, reuse and recycle plastics) strategy; (ii) use of alternatives to plastic (jute bags, environmentally friendly bags, reuse bags); (iii) Use of biodegradable or bioplastic; (iv) imposing ban and levy tax on using single use of plastic bags; (v) implementing international agreements, such as the Annex V of the MARPOL (prevent garbage discharging directly into the sea from ships including plastics), are completely put into practice (*https://www.imo.org/en/OurWork/Environment/Pages/Garbage-Default.aspx#:~:text=*); (vi) use of efficient wastewater treatment plant that can remove nanoplastics, microplastics and microbeads; (vii) promoting awareness education at schools, colleges and universities reflecting harms caused by plastic pollution; and (viii) Monitoring of plastics in waterways, seafood, other foods and drinking water (Kibria et al. 2021). For further reading, please consult the works of Pérez-Silva et al. (2020) on Remediation of Contaminated Waters with Microplastics.

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Chapter 8
Microplastic Pollution in the Black Sea: An Overview of the Current Situation

Levent Bat and Ayşah Öztekin

Abstract Microplastic pollution is one of the most important problems of today. The prevalence of microplastics in the marine environment, as well as their consequences on marine biota, is evident. The Black Sea has been rapidly polluted in recent years and has been described as one of the most affected areas. Microplastics are quite intense from the data obtained from the studies on marine litter and microplastics in the Black Sea. It is also very important to have information about the distribution and sources of microplastics for the Black Sea Region. The purpose of this review is to provide a general assessment of the microplastic pollution of the Black Sea.

Keywords Microplastic · Contaminant · Wastes · Marine litter · The Black Sea

Abbreviations

BS	Black Sea
PC	Polycarbonate
PVC	Polyvinyl chloride
CA	Cellulose acetate
PE	Polyethylene
PVF	Polyvinyl fluoride
ER	Epoxy resins
PET	Polyethylene terephthalate
PMMA	Poly(methyl methacrylate)

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EVA	Ethylene-vinyl acetate
PP	Polypropylene
PTFE	Polytetrafluoroethylene
PA	Polyamide
PS	Polystyrene
PAC	Polyacrylic
PAN	Polyacrylonitrile

8.1 Introduction

Microplastic contamination is one of the most pressing issues of our time. The presence of microplastics in coastal ecosystem and their effects on marine biota is an undeniable fact. In recent years, the Black Sea has been increasingly polluted, and it has been identified as one of the most afflicted locations. Microplastics are quite intense from the data obtained from the studies on microplastics and marine litter in the Black Sea. This is also very considerable to have information about the distribution and sources of microplastics for the Black Sea Region. The goal of this work is to offer a broad appraisal of the Black Sea's microplastic pollution.

8.2 Microplastics

Plastics are among the most frequently utilized goods on the planet; they are deeply embedded in today's culture and contribute significantly to practically every product category (Hammer et al. 2012). Plastics have a variety of useful benefits, inexpensive, flexible and long-lasting, and because of these benefits, production of these material has increased over the last few years remarkably (Andrady and Neal 2009; Hopewell et al. 2009). The longevity of plastics, as well as their expanding use, presents a substantial management challenge (Thompson et al. 2009). Part of this is recycled, but large amounts of plastics have concentrated in the ecosystem and landfills (Thompson et al. 2009).

Plastics are the variest common type of marine litter, accounting for 60–80% of all marine litter today (Derraik 2002; Gregory and Ryan 1997).

The accumulation and fragmentation of plastics is one of the most important problems in recent times (Barnes et al. 2009). Most plastics are degraded as soon as they enter the environment due to a variety of circumstances (Hammer et al. 2012). Microplastics are manmade polymer particles with a diameter of less than 5 mm (Arthur et al. 2009). There is still no agreement on the upper (5 mm or 1 mm) and lower (1 μm or 20 μm) size limits to microplastics. Newly, Frias and Nash (2019) recently suggested a description for microplastics: 'Microplastics are any synthetic solid particle or polymeric matrix, with regular or irregular shape and with size ranging from 1 μm to 5 mm, of either primary or secondary manufacturing origin, which are insoluble in water'.

Microplastics are classified as elementary and secondary microplastics. Elementary microplastics are manufactured to be of microscopic size such as plastic pellets (or nurdles) used in industrial manufacturing and microbeads found in personal care products. Other microplastics are formed by the breakdown of larger plastic (Rogers 2020; Hidalgo-Ruz et al. 2012; Cole et al. 2011). It has been reported that microplastics are present in all types of environments around the world in marine (Van Cauwenberghe et al. 2013; Desforges et al. 2014), freshwater systems (Eriksen et al. 2013; Free et al. 2014) and a range of aquatic organisms from zooplankton (Desforges et al. 2015) to invertebrates (Li et al. 2016), fish (Lusher et al. 2013), sea birds (Carlin et al. 2020) and sea mammals (Nelms et al. 2019). Microplastic consumption (Cole et al. 2013) and trophic transfers (Setälä et al. 2014; Farrell and Nelson 2013) have been reported with experimental studies. It is known that a wide variety of additives may be added to the rosin to raise the yield and view during the production (Andrady and Neal 2009; Napper and Thompson 2018) and plastics also adsorb and intensify contaminants from the ambient environment (Teuten et al. 2009).

The presence of microplastics in marine ecosystem and their effects on marine biota is an undeniable fact. The aforementioned pollution status in the region is quite intense from the data obtained from the studies on marine litter and microplastics in the Black Sea. It is also very prominent to have information about the distribution and sources of microplastics in the Black Sea coasts. One of the most damaged places has been identified as the Black Sea.

8.3 The Black Sea

The Black Sea (BS) lies between 28° and 42° East longitudes and 41° and 46° North latitudes (Fig. 8.1). Ukraine, Russia, Romania, Bulgaria, Turkey and Georgia all border the Black Sea. Just a small opening connects it to the Mediterranean Sea (Bosphorus Strait). It is a semi-closed sea, its natural circulation is very low and its self-cleaning ability is limited.

The BS is one of the world's most spectacular seas, with incredible diversity and richness. It connects and divides the continents of Europe and Asia. The countries of the BS basin are home to almost 200 million people who place heavy demands on the basin's resources. However, since the 1960s, because of green threat, the BS has been appallingly abused (Mee 2005).

Waste from villages, fields and factories flows into the BS; some are source directly from the coasts, but the majority comes from the Europe's second, third and fourth rivers the Danube, Dnieper and Don and other rivers notably Dniester, Southern Bug, Chorokh, Rioni, Sakarya, Kizilirmak and Yesilirmak (Bat 2014, 2017; Bat et al. 2018). The plastic input was accounted to 4.2 tonnes/day (Lechner et al. 2014), and annually input of microplastics was estimated at approximately 2 trillion particles and 500 tonnes by River Danube to the BS (van der Wal et al. 2015).

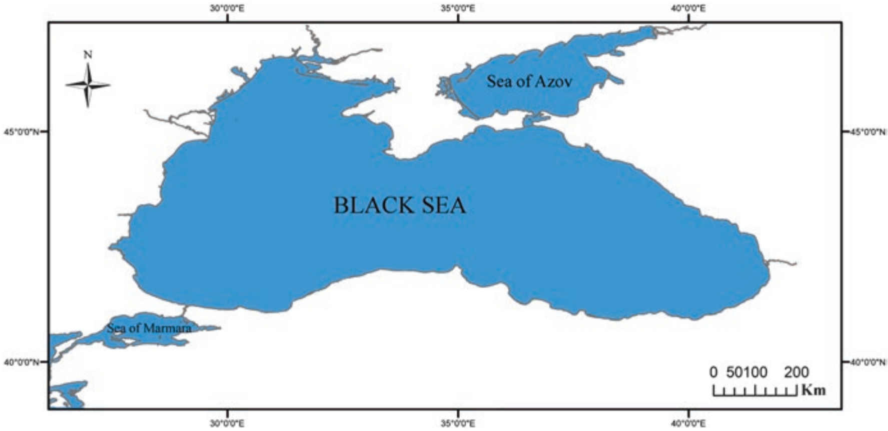

Fig. 8.1 The BS (adopted from Bat et al. 2018)

All types of waste are used, stored and transported in inefficient ways all over the BS (Bat et al. 2018). They then found their way (via rivers) and finally ended up in the BS. This waste could be local or transported from afar (Öztekin and Bat 2017; Öztekin et al. 2020). Furthermore, the BS's dynamic current system allows for cross-border trash movement (Topcu and Ozturk 2010). The constant strain from fishing activities, along with irresponsible touristic activity, is ruining the natural world.

The BS has been rapidly polluted, especially in recent years, by uncontrolled fishing and shipping, dumping of toxic wastes, runoff domestic waste from coastal cities and pollutants from rivers (BSC 2009). Several governmental and private institutions performed marine litter investigations using various approaches and methods by countries that have a coast on the Black Sea (UNEP 2009).

In the BS, marine litter (Öztekin et al. 2020; Terzi et al. 2020; Simeonova et al. 2017; Suaria et al. 2015) and microplastic (Aytan et al. 2020a; Pojar et al. 2021a, b) contamination had been reported by many researchers in the beaches, seafloor and sea surface, and distribution and accumulation patterns of floating litter and microplastics were researched with modelling techniques (Miladinova et al. 2020a, b; Stanev and Ricker 2019).

They cause significant problems in the BS and are potentially fatal to marine life. Many seabirds, fish and other aquatic organisms are certain to ingest plastics (Thiel et al. 2018). Even if the plastic is not toxic, it may cause animal death by obstructing its digestive system. The presence of plastics had been investigated in biota samples in the BS (Şentürk et al. 2020; Tonay et al. 2020). The seriousness of this form of marine contamination should not be underestimated. The Black Sea is one of the world's most endangered marine ecosystems, which comes as no surprise (Bat 2014, 2017; Bat et al. 2018).

8.4 The Current Status of Microplastic Pollution in the Black Sea

In this review, articles investigating the presence of microplastics in the BS region are compiled by accessible articles. Microplastic pollution in the BS were reported by various researchers in the sea surface (Aytan et al. 2016, 2020a; Berov and Klayn 2020; Mukhanov et al. 2019; Öztekin and Bat 2017; Pojar et al. 2021a; Totoiu et al. 2020), water column (Öztekin and Bat 2017; Aytan et al. 2020a), bottom sediment (Aytan et al. 2020a; Cincinelli et al. 2021; Pojar et al. 2021b) and beach sediment (Popa et al. 2014; Şener et al. 2019), and these studies are given extensively in Table 8.1.

8.5 Abundance

The microplastic contamination of surface waters was assessed by various researchers (Aytan et al. 2016, 2020a; Berov and Klayn 2020; Mukhanov et al. 2019; Öztekin and Bat 2017; Pojar et al. 2021a; Totoiu et al. 2020), and they are relatively large in number than sediment and column research. The results are given as particle/km^2, particle/m^3, g/m^3 and μg/m^3. Microplastic concentration was minimum 0.62(±0.73) par/m^3 in Western BS (Bulgaria-Berov and Klayn 2020) and maximum 1.1×10^3 ($\pm 0.9 \times 10^3$) par/m^3 in Southeastern BS (Turkey-Aytan et al. 2016); microplastic concentration in par/km^2 was 4.62×10^4 ($\pm 5.47 \times 10^4$) in Western BS (Bulgaria-Berov and Klayn 2020) and max 0.178×10^6 to 4×10^6 par/km^2 in the Southeastern BS (Turkey-Aytan et al. 2020a).

The microplastic contamination of the BS column waters was assessed by Öztekin and Bat (2017) and Aytan et al. (2020a). The results are given as particle/L and particle/m^3. Microplastic concentration of water column was 20 par/L in the Southeastern BS (Turkey-Aytan et al. 2020a) and 24.48 ± 26.15 par/m^3 in the Southern BS (Turkey-Öztekin and Bat 2017).

The contamination of the BS sediments with microplastic was assessed by various researchers in sea bottom, littoral area and beach sediments (Cincinelli et al. 2021; Aytan et al. 2020a; Pojar et al. 2021b; Popa et al. 2014; Săvucă et al. 2017; Şener et al. 2019). The results are given as particle/g, particle/kg, particle/m^2 and particle/ml. Microplastic concentration was between 106.7 (BS-Cincinelli et al. 2021) and 159.2 ± 138.4 par/kg and 74.1–1778.8 par/m^2 and 0.004–0.192 par/ml (Southeastern BS-Aytan et al. 2020a) in the BS bottom sediment and 3.6–6.4 microfibers. 100 g/sediment and 20.7 par/kg dry wt. in the beach sediment. In general, the abundance data are not yet standardized, so this can lead to preventing complete comparability of data.

Table 8.1 Distribution of microplastics in the BS

Location	Area	Sample type	Detection method	Abundance	Polymer type	Type	Size groups	Colour	References
Southeastern BS–Turkey	Surface water	WP2 net (200 μm)	Microscopic identification	1.1×10^3 ($\pm 0.9 \times 10^3$) par/m^3		Fibres: 49.4% Plastic films: 30.6% Fragments: 20%			Aytan et al. (2016)
Western BS–Bulgaria	Surface water	Manta net (300 μm)	Microscopic identification	4.62×10^4 ($\pm 5.47 \times 10^4$) par/km^2 0.62(±0.73) par/m^3 9.17×10^{-4} ($\pm 2.04 \times 10^{-3}$) g/m^3		Fragments, fibres/ filaments, films, foams, granules and pellets			Berov and Klayn (2020)
Northern BS–Sevastopol	Surface water	Manta net (300 μm)	Microscopic identification Scanning	0.6–7 par/m^3 6–750 μg/m^3					Mukhanov et al. (2019)

Location	Area	Sample type	Detection method	Abundance	Polymer type	Type	Size groups	Colour	References
Southern BS–Turkey	Surface water	Manta net (300 μm)	Microscopic identification	2.67 ± 2.33 par/m^3		Paint particles: 51.56% Fibre: 29.99% Fragment: 10.16% Nylon: 3.49% Polystyrene: 1.31% Others: 3.44%		White: 40.04% Blue: 14.35% Red: 12.74% Black: 10.70% Grey: 8.52% Green: 6.52% Transparent: 3.91% Yellow: 2.87% Purple: 0.35%	Öztekin and Bat (2017)
Western BS–Romania	Surface water	Neuston net (200 μm)	Microscopic identification Pyrolysis GC-MS	7 par/m^3	PP: 75.8% PAN: 12.1% PS: 9.1% PA-6: 3%	Fibres: 74.6% Fibre clumps: 1.5% Foils: 12.7% Fragments: 11.1% Spherules: 0.1%		Black: 46.6% White: 24.6% Blue: 16% Green: 6% Red: 4% Yellow: 3%	Pojar et al. (2021a)

(continued)

Table 8.1 (continued)

Location	Area	Sample type	Detection method	Abundance	Polymer type	Type	Size groups	Colour	References
Southeastern BS–Turkey	Surface water	Manta net (333 μm)	Microscopic identification FT-IR and SEM/EDS	1.783–40.03 par/m^3 0.178×10^6–4×10^6 par/km^2	PE: 44% PP: 22.6% PAC: 14.3% PET: 8.3% PS: 4.8% PS/PAC: 3.6% SBR: 1.2% PA: 1.2%	Fragments: 49% Films: 31.3% Fibres: 17.7% Foams: 1.9% Beads: 0.1%	The average size (mm) Fragments: 1.540 ± 1.065 Films: 1.984 ± 1.022 Fibres: 2.076 ± 1.205 Foams: 2.302 ± 1.225 Beads: 0.670 ± 0.245	White: 34.3% Transparent: 28.9% Blue: 11.8%	Aytan et al. (2020a)
BS–Romania Bulgaria Turkey	Surface water	Neuston net (200 μm)	Microscopic identification (1–5 mm)	Romania: 6.35–78.9 par/m^3 Bulgaria: 2.75–99.45 par/m^3 Turkey: 2.85–45.58 par/m^3					Totoiu et al. (2020)

Location	Area	Sample type	Detection method	Abundance	Polymer type	Type	Size groups	Colour	References
Southern BS–Turkey	Water column	Cylindro-conical plankton net (300 μm)	Microscopic identification	24.48 ± 26.15 par/m^3		Paint particles: 55.10% Fibre: 30.15% Fragment: 9.33% Nylon: 2.39% Others: 3.04%		White: 37.80% Grey: 20.09% Blue: 15.98% Transparent: 11.45% Red: 5.83% Green: 5.62% Black: 1.94% Yellow: 0.86% Brown: 0.43%	Öztekin and Bat (2017)
Southeastern BS–Turkey	Water column	Niskin bottle	Microscopic identification FT-IR and SEM/EDS	20 par/L^1		Fibres: 42.8% Fragments: 35.5% Films: 21.5% Foams: 0.2% Microbeads: 0.05%			Aytan et al. (2020a)

(continued)

Table 8.1 (continued)

Location	Area	Sample type	Detection method	Abundance	Polymer type	Type	Size groups	Colour	References
BS	Sediment	van Veen grab–box corer	2D imaging FTIR	106.7 par/kg	PE/PP: 44.5% PA: 32.0% acrylates and PE-acrylate copolymers: 13.3% Cellulose: 3.9% PVC: 4.7% not-identified polymers: 1.6%	Fibre dominant		Black: 47.7% Blue: 28.1% Light blue: 4.7% Red: 7% Violet: 3.1% Green: 3.9% Transparent: 5.5%	Cincinelli et al. (2021)
Southeastern BS–Turkey	Sediment	Box core	Microscopic identification FT-IR and SEM/EDS	74.1–1778.8 par/m^2 0.004–0.192 par/ml	PE: 57.1% PP: 28.6% PA: 14.3%	Fibres: 66.4% Fragments: 19.9% Films: 13.3% Beads: 0.4%	The average size (mm) Fibres: 1.253 ± 0.954 Fragments: 1.035 ± 0.429 Films: 1.358 ± 0.892 Beads:0.079	Blue: 40.7% Red: 23.5% Transparent: 15.9%	Aytan et al. (2020a)

Location	Area	Sample type	Detection method	Abundance	Polymer type	Type	Size groups	Colour	References
Western BS (coast)–Danube River	Sediment	van Veen grab–manual grab	Microscopic identification Pyrolysis GC-MS	159.2 ± 138.4 par/kg	PET: 31% PP: 26% PS: 18% PAN: 13% PMMA: 4% PTFE: 4% ER: 4%	Danube Delta Fibres: >90% Fragments and clumps: ~3% BS coast Flakes: 80%			Pojar et al. (2021b)
Western BS–Romania	Beach sediment		Microscopic identification	40–218 fiber/beach					Popa et al. (2014)
Western BS–Romania	Littoral area–sediment	Bulk sample	Microscopic identification	3.6–6.4 microfibers.100 g/sediment[1]		Fibres			Săvucă et al. (2017)
Southwestern BS–Turkey	Beach sediment	Bulk sample (1–5 mm)		20.7 par/kg d.w.					Şener et al. (2019)

8.6 Sampling

In the marine environment (sediment, water column, sea surface), sampling of microplastics requires various approaches: samples can be selective (items are collected directly from the environment of items), bulk (the whole volume of the sample is taken without being reduced) or volume-reduced (during sampling, the volume of the bulk sample is frequently reduced for subsequent analysis) (Hidalgo-Ruz et al. 2012). Various researchers used different sampling techniques during their research. The microplastic investigations in the sea surface were sampled by using Manta, Neuston and WP-2 nets. The water column samples were obtained with plankton nets and Niskin bottles. The mesh sizes of nets were mostly 200–300 µm. The mesh sizes of nets limit the size of the retained particles as well as filterable volume, and these nets also allow relatively large volumes of water to be sampled (Dris et al. 2018). The microplastic investigations in sediment samples were investigated with grabs box cores, van Veen grabs, manual grab or bulk sampling. Density separation is used for extraction of microplastics from sediment and mostly used saturated sodium chloride (NaCl) solution.

8.7 Analysing

Microplastics may occur in environment in a variety of shapes and sizes (Rodríguez-Seijo and Pereira 2017). Various organizations and researchers have made suggestions for sampling and analysing of microplastics (MSFD Technical Subgroup on Marine Litter 2013; Masura et al. 2015; Frias et al. 2018). Visual sorting is often used for naming of microplastics (Hidalgo-Ruz et al. 2012). The naming of microplastics is made according to their type (fibre, fragment, pellet, film, foam, etc.), colour (white, blue, black, grey, yellow, red, etc.) and size (1–5 mm, <1 mm, etc.). It is important to analyse polymer types with reliable techniques (spectroscopic approaches: FTIR/Raman, etc.) especially for smaller microplastics (MSFD Technical Subgroup on Marine Litter 2013).

Different categories have been defined for microplastics by different authors: fibres/filaments, fragments, foams, films, granules and pellets (Berov and Klayn 2020); fibre, fragment, nylon, polystyrene, paint particles and others (Öztekin and Bat 2017); fibres, fibre clumps, foils, fragments and spherules (Pojar et al. 2021a, b); fibres, fragments, films, foams and microbeads (Aytan et al. 2020a); etc. There is no agreement on the type of categories used to classify microplastics, but the dominant microplastic types were reported as fibres, fragments and films, in general.

The colour classifications of microplastics have been made by many authors (blue, black, green, orange, red, transparent, white, yellow, pink, etc.) (Öztekin and Bat 2017; Aytan et al. 2020a; Pojar et al. 2021a, b; Cincinelli et al. 2021; Şentürk et al. 2020; Gedik and Eryaşar 2020). Colours have the ability to increase the likelihood of ingestion because they can be mixed with food by a variety of species.

Some commercial fish species and their larvae are visual predators on minute zooplankton, and they can eat microplastics that look like their prey (Rodríguez-Seijo and Pereira 2017).

In general, size definition used by researchers was smaller than <5 mm for microplastic particles. Many researchers investigated microplastics smaller than 5 mm (Aytan et al. 2020a; Pojar et al. 2021a, b), some of them only evaluated microplastics between 1 and 5 mm (Şener et al. 2019; Totoiu et al. 2020) and some of them evaluated between 200 and 300 μm and 5 mm, depending on the mesh size of the seawater sampler (Aytan et al. 2016; Öztekin and Bat 2017; Berov and Klayn 2020).

The identification of the macro- and microplastics with spectroscopic methods provided data on the polymer composition (Dris et al. 2018). There were a limited number of polymer analysis data (Aytan et al. 2020a; Cincinelli et al. 2021; Pojar et al. 2021a, b).

Dominant polymer types in seawater Microplastic polymer types were found as follows: PP (75.8%), PAN (12.1%), PS (9.1%) and PA-6 (3%) (Pojar et al. 2021a, b); PE (44%), PP (22.6%), PAC (14.3%) and PET (8.3%) (Aytan et al. 2020a); and PE/PP (44.5%), PA (32.0%) and acrylates and PE-acrylate copolymers (13.3%) (Cincinelli et al. 2021). The dominant polymer types in sediment samples were as follows: PE (57.1%), PP (28.6%) and PA (14.3%) (Aytan et al. 2020a) and PET (31%), PP (26%), PS (18%) and PAN (13%) (Pojar et al. 2021a, b). In 2015, the total production of plastic worldwide was 36.3% for PE, 21.0% for PP, 7.6% for PS, 11.8% for PVC, 10.2% for PET, 8.2% for polyurethane and 4.9% for other polymers (Malankowska et al. 2021). In general, the dominant polymer types encountered in studies are in parallel with the polymer types which production is intensive.

8.8 Microplastics in Marine Organisms of the Black Sea

The presence of microplastics in the organism was reported by researchers recently in the BS (Table 8.2). It was reported that microplastic was found in the zooplankton *Acartia* (*Acartiura*) *clausi* and *Calanus euxinus* by Aytan et al. (2020b); bivalves *Donax trunculus*, *Chamelea gallina*, *Abra alba*, *Anadara inaequivalvis* and *Pitar rudis* by Şentürk et al. (2020); and *Mytilus galloprovincialis* by Gedik and Eryaşar (2020). The data of these authors includes not only the BS but also the Aegean Sea and the Sea of Marmara, fish *Engraulis encrasicolus* by Aytan et al. (2020b) and recently ingestion reports from the mammals complied by Tonay et al. (2020).

Various digestion processes (H_2O_2, KOH and HNO_3) were applied by authors for extraction of microplastics and microscopic identification used for visual evaluation of microplastics according to type, size and colours. Identified particles were approved with spectroscopic methods by Gedik and Eryaşar (2020). In general, the most common types of microplastics were fibres, fragments and films, and the common colours were blue, black, red and transparent. The dominant polymer types in organisms were PET (32.9%), PP (28.4%), PE (19.4%) and PA (5.41%) (Gedik and Eryaşar 2020).

Table 8.2 Microplastics in aquatic organism in the BS

Location	Species	Abundance	Polymer type	Size groups (mm)	Detection method	Type	Colour	References
Southern BS	*Donax trunculus*	1.69 par/ind		Films: 1.44 ± 0.88 Fibres: 1.42 ± 0.93 Fragments: 0.40 ± 0.21	Microscopic identification	Fibres: 66% Films: 25% Fragments: 9%	Blue: 43% Black: 32% Green: 7% Orange: 7% Red: 4% Transparent: 4% White: 1% Yellow: 1% Pink: 1%	Şentürk et al. (2020)
	Chamelea gallina	2.07 par/ind						
	Abra alba	4 par/ind						
	Anadara inaequivalvis	4 par/ind						
	Pitar rudis	0 par/ind						
Southeastern BS–Turkey	*Acartia* (*Acartiura*) *clausi*	0.002 par/ind		Fragments: 0.121 ± 0.128 Films: 0.038 ± 0.015	Microscopic identification	Fragments: 50% Films: 50%	Red Black	Aytan et al. (2020b)
	Calanus euxinus	0.004 par/ind		Fragments 0.066 ± 0.043	Microscopic identification	Fragments: 100%	Red Black Blue	
Southeastern BS–Turkey	*Engraulis encrasicolus*	0.25 par/fish		0.07–4.94	Microscopic identification	Fibres: 53% Films: 37% Fragments: 10%	Black: 16% Blue: 13% Transparent: 13% Red: 10%	Aytan et al. (2020b)

Location	Species	Abundance	Polymer type	Size groups (mm)	Detection method	Type	Colour	References
Southern BS–Turkish Coast and Marmara and Aegean Sea	*Mytilus galloprovincialis*	0.69 par/mussel 0.23 par/g (fw)	PET: 32.9% PP: 28.4% PE: 19.4% PA: 5.41% PAC: 3.15% CA: 3.15% PVC: 2.25% PS: 1.8% PC: 1.35% EVA: 0.901% PVF: 0.901% PAN: 0.45%	<0.5 mm: 26.58% 0.5–1 mm: 17.12% 1–1.5 mm: 16.22% 1.5–2 mm: 9.01% 4–4.5 mm: 8.11% 2–2.5 mm: 5.86% 4.5–5 mm: 5.86% 3–3.5 mm: 4.50% 2.5–3 mm: 4.05% 3.5–4 mm: 2.70%	Microscopic identification–FTIR	Fragments: 67.6% Fibres: 8.4% Films: 4.05%		Gedik and Eryaşar (2020)

Microplastic consumption was reported from the base of the food web and *Engraulis encrasicolus* (European anchovy) which is the dominant planktivorous fish and main commercial fish stock of the BS (Bat et al. 2014). Resulting from the ingestion of microplastic, the potential danger of plastic-associated contaminants may pose a risk on biota.

8.9 Conclusion

When environmentalists and scientists started discussing the death of the BS at the end of the 1960s, there was little question that if no alternatives to the rising attack on the BS ecosystem could be found, the sea's fate would be sealed for all time. However, experiments and interventions have begun to avert a negative outcome. It is positive that a series of legally binding agreements bring various kinds of contamination under control and improve marine life protection. Remarkably, it had involved all the BS countries Bulgaria, Romania, Georgia, Turkey, Ukraine and Russia to discuss the protection of the sea that they all share. As a result of the participation of Bulgaria and Romania among the European Union countries, the BS has become more prominent. In 2008, the EU's Marine Strategy Framework Directive (MSFD) went into effect. Member states are required by the MSFD to achieve and/or preserve good environmental status in their marine waters, as well as to take steps to meet the set targets. The Marine Strategy Framework Directive reported 11 descriptors, and the tenth definition concerns marine litter and microplastics.

Microplastics are considered as an emerging threat for aquatic ecosystems. So, it is needed to better assess the amount, distribution and sources of this pollutant in the environment; in addition further research are required to the toxicological and ecological risks of these particles on the ecosystem.

The investigations on the presence of microplastics have continued to increase in recent years, and the deficiencies in their distribution in the marine environment are still scarce. There are a limited number of studies especially in the water column and sediment. The current lack of comparable data makes it impossible to estimate future trends in microplastics in the BS. Therefore, another handicap in these studies is the lack of a common methodology. Develop monitoring and assessment approaches, methodology, evaluation criteria and reporting standards for regional and national monitoring and assessment in the BS.

The ingestion of microplastics by aquatic organism all over the world is known, but regional studies on aquatic organisms in the Black Sea are very limited. The ingestion reports are about only few species. Therefore, the investigation of microplastics by organisms needs to be improved.

Pollutant interaction caused by plastics is one of the biggest shortcomings in the BS region. More research is needed about plastic additives associated with microplastics and adsorbed contaminants from the surrounding environment in the BS.

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Chapter 9
Occurrence and Fate of Microplastics in Freshwater Resources

Simin Nasseri and Nahid Azizi

Abstract Microplastics are one of the emerging pollutants in the world. This pollutant is present in all parts of the environment, especially in water. The principal sources of microplastics in water are divided into two categories, primary and secondary. The primary microplastics are primarily produced in micrometer sizes in factories and enter the water through the wastewater treatment effluent. However, secondary microplastics originate from the decomposition of larger plastics deposited in shorelines and gradually enter the water over time. Microplastic entry into the water also occurs through transporting from the atmosphere and soil; for example, microplastics in the atmosphere can deposit on the soil surface or into the water. In addition, microplastics in the soil can be washed into freshwater through runoff and eventually enter the seas and oceans. Ultimately, the microplastics in the water either settle into the sediments or enter the body of aquatic organisms in various ways. Therefore, accumulating microplastics in the body of aquatic organisms originates health problems. Furthermore, microplastics can cause problems for humans who may consume them as seafood. Therefore, it is clear that there is an urgent need to develop removal methods for this contaminant. Wastewater treatment plants cannot entirely remove microplastics, so specific removal techniques are being developed in recent years.

Keywords Occurrence · Fate · Microplastic · Freshwater

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9.1 Introduction

Plastic mass production has increased rapidly since 1940, and it has become an essential material in all parts of the industries because of its properties such as flexibility, hardness, elasticity, temperature resistance, and chemical stability (Herbort et al. 2018; Parenti et al. 2019). Annual plastic production has reached 280 million tons, and global production of resins and fibers has increased from two million tons in 1950 to 380 million tons in 2015, indicating 8.4% of the Compound Annual Growth Rate (CAG). There are evidences that 79% of all these produced plastic materials, especially disposable plastic products, enter the environment as waste. Plastic wastes can accumulate in different environments and eventually turn into environmental pollutants (Geyer et al. 2017; Herbort et al. 2018; Zaman et al. 2019; Parenti et al. 2019). These plastic wastes and garbage are broken down into smaller pieces in different parts of the environment under various processes and finally turn into microplastics, which are identified as insoluble polymer particles with a size of less than 5 mm (Koelmans et al. 2019). Microplastics have been detected in various environmental components such as the atmosphere, soil, water resources and all aquatic environment (including oceans, rivers, lakes, beaches, and swamps), sediments, and the digestive track of vertebrates and invertebrates around the world. Marine environments, freshwater, terrestrial, and atmosphere are interconnected by various source-path-sink connection networks that can affect the movement and persistence of microplastics in environmental matrices (Huang et al. 2020b; Zhang et al. 2020). Microplastics can enter the food chain in various pathways because of their small size and may harm the organisms physically or through chemical leakage (Collard et al. 2019).

9.2 Basic Source of Microplastics in Freshwater

The origin of microplastics can be primary or secondary. Primary microplastics can enter the freshwater through the discharge of domestic wastewater including polyethylene, polypropylene, and polystyrene particles in cosmetic products, such as soaps, hand and face cleansers, toothpaste, cleansing gels, deodorants, and shampoos. These particles are often smaller than 300 μm and may contain additives such as dyes. Moreover, industrial products include resin powder or granules, and the raw materials, which are used to make plastic products, can be another source of microplastics (Eerkes-Medrano et al. 2015; Storck et al. 2015). On the other hand, secondary microplastics are originated through breaking large plastic products. Microplastics may break before or after entering the environment. For example, synthetic fibers from the laundry can be separated and released into the atmosphere as microplastics and finally get into water resources. However, the polymer structure of plastic wastes can be weakened by ultraviolet light exposure, mechanical stresses, or additive leakage and may turn into microplastics after disposing along shorelines. Therefore, microplastics with the secondary source have a longer residence time in the environment (Leslie et al. 2013; Eerkes-Medrano et al. 2015).

9.3 Atmospheric Microplastics as a Source of Microplastic in Water

9.3.1 Atmospheric Microplastic Accordance

Nowadays, the presence of microplastics in the atmosphere is one of the primary concerns and considered as an emerging air pollutant. Furthermore, atmospheric microplastics are one of the significant factors for microplastic entry into other environmental components such as water and soil. The atmospheric microplastics' occurrence, quality, and characteristics in remote areas, urban areas, and industrial areas are the topics of recent publications.

Microplastics can be studied in suspended atmospheric particles, atmospheric precipitation, and urban deposited dust (Huang et al. 2020b; Zhang et al. 2020). The frequency of atmospheric microplastics varies in different regions and increases significantly while decreasing their size. In a study, 2–355 microplastics/m^2.day were collected, and 29% of them were identified as synthetic fibers. It is reported that about 3–10 tons of microplastics could deposit in a 2500 km^2 area annually (Crawford and Quinn 2016). It is worth noting that the concentration of outdoor microplastic fibers is much lower (0.3–1.5 fibers/m^3) compared to indoor samples (1–60 fibers/m^3). These results indicate the microplastic transportation from indoor to outdoor through air conditioning; therefore, the outdoor microplastic concentration levels significantly reduce due to the dilution phenomenon (Dris et al. 2017; Huang et al. 2020b; Zhang et al. 2020).

9.3.2 Atmospheric Microplastic Characteristics and Sources

Atmospheric microplastic has various characteristics in different regions. According to previous studies, microplastics have diverse shapes, including fiber, foam, fragment, and film, in the atmosphere. Compared to microplastics of aqueous environments and sediments, the principal size of atmospheric microplastics is much smaller. Fibrous microplastic size in the atmosphere has been reported in the range of 1–500 μm in thickness and/or width; moreover, fragment and film microplastic size are reported <50 and 50–200 μm, respectively. It is worth noting that the most common microplastic dyes in the atmosphere are blue and red fibers (Huang et al. 2020b; Zhang et al. 2020).

Today, the plastic industry is facing an increase in synthetic fiber production (clothing, upholstery, and carpets), which has caused the increase of fibrous microplastics in the atmosphere. However, fragment microplastics are originated by exposing larger plastics to strain, fatigue, or ultraviolet light. Therefore, it is expected that fragment microplastics identify less than fiber in the atmosphere. On the other hand, polyethylene terephthalate (PET) has the highest percentage of the atmospheric microplastic composition. This polymer is commonly used to produce polyester fiber, fabric, and cording for textiles. It can be concluded that fabric clothes are probably the principal source of microplastics in the air, which may enter the atmosphere by drying them under the natural sunlight. On the other hand, fibers may also enter the atmosphere by using household appliances such as carpets and curtains (Liu et al. 2019; Huang et al. 2020b; Zhang et al. 2020). In addition, dust can be a secondary source of microplastics in the atmosphere, because the deposited microplastics can be resuspended and pollute the atmosphere (Huang et al. 2020b).

9.3.3 Atmospheric Microplastic Entrance to Water

The mechanical erosion and chemical weathering role in microplastic decomposition can be clarified by exploring the surface texture of the fibers (such as attached particles, fragments, and flake surface). Moreover, microplastics turn into finer particles through optical oxidative degradation or wind shear. For example, epoxy and alkyd resins can gradually turn into fragment microplastics in the atmosphere after a long time of exposure to ultraviolet radiation and physical erosion (Huang et al. 2020b; Zhang et al. 2020).

The atmosphere is a significant route of regional and global transport for suspended solids because of the processes such as concentration gradient, wind speed and direction, airflow up/down, convective flow, turbulence, temperature, and humidity. Therefore, microplastics can be transported to remote areas by the atmosphere due to their light weight, durability, and other inherent properties. In addition, the density and shape of the microplastics affect the aerodynamics and consequently its atmospheric transfer. For example, the film is more likely to have atmospheric dislocation than fragment microplastics with the same weight due to its thin and flat surface. Atmospheric transport allows microplastics to reach remote areas (up to 95 km away), even to the regions without a local source of plastics. For instance, lower concentrations of microplastics in snow samples of polar regions than populated areas indicate microplastic transportation through atmospheric precipitation (Allen et al. 2019; Bergmann et al. 2019; Huang et al. 2020b; Zhang et al. 2020). Transferring through dry or wet deposition leads to the atmospheric microplastic displacement to other environments. For example, fiber can enter runoff through atmospheric precipitation (such as snow) and eventually enter the aquatic and soil environment and food chain. Furthermore, indoor microplastic can transport to water environments by wastewater through washing the deposited fibers on the floor (Dris et al. 2017; Bergmann et al. 2019; Huang et al. 2020b).

9.4 Microplastics in Soil as a Source of Microplastic in Water

9.4.1 Accordance of Microplastics in Soil

Factors such as the lack of appropriate technology and the difficulty of particle analysis in complex matrices have limited the study of the presence and distribution of microplastics in the soil. According to researches, microplastics are abundant in the soil as an emerging contaminant. The microplastic number is reported from zero to tens of thousands per kilogram of soil around the world. Even in agricultural lands, where no direct artificial operations have been performed (Tibetan Plateau), between 20 and 110 microplastics per kilogram of soil have been observed (Yang et al. 2021). Regarding soil type, microplastic concentration in the forest (4.1×10^5 items/kg) is significantly higher than vegetable soil (1.6×10^5 items/kg) and vacant lands (1.2×10^5 items/kg) (Zhou et al. 2019).

9.4.2 Characteristics and Sources of Microplastics in Soil

The main shapes of microplastics in the soil are fragment and fiber. The reason for high percentage of microplastic fragments in the soil is the decomposition of plastic film residues (related to mulching, plastic bags, and pesticide and fertilizer containers) around agricultural lands. On the other hand, sewage sludge use in the soil as fertilizer increases fiber microplastics in the soil ecosystem. The range of microplastic size in the soil system is 0.1–2 mm, and smaller particles (less than 1 mm) have a higher percentage in the soil environment, similar to atmospheric microplastics (Yang et al. 2021). The principal sources of microplastic entry into the soil are summarized in three categories: (1) through the use of sewage sludge as fertilizer, (2) agricultural and gardening operations, and (3) the effect of disposition, runoff, and breaking of larger plastics (Hurley and Nizzetto 2018).

9.4.3 Soil's Microplastic Entrance to Water

Microplastics in soil may be stored, displaced, or subjected to processes such as erosion and decomposition. The soil aggregation affects its properties like pore size (which is the transfer route for gas and water) and limits the movement of the living organism. As a result, soil aggregation limits microplastic exposure to soil organisms and prevents their transformation (Rillig and Lehmann 2020). It is also possible that microplastics bury in the soil during a flood event, which causes microplastic accumulation and limits the decomposition forces. Therefore, it increases microplastic persistence potential in the soil system. Furthermore, soil organisms can displace the microplastics by attaching the microplastics to the outside of their body or microplastic ingestion. Ingestion of microplastics by these organisms may consider as its removal, but in fact, it will lead to regular displacement and cause health problems simultaneously. As observed by Horton et al., organisms such as earthworms (e.g., *Lumbricus terrestris*) move the accumulated microplastics on the soil surface through their tunneling activities and distribute them in deeper layers of the soil (Horton et al. 2017; Hurley and Nizzetto 2018).

On the other hand, soil microplastics can decompose by several mechanisms, including the following items:

1. Optical and thermal oxidation decomposition.
2. Some degrees of biodegradation by microorganisms after a long period of exposure to the environment and oxidation.

After optical oxidative and thermal decomposition, biological degradation plays a critical role in the final fate of microplastics in the soil. The significant factor that influences the microplastic degradation includes molecular weight, chemical structure and morphology, hydrophobicity, water absorption, and surface hardness of materials. Obviously, biodegradation is the process of mineralizing organic compounds by microorganisms to produce carbon dioxide, water, and methane under aerobic or anaerobic conditions (Ng et al. 2018). However, according to the microplastic characteristics, they will not be completely decomposed in the soil. Therefore, microplastics will eventually be taken by plants and enter the food chains or transfer from the soil system to the streams and rivers through erosion by water and wind (Hurley and Nizzetto 2018).

9.5 Accordance of Microplastics in Freshwater

After entering microplastics into water, they remain in the water sources for thousands of years due to their chemical stability. According to previous studies, the concentration of microplastics is increasing every year in rivers, lakes, and other water resources (Crawford and Quinn 2016; Collard et al. 2019). The concentration of microplastics has very different levels in various regions, and water resources depend on diverse factors such as population density, proximity to urban centers, and hydrological and metrological conditions. Furthermore, due to the lack of modern wastewater and waste management, even water resources around a small population contain high microplastic pollution. Correspondingly, the frequency of microplastics can vary depending on the sampling location as the microplastic concentration elevates by increasing the water column depth and proximity to the shores. According to previous studies, the concentration of microplastics in the aqueous medium ranges from undetectable to more than 100,000 particles/m^3 (Herbort et al. 2018; Xu et al. 2020; Zhang et al. 2020). The microplastic concentration in water increases with the increasing disposal of plastic wastes; therefore, a large part of the microplastics in water has secondary origins (Li et al. 2020; Xu et al. 2020).

There are a few studies on microplastic detection in drinking water. However, there is evidence of microplastics' presence in the influent and effluent of water treatment plants. For example, in one study, the concentrations of 50–150 μm microplastics in raw and treated water were 0–7 and 0.7 particles/m^3, respectively. Despite the small number of microplastics in drinking water, it should consider as a threat to human health, because of their small size (Mintenig et al. 2019).

9.6 Characteristics of Microplastics in Freshwater

Microplastic characteristics vary depending on the shape of the primary microplastics, the degradation processes, and the residence time in the environment. Some of the microplastics appear spherical (most of them are "primary microplastics"), and the others have fibrous or random shapes (most of them are "secondary microplastics"), although over time, these shapes may also change in the environment and make it difficult to identify their sources. A standard classification for plastic particle size includes microplastic (MP) (1–5 mm), mini-microplastic (MMP) (1 μm to 1 mm), and nanoplastic (NP) (less than 1 μm) and in terms of shape is divided into five categories including pellet (PT), fragment (FR), fiber (FB), film (FI), and foam (FM) (Crawford and Quinn 2017). There are challenges in detecting microplastics in three different aspects, including water sampling, microplastics separation from other materials (organic and inorganic), and identifying the microplastics' composition. Furthermore, there is no standard protocol for detecting this contaminant in the aquatic environment. In addition to the standard method requirement, it is necessary to state sampling information, including the type of equipment, its period, date, and place to ensure the validity and comparability of all microplastic researches (Lv et al. 2019; Zhang et al. 2020). The microplastic characteristics can lead to source identification. According to previous studies, polyethylene and polypropylene have the highest percentage of polymer type, and fiber and fragment have the highest microplastic shape in freshwater (Li et al. 2020; Xu et al. 2020).

9.7 Health Problems of Microplastics in Water

Detected microplastics in oceans and aquatic environments make up only 1% of the total microplastics that enter this environment. The remaining amount can deposit in sediments, enter the shores, or be ingested by aquatic organisms (Rhodes 2018). Consequently, excessive accumulation of these microplastics in aquatic organisms may cause physical damage, physiological defects, slow growth, and endocrine disorders. Recent studies have also shown changes in their immune system, metabolism, neurotransmission, and reproduction. It is worth noting that if nanoplastics enter the fish body, it can cross the blood-brain border and lead to brain damage and behavioral changes (Horton et al. 2018; Naidoo et al. 2020; Wu et al. 2020). The other problem of microplastic ingestion can be posed by contaminants that are adsorbed on their surface, including additives such as dyes, stabilizers, lubricants, and flame retardants, which can release into living organisms' bodies and cause toxicity (Slootmaekers et al. 2019; Huang et al. 2020a, b). Conclusively, microplastics can enter the human body through drinking water and contaminated food consumption, especially seafood.

As a result, seafood consumption can increase the hazardous chemical load in human bodies due to the environmental contaminants adsorbed onto microplastics, including toxic organic chemicals; various heavy metals, such as zinc, copper, lead, silver, and arsenic; and some nanoscale adsorbents like titanium dioxide (TiO_2) (Schmid et al. 2018; Ma et al. 2019a, b). The presence of microplastics in human feces has also been reported, but according to a 2019 report by the World Health Organization (WHO), microplastics with a size larger than 150 μm cannot pass through the wall of the gastrointestinal tract, although smaller particles are unsafe and potentially dangerous (Zaman et al. 2019).

9.8 Fate of Microplastics in Freshwater

As mentioned in microplastics in soil section, microplastics are washed off the ground through runoff; so, the sea and ocean can be considered as the ultimate destinations for microplastics in the atmosphere, soil, and freshwater. Danube River, for instance, imports annually an average of 1553 tons of microplastics into the Black Sea (Geyer et al. 2017; Horton et al. 2017; Xu et al. 2020). Sediments are the final destination of microplastics in the aqueous environment. As a result, microplastics have abundantly been found in these sediments with a concentration of about 30,000 particles/kg of dry weight. Indeed, the occurrence of small size (20 to 50 μm) and low-density microplastics without the settling properties are reported in the sediments. According to a study, microplastics with a less than 200 μm diameter and specific gravity (particle density/water density) less than one can transfer directly to the oceans with the most limited interaction with river bed sediments (Woodall et al. 2014; Nizzetto et al. 2016; Drummond et al. 2020). It is assumed that the microplastics introduced into the sediments remain immobile under stable conditions, but when flooding events occur, there is the possibility of microplastic resuspension and returning to the rivers and water wells (Drummond et al. 2020).

On the other hand, a large number of wastewater treatment plants are located near the water resources. Although microplastics may be removed from up to 99% of wastewater by conventional wastewater treatment plants through primary and secondary treatment processes, there is still the possibility to release a high level of microplastics into the aquatic environment due to the large volume of wastewater treatment plants' effluent discharge (Geyer et al. 2017; Zaman et al. 2019).

9.9 Technologies for Microplastic Removal from Water

Since the aqueous environment is the last destination of microplastics, its removal is very necessary to prevent human exposure to this contaminant. Nowadays, many methods have been used to remove this pollutant from the environment.

9.9.1 Microplastic Removal in Wastewater Treatment Plants

In wastewater treatment plants, the pretreatment step has the most significant effect on the size distribution of microplastics and effectively removes (7–45%) microplastics with larger size (100–300 μm and >300 μm). Therefore, the percentage of smaller microplastics (20–100 μm) will increase in the effluent (Talvitie et al. 2017). Similarly, in the secondary treatment step, there is a possibility of removing large microplastics (300–500 and >500 μm), which leads to the relatively low frequency of bigger size at the effluent of this step (Ziajahromi et al. 2017; Sun et al. 2019). Regarding the shape of microplastics, studies have shown that in the pretreatment step, granular microplastics have higher removal percentage than other shapes. Moreover, fibers can be more efficiently separated from wastewater than fragments. On the contrary, the secondary treatment step is more efficient for fragment removal, which increases the relative abundance of fiber microplastics in secondary effluent. Finally, tertiary treatment can eliminate most of the remained microplastics. However, the microplastics at the inlet and outlet of the tertiary treatment can have a low concentration (less than 1 particle/L in most cases) (Sun et al. 2019). Previous studies have examined advanced technologies, including disk filter (DF), rapid sand filter (RSF), dissolved air flotation, and membrane bioreactor (MBR), and have been reported more than 95% removal for microplastics (>20 mm) from secondary effluent (Eerkes-Medrano et al. 2015; Talvitie et al. 2017).

9.9.2 Membrane Filters for Microplastic Removal

Since membrane filters are an acceptable method for removing low-density, nonbiodegradable particles, they can also be a reliable option for microplastic removal. In one study, the influent turbidity originated by microplastics (195 NTU) was reduced to less than 1 NTU in the effluent by a membrane filter in 20 min, and elsewhere the number of microplastics was decreased from 1 to 0.4 particles/L using MBR as the tertiary treatment. These results indicate that membrane filters are one of the most remarkable technologies for removing microplastics. It is worth noting that the bioreactor membrane filter is more efficient than dynamic membrane filters in microplastic removal (Lares et al. 2018; Li et al. 2018; Padervand et al. 2020).

9.9.3 Adsorption and Ingestion of Microplastics by Aquatic Organisms

As mentioned in various sections of this chapter, microplastics can adsorb environmental pollutants, and this shows that these resistant materials can be adsorbed by other environmental components as well, so the researchers investigated the adsorption of nanoplastics on algae. According to the results, neutral and positively charged plastic particles can be adsorbed on the cell wall of *P. subcapitata*. However, the properties of the material and the media conditions have a significant effect on adsorption efficiency (Nolte et al. 2017). Polyethylene microplastic (53–500 μm) consumption by the Red Sea giant clam is also investigated, which resulted in the removal of 7.55–1.89 beads per day (66.03% microplastic removal from the water column), although there is the possibility of health risks to the organisms (Arossa et al. 2019).

9.9.4 Microplastic Removal by Coagulation

Nowadays, coagulation is the principal technology for removing pollutants in water treatment plants, and due to the high quality of treated water, its application is expected to continue for the next few decades. Furthermore, this process is a conventional pretreatment method in water treatment plants to remove natural colloidal or suspended particles by producing settleable particles (through increasing particle size and density). The coagulation process for microplastic removal in wastewater treatment plants has been investigated, and the results show that the smaller polyethylene particles can be trapped in flocs more efficiently and may lead to higher performance. Therefore, coagulation can be considered as a suitable process for microplastic control and removal. The average microplastic removal of three wastewater treatment plants in the primary, secondary, and tertiary treatment steps (coagulation) were 75, 92, and >98%, respectively (Ramirez et al. 2016; Hidayaturrahman and Lee 2019; Ma et al. 2019a, b).

9.9.5 Microplastic Degradation

Biodegradation of petroleum plastics, especially polyethylene, polypropylene, and polystyrene, has been started since the 1970s. Polyethylene, polypropylene, polystyrene, and polyurethane are generally considered nonbiodegradable without heat or ultraviolet (UV) pretreatment. Therefore, they can persist hundreds of years in the environment. However, recent research shows the potential for biodegradation of polyethylene, polystyrene, and polyethylene terephthalate by some organisms (Wu et al. 2017). These organisms include the famed *Ideonella sakaiensis*, the bacterium of *Lumbricus terrestris* earthworm's stomach, *Plodia interpunctella* waxworms, marine mushrooms, *Zalerion maritimum*, or caterpillars of the wax moth *Galleria mellonella* (da Costa et al. 2019).

9.9.6 Controlling Microplastic Entrance to the Environment

Since there is no 100% effective method to remove microplastics from different parts of environment, the best solution to reduce this pollutant is controlling their input sources. Nowadays, 6% of petroleum in the world is used to manufacture plastic products, which will reach 20% by 2050. As a result, the current approach to the manufacture and use of plastics (especially their end-use) requires immediate modification (Ma 2018). Increasing the collection and recycling of plastic wastes to reuse or manufacture new products can cause the reduction of raw plastic consumption. This action is a critical aspect of reducing the plastic waste amount. It is clear that in the use of plastics and fossil fuels, the process of "extraction, manufacturing, disposal (waste generation)" must be replaced with a "reduction, reuse, recycling, reproduction" system to guarantee the future of public health of humans and the earth and decreasing the microplastics in the environment (Rhodes 2018).

9.10 Conclusions

Microplastics are one of the emerging pollutants in the world. This pollutant is present in all parts of the environment, especially in water. The principal sources of microplastics in water are dividing into two categories, primary and secondary. The primary microplastics are primarily produced in micrometer sizes in factories and enter the water through the wastewater treatment effluent. However, secondary microplastics originate from the decomposition of larger plastics deposited in shorelines and gradually enter the water over time. Microplastic entry into the water also occurs through transporting from the atmosphere and soil; for example, microplastics in the atmosphere can deposit on the soil surface or into water. In addition, microplastics in the soil can be washed into freshwater through runoff and eventually enter the seas and oceans. Ultimately, the microplastics in the water either settle into the sediments or enter the body of aquatic organisms in various ways. Therefore, accumulating microplastics in the body of aquatic organisms originates health problems. Furthermore, microplastics can cause problems for humans who may consume them as seafood. Therefore, it is clear that there is an urgent need to develop removal methods for this contaminant. Wastewater treatment plants cannot entirely remove microplastics, so specific removal techniques are being developed in recent years.

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Chapter 10
Occurrence of Microplastics in Freshwater

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Abstract Healthy freshwaters contribute to the conservation of a wide range of species and provide several ecosystem services indispensables for our society. However, freshwater contamination is an issue requiring awareness and management actions. Microplastics are one of the latest persistent pollutants in freshwaters, widespread worldwide, and to date contaminating rivers and lakes of all continents. In this chapter, we provide an overview on the occurrence of microplastics in freshwaters, mainly discussing (1) methods detecting them in rivers and lakes, (2) contamination quantification and localisation, (3) observations on impacts due to microplastics on biota and (4) plastic pollution origin.

Keywords Microplastic occurrence · Lakes and rivers · Freshwater biota · Temporal distribution · Spatial distribution

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10.1 Introduction

10.1.1 Freshwaters

The 97% of Earth's hydrosphere is contained in seas and oceans; regarding the remaining 3%, only less than 1% is available in continental superficial freshwater (Lupia Palmieri and Parotto 2008). In particular, superficial freshwaters are characterized by lentic ecosystems, such as lakes and wetlands, or lotic ecosystems, such as rivers and streams (Fig. 10.1). These ecosystems are different due to their abiotic characteristics and consequently biotic communities. Freshwaters provide several ecosystem services of high value. Specifically, freshwater may provide water supply for domestic use (e.g. drinking, cooking, washing), industry (e.g. thermoelectric power and manufacturing), agriculture (e.g. crops) and aquaculture (e.g. fish); supply of other food resources such as fish and mussels; biodiversity conservation; flood control; dilution of pollutants; recreation services (e.g. swimming and boating); and aesthetic values (Postel and Carpenter 1997). Over the years several regulations have been developed for the protection of these ecosystems, such as the Water Framework Directive (2000/60/EC) for all superficial freshwaters and the

Fig. 10.1 Types of freshwater ecosystems: (**a**) Lake Bracciano (central Italy), a volcanic lake; (**b**) Torre Flavia wetland (40 km near Rome, central Italy), a coastal wetland; (**c**) the potamal tract of River Marta (northern Latium, central Italy); and (**d**) River Ninfa-Sisto (southern Latium, central Italy)

Convention on Wetlands of International Importance (Ramsar 1971) for the wetlands and aquatic birds. In fact, several factors threaten freshwater ecosystems, such as water pollution, eutrophication, excessive collection of water, construction of dams or barrages, artificialization of water basins, alien species and climate change (Ericksen 2007).

10.1.2 The Beginning of Microplastic Research in Freshwater

Among the contaminants, microplastics (MPs, plastic <5 mm) represent an emerging issue in freshwater ecosystems. Research on MP in freshwaters is a recent topic of investigation if compared to the first report of plastic contamination in oceans, which date back to the 1970s (Carpenter and Smith 1972). Indeed, the studies in freshwaters started increasing since the first years of 2000 (Blettler et al. 2018). This could be explained by the fact that rivers have begun to be evaluated as plastic sources into seas and oceans. In fact, they can collect and transport plastic waste during the flowing and discharge them at their mouth (Schmidt et al. 2017). In the last few years, it has been understood the importance of monitoring MP for the impacts that can cause to freshwaters themselves. Research on MP in freshwaters has now its own autonomous field of investigation.

In this chapter, we focussed on various components of rivers and lakes, such as water, sediment and biota, and assessed the methods of MP detection, the spatio-temporal distribution of MP, the origin of plastic pollution and the impacts of MP on biota.

10.2 Microplastic Sampling and Analysis

Plastics have been studied for decades; however there are not standardized methods for sample collection, sample treatment, quantification and identification. Thus, it resulted in difficult comparisons among the studies (Cera et al. 2020). Further studies are mandatory to provide guidelines to achieve a standardization and harmonization of results. For each investigated matrix and biota, some available standards are provided by scientific literature, as briefly overviewed below in the following chapters.

10.2.1 Water and Sediment Sampling

In general, the sampling methods for MP detection in water and sediments are different. Water samples are usually volume-reduced, but they can also be sampled entirely, while bulk sediments are always sampled without being volume-reduced (Hidalgo-Ruz et al. 2012; Eerkes-Medrano et al. 2015).

For water sampling, either the surface or the column of water can be collected. However, the analyses limited only to the surface layer may not adequately reflect the actual MP abundance in the whole water body (Szymanska and Obolewski 2020). To sample MP in water, researchers can use (1) nets, (2) surface microlayer method, (3) hand nets and (4) bulk water sampling (Li et al. 2018) (Table 10.1). Regarding the first method, manta nets, neuston nets or plankton nets have the benefit of covering large sampling areas and reducing the sample volume. The most common net used is the manta net having 330μm mesh (Li et al. 2018). Regarding the second method, the surface microlayer method is performed manually with a sieve to allow the collection of upper-layer water. Regarding the third method, the surface water is filtered by a hand net, and a volume-reduced sample is collected. For the last method, water is sampled in containers by hand or by pumps, and its volume is not reduced; the sampled water volume might vary among researches, passing from 100 L samples to 100 mL (Li et al. 2018). Nets are the most used tool, followed by pumps and sieves (Prata et al. 2019). In fact, nets have various advantages. For example, neuston and manta nets can sample large volumes of water and are easy to use, resulting largely used by scientists. However, they are an expensive and time-consuming equipment, also requiring a boat. In addition, they present the power limit of detection of 333μm (Prata et al. 2019). Nets with smaller mesh, even 80μm, can be used to filter water, but the diameter is usually smaller (Gallitelli et al. 2020). At the same time, pumps sample large volumes of water and do not collect a subsample based on mesh size; however, they require specific equipment and high energy to work. Instead, samples may be easily filtered during the fieldwork by sieves and a known volume of water (Prata et al. 2019). Tamminga et al. (2019) carried out a comparison between manta trawling and pump sampling finding that these are complementary techniques as ‘the

Table 10.1 Tools and methods used to sample and analyse MP in water and sediment

Sample	Type	References
Water	Neuston and manta nets	Prata et al. (2019), Tamminga et al. (2019), Campanale et al. (2020)
	Plankton net	Dris et al. (2015), Gallitelli et al. (2020)
	Sieves	Masura et al. (2015) (NOAA)
	Pumps	Desforges et al. (2014), Jönsson (2018), Tamminga et al. (2019)
Sediment	Manually by non-plastic tools	Faure et al. (2015), Blair et al. (2019), Gallitelli et al. (2020)
	Grabber	Castañeda et al. (2014), Pojar et al. (2021)
	Box corer	Vianello et al. (2013), Stock et al. (2019)

pump sampling covers small microplastics, which are greater in number and the volume-reduced sampling (manta) covers large microplastics, being less abundant but still important when it comes to weight estimates'.

For sediment sampling, as reported by Stock et al. (2019), the sampling in freshwater habitats is usually conducted with a grabber (e.g. Castañeda et al. 2014) or a box corer for superficial sediments (Vianello et al. 2013) (Table 10.1). In addition, sediments can also be collected manually by spade (Faure et al. 2015; Blair et al. 2019). To allow comparisons among studies, the surface sampled, the bulk weight, or the volume of sediments should be provided. For instance, Gallitelli et al. (2020) collected 12 samples of 250 mL of sediment from each site. Sediment was taken manually by metallic tools from the 5 cm upper layer of the substrate. After sampling, collected substrates were stored in 250 mL sterilized aluminium containers.

10.2.2 Water and Sediment Treatment

After sampling, most water and sediment samples are pre-treated with a chemical or enzymatic digestion for destroying the organic matter. The purification process can be divided in chemical degradation and enzymatic degradation. In the first type of degradation, MP samples are chemically treated mainly using hydrogen peroxide (H_2O_2) (Nuelle et al. 2014; Lusher et al. 2017). When using these strong oxidizing acids such as sulfuric acid and nitric acid, it should be considered that chemical degradation might damage MP polymers. Moreover, the enzymatic degradation is another technique for removing the organic matters. Indeed, instead of chemical degradation, MP samples will be put together with a mixture of enzymes such as lipase, amylase, proteinase, chitinase and cellulose (Cole et al. 2014; Li et al. 2018). In this manner, samples were purified through enzymatic process and then undergo spectroscopic analyses (Löder et al. 2017). However, enzymatic purification is a more cost-effective process than chemical purification. Indeed, these two methods require to be optimized in order to produce a well-purified sample.

Thereafter, MP analysis is based on the following steps: (1) extraction of MP from the natural matrix, (2) quantification and (3) chemical characterization of MP polymers (Table 10.2). The methods of chemical characterization are described in Sect. 10.2.4.

Regarding step (1), water samples are usually filtered to extract MP (Löder and Gerdts 2015). Density separation is one of the most used techniques for sediments. The density separation uses fluidization and floatation. To extract MP from organic particles, salts (e.g. NaCl and NaI) are added to water samples to increase water density. Then, the sample is mixed with the saturated salt solution, allowing the floating and separation of MP based on density (Hidalgo-Ruz et al. 2012). The best solution is the combination of fluidization in a NaCl solution and floatation in NaI solution (Nuelle et al. 2014). The most common solution is NaCl with a density of 1.2 kg/L due to its low cost and no toxicity to humans. For sediment, samples can be washed with nitric acid and then MP extracted (Claessens et al. 2013).

Table 10.2 Sample treatment for MP in water and sediment

Sample	Treatment			
	Purification	Separation	Quantification	Characterization
Water	Chemical: hydrogen peroxide, sulfuric acid and nitric acid	Saturated salt solutions	Stereoscope, microscope, visual counting	FTIR, μFTIR, Raman, μRaman, SEM-EDS, Pyr-GC-MS
Sediment	Enzymatic: lipase, amylase, proteinase, chitinase and cellulose	Oven at 60 °C for 48 h		
		Saturated salt solutions		

FTIR Fourier transform infrared spectroscopy, *SEM-EDS* Energy-dispersive X-ray spectrometry, *Pyr-GC-MS* Pyrolysis-gas chromatography-mass spectrometry

About step (2), quantification is usually conducted visually by stereomicroscope (or dissecting microscope). Recent attempts of standardisation include the definition of a protocol which describes the category which can be used to define the shape and colour of MP (Lusher et al. 2017; Lusher et al. 2020).

10.2.3 Biota Sampling and Treatment

In addition to water and sediments, MPs have been searched and extracted by several organisms (Lusher et al. 2017). To allow comparisons among results, several methods are used for sampling and sample preparation (e.g., organic digestion and density separation) (Stock et al. 2019). Regarding the sampling methods, biota can be collected in freshwaters by several tools. For instance, benthic invertebrates can be sampled by grasps, traps, creels or bottom crawling, while planktonic invertebrates by manta or bongo nets. Moreover, trawls and electrofishing can be used to collect fish, while bivalves and crustaceans can be collected by hand (Stock et al. 2019). In addition to being collected in field, many organisms may be bought at markets (such as fish, molluscs and crustaceans) (Stock et al. 2019). After collecting organisms, biota must be frozen, desiccated or preserved in formalin or ethanol. This procedure will avoid the MP loss due to defecation that can occur after biota collection (Lusher et al. 2017; Stock et al. 2019).

Regarding the protocol to biota treatment after sampling, smaller organisms were analysed entirely (e.g. macroinvertebrates, bivalves, zooplankton, shrimps, etc.), while many studies analysed the content of the digestive tract of larger animals (e.g. fish, birds, amphibians) (Lusher et al. 2017; Stock et al. 2019). For isolating MP from biota, several methods have been developed including dissection, depuration, homogenisation and digestion with chemicals or enzymes as for water and sediment (Lusher et al. 2017; O'Connor et al. 2020). One of the most used methods is the chemical digestion using H_2O_2 (30%) such as for bivalves (Li et al. 2015; Su et al. 2018), caddisfly cases (Gallitelli et al. 2021) and other biogenic matter (Nuelle et al. 2014). Digestion can happen rapidly with high temperature (24 h at 60° + 48 h at 20°) (Gallitelli et al. 2021; Li et al. 2015) or slowly at room temperature (7 days at 20°) (Nuelle et al. 2014). Su et al. (2018) used 30% H_2O_2 for ≤ 72 h (65 °C) for mollusc tissue degradation. For macroinvertebrates, Windsor et al. (2019a, b) degraded the whole organisms using the density separation with NaCl (1.2 g/cm^3) and 15% H_2O_2 for 48 h (25 °C).

As case study, here it is reported the interactions of MP on freshwater gastropods *Lanistes varicus* (Müller 1774), *Melanoides tuberculata* (Müller 1774) and *Theodoxus fluviatilis* (Linnaeus 1758). For digesting gastropods, Akindele et al. (2019) used a mixture 1:1 of KOH (10 M) and H_2O_2 (34.5–36.5% v/v) to the tissue. Then, the solution was put on a laboratory shaker for 96 h. Then, 3.89 mL formic acid/10 mL KOH was added to each sample to neutralise it. At the end, each sample was filtered on a filter paper and oven-dried at 50 °C for 48 h. After this, sample was ready to be analysed for MP presence and abundance. After treatment, samples can be analysed to search MP. Among all methods, visual identification is one of the most utilized (Lusher et al. 2017). Using micro- and stereoscope, the number and the shape of MP are assessed. After identifying the presence of MP, there will be a chemical characterization to assess whether putative MPs are correctly identified as plastic polymers.

10.2.4 Chemical Characterization

In scientific literature, 81% of studies performed an identification of suspected MP polymers using different techniques. Fourier transform infrared (FTIR) spectroscopic analysis is the most common method used for water and sediment, while FTIR, μFTIR and Raman are equally used for detecting MP in biota (Cera et al. 2020).

These techniques used to characterise MP polymers were used in combination with other equipment. Indeed, FTIR and Raman might be utilized with optical microscopy (micro-spectrometer) (Song et al. 2015). To detect smaller particles, a μFTIR imaging is used, such as for MP in drinking water (Mintenig et al. 2019). In addition, attenuated total reflectance crystal attached to the microscope (ATR-μFTIR) is preferred when MPs reduce the light transmittance (Li et al. 2020a, b, c).

If a greater spatial resolution is needed, SEM-EDS could be used instead of μFTIR and μRaman (Zhao et al. 2017). Indeed, SEM technique makes possible resolutions >1 nm (Busquets 2017), although a quantitative analysis results to be difficult as SEM analyses the sample in a very localised manner. Therefore, if MP sample is not homogeneous, it could be a problem to perform an accurate quantitative analysis.

Another method to identify MP composition is the pyrolysis approach (Pyr-GC-MS; see Dierkes et al. 2019). Differently from FTIR and Raman, this technique is based on the pyrolysis of the polymer to detect the polymer composition, such as 0.1–0.5 mg polymer, i.e. at 700 °C for 60 s in Nuelle et al. (2014). Then, after this thermal degradation, products can be identified by their mass spectrum. A comparison with spectral libraries allows scientists to identify the polymers.

10.3 Spatio-temporal Distribution of Microplastics in Freshwater

Microplastics were detected in the freshwater of all continents, including Antarctica (Gonzáles-Pleiter et al. 2020a). However, the location of the sampling sites is not uniformly distributed around the globe. Developed countries have collected more data on freshwater contamination than developing countries (Yao et al. 2020). In particular, the USA, People's Republic of China and Germany are the main investigated countries (Szymanska and Obolewski 2020; Yao et al. 2020).

As described in introduction, freshwaters include different habitats. Research on MP in freshwaters focusses on rivers, lakes and estuaries, while groundwaters, reservoirs and wastewaters are considered insufficiently studied (Yao et al. 2020). Among the most studied habitats, only a few studies are on estuaries (Alves and Figueiredo 2019; Firdaus et al. 2020; Leslie et al. 2017; Maes et al. 2017; Peng et al. 2017; Vianello et al. 2013; Willis et al. 2017; Wu et al. 2019), and lakes are less studied than rivers, thus highlighting research gaps (Cera et al. 2020).

In all the habitats investigated, sampling based on temporal trends is less frequent than spatial samplings (Sarijan et al. 2021). Spatial samplings generally assess that sediments are more polluted than waters, possibly because the contaminants in waters deposit on sediment after some time; thus sediments can be considered sinks for MP (Cera et al. 2020). The level of contamination of the sampling sites is highly variable in each continent according to the sampling locations (Cera et al. 2020). This could be due to the presence of local factors influencing the contamination. For detailed information of the sources of pollution, please see below. Based on spatial samplings, Asia is the most contaminated continent (Wu et al. 2018). However, Europe and North America also have a high level of contamination (Li et al. 2020a, b, c). There is a lack of studies in many regions of the world, especially Africa, Oceania and South America. Regarding the few studies on temporal samplings, research is still failing to solve the questions regarding the seasonal variability and factors influencing MP loads in water and sediment (see Sects. 10.3.2 and 10.3.4).

The most common MP polymers found in freshwater environments are polypropylene (PP), polyethylene (PE), polystyrene (PS), low-density polyethylene (lPE) and high-density polyethylene (hPE) (Wong et al. 2020). PE, PP, PS and polyethylene terephthalate account for 70% of the total MP in freshwater (Li et al. 2020a, b, c).

10.3.1 Spatial Occurrence of Microplastics in Water

Water sampling is conducted in all continents mostly in rivers and lakes rather than wetlands, urban canals, stormwater and estuaries (Koutnik et al. 2021). The sites where water sampling was conducted to assess MP pollution are highlighted in Fig. 10.2. Europe, Southeast Asia and the region of the Great Lakes are the most

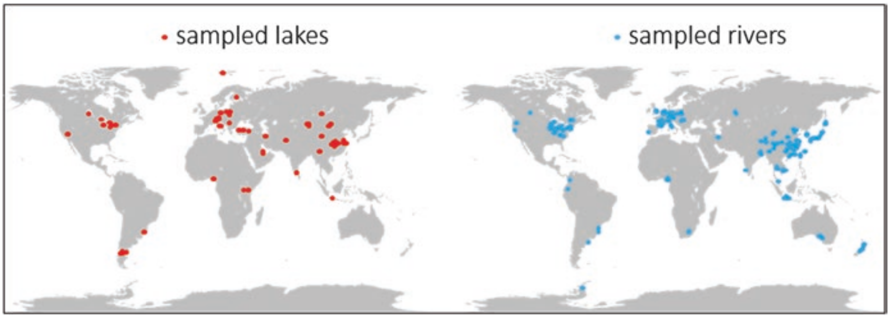

Fig. 10.2 Location of water sampling sites where microplastics are detected in lakes and rivers

sampled areas in the world. Recently, Africa, South America and Oceania have recently increased the number of sampled sites, thus highlighting a positive trend on research activities on MP in freshwater in these regions. Moreover, information was recently available from remote regions such as both Arctic and Antarctic Circles. In these works, microplastics were surprisingly found where human impacts are very low (Gonzáles-Pleiter et al. 2020a, b). Based on the available data of occurrence, the contamination of waters by MP is a widespread issue. However, a relevant knowledge gap is evident, as contamination is unknown in many freshwater bodies, especially in the tropical region (Fig. 10.2).

Fibres are the most frequent shape of MP found in the water column; however, fragments are also abundant (Liu et al. 2020). PP is the polymer more abundant in wetlands and stormwaters, while PE and PET have increasing abundance in estuaries, rivers and lakes; urban canals have a prevalence of PS (Koutnik et al. 2021).

Southeast Asia and Europe are the most polluted geographical areas regarding water (Cera et al. 2020). Water contamination is generally higher in lakes than rivers, having a mean contamination of 11,128 items/m^3 and 2561 items/m^3, respectively (Cera et al. 2020). The level of pollution of lakes ranges from very low values, such as 0.0005 items/m^3 in Lake Dianshan (Asia, China), to high values, such as 400,500 items/m^3 in Lake Winnipeg (North America, Canada and the USA) (Anderson et al. 2017; Su et al. 2018). Wetlands, urban canals, stormwater and glacier and snow have generally a higher contamination than lakes (Koutnik et al. 2021). Instead, the pollution of rivers ranges from zero items/m^3 (Asia, Japan) to 510,140 items/m^3 (Asia, Vietnam) (Kataoka et al. 2019; Lahens et al. 2018). This great variability of concentration ranges shows how the level of contamination is strongly dependent of local factors, such as the presence of wastewater treatment plants. Moreover, it underlines the importance of a standardised monitoring protocol, which can scientifically support the comparison of data.

10.3.2 Temporal Sampling of Microplastics in Waters

Few studies are available on temporal distribution of MP in waters. A study conducted during 3 years of sampling on the same lake (Lake Winnipeg, Canada) showed that the density of MP is similar between the years (Anderson et al. 2017). Regarding seasonal samplings, evidence suggests that the wet season has a lower concentration of MP in waters than the dry season due to dilution (Fan et al. 2019). However, rains can also increase the concentration of MP in waters (Moore et al. 2011). This is confirmed also by samplings conducted in River Nakdong (South Korea), where the wet season contributes to 71% of the total number of MPs discharged throughout the year (Eo et al. 2019). In these studies, plastic waste collection by run-off could be the cause of the increased number of MPs. It is not clear yet how seasonal changes influence the concentration of MP as other factors, such as improper waste disposal, could alter the observations (Rodriguez et al. 2018). Despite the uncertainties due to the lack of data, it seems that season influences the MP load in waters.

10.3.3 Spatial Occurrence of Microplastics in Sediment

In this chapter, the differences in sampling efforts are described for each type of freshwater, i.e. lentic or lotic. Sediment sampling is conducted in all continents and mostly in rivers than lakes (Fig. 10.3). Considering also beaches and shoreline, 71 studies were carried out for analysing MP in the sediments of lentic ecosystems, while 108 for the lotic ones. Without beaches and shoreline, the number of studies on MP in sediment decreased to 44 studies for lentic ecosystems and 104 for lotic ecosystems (Cera et al. 2020).

MPs in sediment were found mostly in lakes rather than rivers. Although the difference in riverine and lacustrine sediments is not evident, median MP concentration is lower in rivers (121 items/kg) than in lakes (150 items/kg) (Cera et al. 2020). This could be explained by the fact that lakes are closed waterbodies that can act as a sink

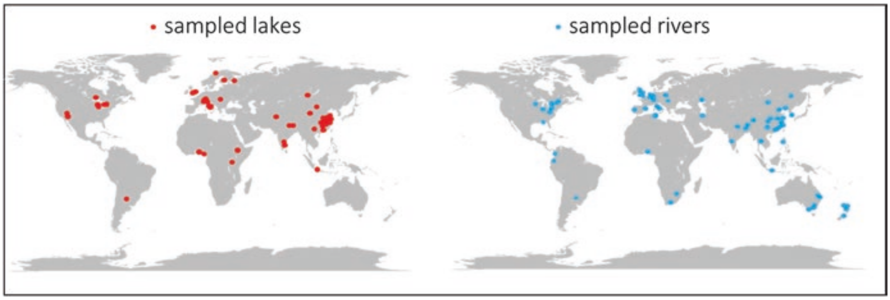

Fig. 10.3 Location of sediment sampling sites for detecting microplastics in lakes and rivers

of MP, while rivers are running waters flowing to the sea; thus, they accumulate less MP.

Among all the continents, the most polluted areas are Southeast Asia, Europe, Africa and North America (Lebreton et al. 2017; Blettler et al. 2018) (Fig. 10.3). In this regard, it is difficult to understand whether some regions (e.g. Australia or South America) are more or less polluted than others (e.g. Asian and European regions) due to lack of data and also to unstandardized methods.

For lakes, the highest MP concentration is 13,925 items/kg and was found in beach and nearshore of Lake Ontario in America (Ballent et al. 2016), while the lowest is 0.24 mg/g, found in the beaches of Lake Bracciano in central Italy (Corti et al. 2020). The highest concentration of MP in river sediments in the world is found in River Wen-Rui Tang with 32,947 items/kg (Wang et al. 2018) and in River St. Laurent with 13,759 items/m^2 (Castañeda et al. 2014). On the other hand, the lowest concentration is found by Alam et al. (2019) reporting 0.0000303 items/kg in sediment of River Ciwalengke and 5 items/m^2 in River Gaolan (Zhang et al. 2019).

Recently, other studies on MP in sediments were published. For instance, Felismino et al. (2021) found 8.3–1070 particles/kg in sediment samples of Lake Simcoe in Ontario, Canada. Microfibres were the dominant shape (89.2%). As the colour regards, MP colour was dominated by blue (46.7%), followed by black (20.8%) and red (11.7%). Surprisingly, Pastorino et al. (2021) did not found any MP in the sediment of a high-mountain lake from Carnic Alps, while only snow samples contained PET MP at very low levels (0.11 ± 0.19/L). On the other hand, there are remote areas that are contaminated, such as a remote and uninhabited lake in Switzerland (Lake Sassolo). Here, Negrete Velasco et al. (2020) aimed at investigating the presence and abundance of MP and fibres, finding 33 MP and 514 fibres per kilogram in the lake sediment.

Despite the presence of MP detected on all continents (also in Antarctica; see Kelly et al. 2020), North America, Asia and Europe are the most studied areas, while Oceania and Africa the less studied. Indeed, there is a large knowledge gap on the global MP pollution. In particular, Cera et al. (2020) highlighted that many countries, belonging to the top 5 countries for plastic inputs in marine waters, such as the Philippines and Sri Lanka, are a clear example of this gap. Recently, the number of articles increased. For example, Singh et al. (2021) reduced the gap exploring MP in sediment of River Ganga in India. MP number and mass ranged between 36 items/kg and 10 to 45 mg/kg, respectively. In addition, in all samples were found white colour and film type of MPs. Among all MPs, the 2.5–5 mm sizes were predominant in number and mass rather than others (Singh et al. 2021). An abundance of MP items was also occurring in Nam Lake (India) with a mean concentration of 309 items/kg (Bharath et al. 2021). In Thailand, sediments of River Chao Phraya were contaminated by 91 ± 13 items/kg and 4.9 ± 3.4 mg/kg of MP (Ta and Babel 2020). The main morphologies of found MP were fragments and fibres, while the colour was mostly white. As known, Southeast Asiatic countries are among the most famous manufacturers and users of plastic products in all the world. For instance, in Taiwan, River Tamsui is expected to be one of the top 20 polluting rivers that introduced 1.47×10^4 tonnes/year of plastic into the ocean – according to the model of

Lebreton et al. (2017). However, MP monitoring in rivers is scarce in Taiwan. In this regard, a study was conducted for MP monitoring in River Fengshan, and 508–3987 items/kg was found in the sediment samples (Tien et al. 2020).

Only few new studies were conducted in Africa recently, not reducing the knowledge gap on MP occurrence in sediments in that continent. Preston-Whyte et al. (2021) monitored MP in harbour environments in the Port of Durban, South Africa. In Africa, a new study highlighted the MP pollution in lake sediments. The abundance of MP ranged between 310 and 2319 items/kg in Lagos Lagoon (Olarinmoye et al. 2020). The majority of MP were fibres, and the most common colours were transparent (41%) and black (30%). Another research analysed MP in sediments and waters of Ox-Bow Lake in Nigeria finding an abundance between 1004–8329 items/m^3 for dry season and 201–8369 items/m^3 for raining season, respectively (Oni et al. 2020).

Concerning South America, only few studies were conducted, although two of the 20 top polluting rivers (e.g. River Amazon and River Magdalena) flow in Brazil, Peru, Colombia and Ecuador (Lebreton et al. 2017). Furthermore, in Colombia, 5.5–102.4 fibres/kg and 0.4–12.7 fragments/kg were found in shoreline sediments of River Magdalena (Silva and Nanny 2020).

Furthermore, another understudied region is the Mediterranean area. Plastic outputs from rivers into the Mediterranean Sea were analysed mainly by models (Lebreton et al. 2017); thus further research is needed to fill this gap. In this regard, considering that only few researches were conducted on rivers flowing into the Mediterranean Sea, few studies contribute to fill the knowledge gap on plastic contamination of Mediterranean rivers recently highlighted in the scientific literature (Guerranti et al. 2020). In central Italy, a range of plastic concentrations between 0 and 2.45 items/kg were assessed in River Mignone (Gallitelli et al. 2020). In addition, at the River Cecina estuary, 72–191 items/kg were found in sediments (Blašković et al. 2018), while 45–1069 items/kg in River Ombrone (Guerranti et al. 2017). In France, studies focussed on River Têt, finding 33–798 items/kg (Constant et al. 2020). In Spain, 1306–2798 items/kg were assessed in the River Ebro delta by Simon-Sánchez et al. (2019).

We highlight the knowledge gap, that many countries and areas resulted isolated and understudied. Therefore, a greater sampling effort in these regions is suggested to understand better plastic distribution and abundance.

10.3.4 Temporal Sampling of Microplastics in Sediment

Two different types of temporal analysis are conducted on MP in sediments: core analysis and superficial sediment samplings. Core analysis consists in collecting sediment cores from the sampled sites and then slicing them at regular intervals. Each slice is aged, and the MP load is evaluated by methods used for the analysis of MP in sediments. MPs are observed to decrease according to the age of the samples (older slices contain fewer MPs) (Li et al. 2020a, b, c). Older samples of MP in cores collected from a lake date back to the 1950s and highlight increasing levels of MP pollution in recent years (Turner et al. 2019).

Superficial sediment samplings are collected in the same sites but in different periods of the year and evaluate recent contamination. Similar to water, some factors related to seasonal changes probably influence the load of MP in sediments (Rodriguez et al. 2018). However, temporal information of MP in freshwater sediments is still strongly lacking.

10.4 Origin of Pollution

MP can be divided into primary and secondary, according to their origin. Primary MPs are products of factories, such as preproduction resin pellets, industrial scrubbers and cosmetic microbeads, while secondary MPs are products of plastic litter degradation (Cera et al. 2020). In fact, environmental exposure can degrade plastic litter by physical, chemical or biological factors (Wagner and Lambert 2018). A further category of MP is those generated as a consequence of product wear, such as the abrasion of tires and fibres from clothes (Eerkes-Medrano and Thompson 2018).

MPs may be dispersed into the environment by several sources and widely contaminate water, soil and air (Windsor et al. 2019a, b). The anthropogenic sources can be domestic and industrial discharges, wastewater treatment plants, abrasion of vehicle tires and agricultural plastic wastes (Li et al. 2020a, b, c; Fig. 10.4). In

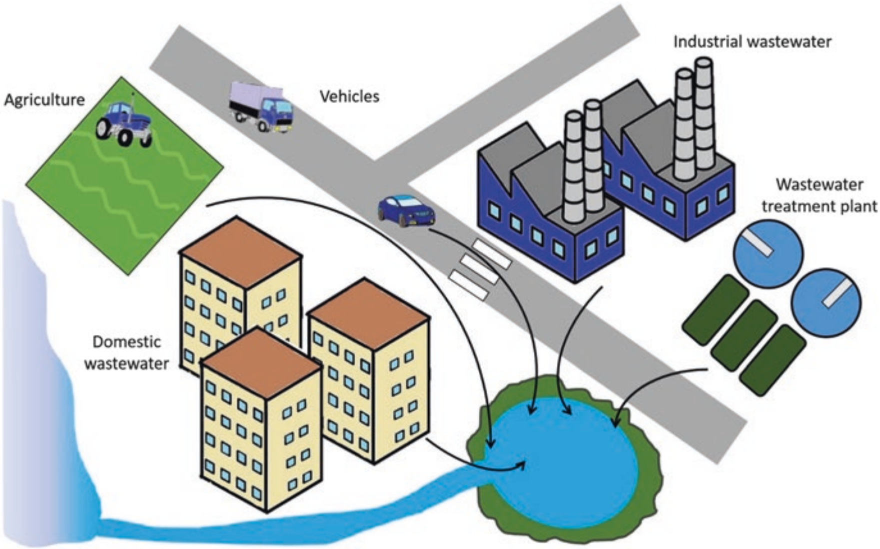

Fig. 10.4 Possible sources of microplastic pollution in aquatic environments. The black arrows indicate the factors of transport (e.g. rain, wind, human activities) of microplastic from the different sources

particular, environmental contamination by MP is enhanced by the low efficiency of urban and industrial wastewater treatments, use of plastic mulches and application of sewage sludge to fields (Cera et al. 2020). Natural factors such as wind, rain, floods, currents, flow rate and water chemistry contribute to the dispersal of MP (Xu et al. 2020a, b).

The oceans are the receiving basin of different types of MP, while the MP pollution of rivers and lakes is a specific reflection of the anthropogenic and environmental factors surrounding it. MP pollution in freshwater is influenced by the density and presence of human settlements, proximity to urban areas and atmospheric and hydrological conditions. In fact, the land-use composition of the territory surrounding the catchment affects the MP concentration in rivers. The human population density and the proximity of urban centres are the most important factors that influence the MP quantity in freshwater systems (Bellasi et al. 2020; Eerkes-Medrano and Thompson 2018). This trend can be observed both for lakes (Faure et al. 2015; Wang et al. 2017) and rivers (Kataoka et al. 2019). However, even low-density areas, such as remote lakes, are contaminated by a higher concentration of MP than expected, probably due to the lack of waste management facilities and the absence of emissaries (Zhang et al. 2016). In addition, wind can contribute to the transport and distribution of plastics in remote lakes (Free et al. 2014). Also, higher MP concentrations were observed in rural areas compared to urban areas (Nan et al. 2020; Yin et al. 2019). Therefore, although the populations are low density, the lack of proper waste management strongly affects the plastic pollution of freshwater systems.

Regarding point source pollution, several studies highlighted the wastewater treatment plants as an important source of MP in aquatic environments (McCormick et al. 2014; Murphy et al. 2016). In several studies the presence on MP in the effluents of wastewater treatment plants has been demonstrated (Xu et al. 2020a, b). In general, higher MP concentrations were observed downstream of wastewater treatment plants compared to upstream (Blair et al. 2017). Therefore, the low efficiency of urban and industrial wastewater treatments enhances the concentration of MP pollution in freshwaters. Among the sources of MP entering wastewater treatment plants, the washing of synthetic clothes constitutes the main input (Napper and Thompson 2016). Plastic fibres are linked to the release of synthetic clothing fibres from washing machines. Moreover, fibres represent one of the most common plastic pollutants found in aquatic environments and in every type of samples, such as water, sediments and biota (Browne et al. 2011; Gallitelli et al. 2020). Polyester, acrylic and polyamide are the main polymers of MP that derive from washing clothes (Browne et al. 2011). In the sewage treatment plant, fibres can accumulate due to the fact that bacteria cannot quickly decompose them and can reach the oceans via wastewater treatment plants and rivers (Napper and Thompson 2016). When the sewage sludge is used as fertilizer on the lands, the plastic fibres can be also released into the terrestrial environment (Mintenig et al. 2017; Windsor et al. 2019a, b). Moreover, the atmospheric fallout contributes to the spread of fibres on different ecosystems (Dris et al. 2016; Truong et al. 2021; Windsor et al. 2019a, b).

Once MPs are released into the aquatic environments, they may be taken up by biota through different physiological pathways and then translocated to diverse tissues or organs causing several effects (de Sá et al. 2018). The different properties of MP influence how they are ingested and accumulated by organisms. In particular, the dimension is an important factor which determines the possible interaction (Jâms et al. 2020; Xu et al. 2020a, b). The possible consequence of MP bioaccumulation is different through the trophic chain (Au et al. 2017). In particular, the benthic organisms are generally exposed to higher concentrations of MP, as sediment is a sink for MP (Cera et al. 2020; Leslie et al. 2017).

10.5 Impact of Microplastics to Freshwater Living Resources

The detrimental effects of MP on organisms are partially understood (Reid et al. 2019). Laboratory research support the investigations on assessing the impacts of MP on organisms, especially marine ones (Blettler et al. 2018). MPs have a negative impact on organisms based on their physical properties, the individual's developmental stage and the combined effect of MP with other pollutants (Xu et al. 2020a, b).

A few experiments examined the negative impacts of MP on freshwater species, for instance, on *Danio rerio* (Lei et al. 2018; LeMoine et al. 2018; Lu et al. 2016; Wan et al. 2019) and *Daphnia magna* (Rehse et al. 2016). It is to be considered that MPs do not always cause negative effects on organisms. For instance, experiments on *D. magna* conducted by Rehse et al. (2016) did not provide observation of an effect. Similarly, the presence of MP did not cause toxicity in *Dreissena polymorpha* (Magni et al. 2018).

This chapter describes recent findings on the impacts of MP to freshwater organisms from field observations in lentic and lotic habitat.

10.5.1 Freshwater Taxa Examined by Lentic or Lotic Habitat

Lentic ecosystems are less studied than lotic ecosystems in regard to the exposure of biota to MP (Cera et al. 2020). The investigated biota evaluated the interactions and impacts of MP on microorganisms, invertebrates and vertebrates. Microorganisms include bacteria, cyanobacteria, algae, plants and fungi. They are studied to determine the characteristics of colonising MP especially on lotic freshwaters. Similarly, the invertebrates are more studied in rivers than lakes. Among invertebrates, Mollusca is the main investigated taxonomic group, most of all bivalves (Cera and Scalici 2021). Particularly, the scientific literature on lentic freshwater examines only Bivalvia. Furthermore, scientific literature examines also Arthropoda (Insecta and Crustacea), while one study is on Annelida. Freshwater vertebrates are more investigated than invertebrates with regard to MP exposure, especially in lentic freshwaters. In particular, fish are the most studied taxa. Birds and amphibians are not much studied, and no study concerns reptiles nor mammals. Few information on plants are available (Cera and Scalici 2021).

10.5.2 The Taxa Examined by Scientific Literature

10.5.2.1 Microorganisms

Microorganisms are especially studied in relation to the 'Plastisphere', the phenomenon described by Zettler et al. (2013). The 'Plastisphere' is the community living on buoyant plastics. It was described in marine waters, but the same phenomenon occurs also in inland waters. Several studies described interesting aspects in relation to the changes of the community composition living on MP in lakes. For instance, the abundance and functional diversity of the microbial community living in water were compared to the one on MP in three types of lake: oligo-mesotrophic, eutrophic and dystrophic. The results suggested an environmental impact of MP on heterotrophic activities and possibly on the carbon cycle in lakes (Arias-Andres et al. 2018). Furthermore, other factors influence the community, such as depth of buoyant plastics (epi-, meta-, hypo-limnion), and the type of polymer (polyethylene terephthalate, polystyrene, polyethylene) (Leiser et al. 2020). Several studies also evaluated the effects of effluents from wastewater treatment plants on the microbial assemblages on the MP in rivers, finding an impact (McCormick et al. 2014; Hoellein et al. 2017; Oberbeckmann et al. 2018; Kettner et al. 2019; Kelly et al. 2021). Larval or juvenile stages of Annelida, Rotifera and Nematoda are also possibly living on buoyant MP (Kettner et al. 2019).

10.5.2.2 Invertebrates

Mollusca is the main investigated invertebrate phylum, above all bivalves but also gastropods (Berglund et al. 2019; Xu et al. 2020a, b). A high density of human population and the contaminated effluents of sewage treatment plants are suggested to impact the concentration of MP in bivalves (Berglund et al. 2019; Domogalla-Urbansky et al. 2019). The type of MP found is mainly fibre, although spherules also occur (Berglund et al. 2019). Regarding spherules, it is suggested that a selectivity occurs, that is, only large bivalve species (>3 cm) are able of accumulating MP spherules (Schessl et al. 2019).

To date, one study analyses the interactions between Annelida and MP (Hurley et al. 2017). The study showed no evident effects on their fitness. Regarding the shapes of MP, fibres were the most abundant. Regarding polymers, polyester, polyethylene terephthalate and polystyrene were the main types found.

Among arthropods, the ingestion of MP by crustaceans (e.g. Nan et al. 2020) and insects is investigated (Simmerman and Coleman Wasik 2019; Windsor et al. 2019a, b). The contamination of macroinvertebrates is probably influenced by the volume of the river flow, the presence of wastewater treatment plant and the distance from cities (Simmerman and Coleman Wasik 2019; Windsor et al. 2019a, b). However, the taxonomic group and biomass also explain the variability of contamination (Windsor et al. 2019a, b). Moreover, some insects called caddisfly can also use MP to build casing (protective involucres) (Ehlers et al. 2019; Gallitelli et al. 2021; Tibbetts et al. 2018).

10.5.2.3 Vertebrates

Fish ingestion of MP is the main topic of investigation in vertebrates. The contamination of the gastrointestine of fish varies greatly according to the study area, such as from 7.5% to 100% of contaminated fish (e.g. Faure et al. 2012; Sanchez et al. 2014; Biginagwa et al. 2016; Faure et al. 2015; Campbell et al. 2017; Zhang et al. 2017; Horton et al. 2018; Xiong et al. 2018; Roch et al. 2019; Slootmaekers et al. 2019; Yuan et al. 2019; Khan et al. 2020). The Asian region is generally highly contaminated, but the methods used to identify MP may also influence the outcome. The occurrence of MP in fish could be positively enhanced by environmental contamination, niche connected with benthic habitats, feeding activity, complex morphology of the gastrointestine (GI) and the length of fish (Peters and Bratton 2016; Silva-Cavalcanti et al. 2017; McNeish et al. 2018). It is unsure if the feeding guild affects the contamination of fish (Andrade et al. 2019; Hurt et al. 2020). In addition, the presence of MP positively correlates with the ingestion of food items, suggesting that the activity of feeding increases the chances of MP incidental ingestion (Jabeen et al. 2017; Peters and Bratton 2016; Silva-Cavalcanti et al. 2017). The presence of an accidental uptake is also supported by Roch et al. (2019), as in their study the biotic and abiotic factors influence the outcome of the ingestion limitedly. Impacts on livers are also an important topic of study, exploiting a different perspective in fish toxicity due to MP. In addition to studies on ingestion, MPs in the livers and muscles of fish were investigated. Some MP fragments were detected in fish livers (Collard et al. 2018). Furthermore, histological observations revealed changes in livers of the MP-contaminated areas compared to those obtained from a control area (Li et al. 2020a, b, c). Instead, no MPs were found in muscles (Collard et al. 2018; Park et al. 2020).

Few studies examine the exposure of amphibians. One study sampled 31 GI contents of different species of amphibians, all anurans, but no plastics were found in their digestive tracts (Schessl et al. 2019). Instead, MPs were found in the diet of adult *Triturus carnifex* Laurenti (1768) (Iannella et al. 2020) and tadpoles (Karaoğlu and Gül 2020). Further evaluation of the occurrence of MP in anurans and other amphibians is suggested, especially as it is commonly known that they are an endangered taxonomic group.

Birds were analysed since 2012 (Faure et al. 2012); however, there are not many studies available. Three species of birds had ingested MP in Lake Geneva, but their number is low (n <10 individuals) (Faure et al. 2015). A larger study examines 350 specimens belonging to 17 species (included a marine one) in Canada. These birds ingested several anthropogenic debris, and MP had an occurrence of 9.7% (Holland et al. 2016). Chicks can also ingest MP, as demonstrated by Brookson et al. (2019) in the Laurentian Great Lakes, and the occurrence of MP is suggested to be higher than adults.

To date, reptiles or mammals have been rarely studied for MP in freshwaters. Therefore, it is unknown whether negative impacts affect them.

10.5.2.4 Plants

Plants are rarely studied in the field. However, the effects of MP on vascular aquatic plants were evaluated in laboratory (Dovidat et al. 2019; Mateos-Cárdenas et al. 2019). For instance, the growth of shoots and roots are inhibited by MP (Kalčíková et al. 2017; Pflugmacher et al. 2020). Moreover, MP can be adsorbed by plants, thus potentially providing a depurative activity. However, the MP accumulated in plants can feed animals feeding on those plants and enter the food web, potentially creating negative effects (Kalčíková 2020).

Regarding field studies, Chlorophyta and other microorganisms, such as Cyanophyta and algae (Bacillariophyta, Cyanophyta, Cryptophyta, Euglenophyta, Pyrrophyta), can colonise MP according to the season (Chen et al. 2019).

10.6 Conclusion

Microplastics contaminate a wide range of freshwater habitats. Although many areas of the world are understudied, microplastics are detected in almost every waterbody examined by actual scientific literature. Usually the presence of areas densely populated by humans affects the contamination of water bodies. In addition, environmental factors, such as wind, and anthropogenic factors, such as low efficiency of wastewater treatment plants, play an important role in spreading the pollution. Due to local differences in natural and anthropogenic factors, the pollution of freshwater by microplastics varies greatly according to the sampled site. Temporal variations, for instance, seasonal differences, are also emerging information which are to be considered in future research. Even more, the standardization of sampling methods and analysis of the occurrence of microplastics are a priority in this field for allowing the precise comparison of results worldwide.

In addition, as microplastics widely contaminate freshwaters, aquatic biota is considered highly disturbed by them. Since biota can be negatively affected by microplastics, the main future goal should be reducing the load of microplastics in freshwater by either lowering their release into the environment or increasing the efficiency of treatment plants.

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Chapter 11
Occurrence of Microplastic Pollution in Coastal Areas

Cem Çevik and Sedat Gündoğdu

Abstract Due to the increasing plastic production in the world, it is predicted that the amount of microplastic will increase in the future in all ecosystems in the world, especially in aquatic ecosystems. It is obvious that the negative effects of this pollution will increase in the same way. In this section, the coastal ecosystem, which is an important part of aquatic ecosystems, and the status of microplastic pollution in beaches, estuarine regions and sea meadows, which are important parts of this ecosystem, have been compiled from recent publications.

Keywords Estuaries · Microplastic pollution · Seagrass meadows · Marine pollution

11.1 Introduction

Plastics are materials that are made out of synthetic or semi-synthetic organic compounds. International Union of Pure and Applied Chemistry (IUPAC) defines plastics as polymeric materials that might contain other materials to improve performance. The word plastic has its roots in the Greek words 'plasticos' and 'plastos', which mean 'malleable' (Rocha-Santos and Duarte 2017). Ease of production, low cost, resistance against water and many chemicals and durability against temperature and light effects make plastics superior to other materials.

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These characteristics allowed plastics to replace many frequently used materials like wood, paper, stone, leather, metal, glass or ceramics in our lives. Today, plastics have a wide range of uses in many fields, from a simple bookmark to spaceships. While the value of plastics for humanity cannot be denied, certain types of widely used plastics like single-use plastic products or packaging materials tend to accumulate in the environment. It is estimated that 10% of the waste collected by municipal authorities worldwide from various settlements consists of plastics. Although some of these plastics are recycled, most of them are buried in landfills, taking hundreds of years to degrade. However, the greatest concern is the plastics that end up in seas and oceans that make up 10% of the plastics produced. These plastics that end up in the sea, called macroplastics, tend to accumulate in areas where surface currents come together, like the Great Pacific Garbage Patch, and have been subject to long-term environmental studies. This floating material is churned by wind and wave action and spread to very large areas. This both causes unsightly scenes and harms tourism and causes significant environmental harm due to marine life getting entangled, hooked, swallowing, choking on them or helping invasive species spread. In addition, they cause problems for various maritime activities like power production, tourism, aquaculture and especially maritime and fishing.

It was reported that there are species that live connected to these accumulated masses of plastics, distinct from free-floating microbial populations found in the ocean, called the 'plastisphere' (Amaral-Zettler et al. 2020).

11.2 Microplastics

The term 'microplastics' was first used in a report by the USAF Materials Lab in 1968. In this report, microplastics were used to describe particles created due to the deformation of plastic materials under high tension (Webb et al. 2013).

The presence of microplastics in marine environments first entered the global consciousness with the report of the high amount of tiny plastic particles floating in the surface waters of the Sargasso Sea (Carpenter and Smith 1972). In 2004, these tiny particles were described as microplastics (Thompson et al. 2004). Later on, the Steering Committee of the National Oceanic and Atmospheric Administration (NOAA) Marine Debris Program defined microplastics as plastic particles smaller than 5 mm (Arthur et al. 2009). Afterwards, various sizes of plastic particles were standardized. According to the standard, macroplastics were defined as particles with sizes ≥25 mm, mesoplastics as <25–<5 mm, microplastics as ≤5–1 mm, mini-microplastics as <1 mm–1 μm and nanoplastics as <1 μm (Webb et al. 2013). While the main source of microplastics in aquatic environments is microplastics generated by the fragmentation of large plastic pieces, microplastics are also produced industrially for various purposes. Industrially produced microplastics were defined as primary microplastics. The others were defined as secondary microplastics.

11.2.1 Primary Microplastics

Primary microplastics are produced mostly in the form of microbeads. The purpose of the production of microbeads is to add intentionally in cosmetics, personal care products, dermal exfoliators, cleaning products and sanding equipment. Other types of primary microplastics are pellets produced to melt and cast to create larger plastic products. Another is synthetic fibres produced for clothing industry. All these microplastics are transported to freshwater and marine environments by wind or wastewater from cities. Due to the widespread use of microplastics over the years, these have become widespread at all sea compartments.

11.2.2 Secondary Microplastics

Secondary microplastics are irregularly shaped plastic pieces generated by the fragmentation of larger plastic products, like plastic bags, cases, bottles, ropes and nets (Reisser et al. 2013). Over time, larger plastics are fragmented into smaller pieces due to solar ultraviolet radiation that caused degradation or mechanical action such as the tide. An experiment has shown that a 1 cm^2 polystyrene coffee cup cover can produce 126,000,000 nanoparticles on average over 56 days when exposed to 320–400 nm ultraviolet light for 24 h at 30 °C (Lambert and Wagner 2016). This shows that these tiny plastic fragments can be easily distributed into the entire water column and mistaken for food and consumed by many marine organisms (Bergami et al. 2017).

11.3 Coastal Systems

While coastal ecosystems are defined as areas close to shore where fresh and saltwater mix up, there is no single definition of a coastal area. Some sources define them as land areas close to and affected by the sea, and others as sea areas close to and affected by the land. Another definition describes it as the continental shelf up to 200 m, upper and lower bounds of the tidal zone, periodically inundated by the sea and the area right next to these. This system has two sub-regions: areas close to shore, fresh and saltwater mix and open ocean sea areas. In areas close to shore, human settlement is heavy, which negatively affects the marine ecosystem. On the other hand, sea areas supply two-thirds of marine fishing around the world. The region described as the coastal ecosystem covers many different ecosystems, including freshwater and bitter wetlands, mangrove forests, river mouths and estuaries, swampland, lagoons, saltwater pools, rocky and muddy tidal flats, sandy beaches, coral reefs, seagrass, kelp forests, coastal islands, semi-enclosed seas and waters on the continental shelf close to the shore.

While the coastal areas including these systems only make up 20% of the land area (excluding Antarctica and the water column), over 2 billion people, 40% of the world population, live in this region. These habitats are an essential source of protein for people who live in coastal areas and make up a significant chunk of the population through fishing. However, most of these habitats are either unprotected or marginally protected. As a result, most of the services provided by these habitats in many areas are in danger.

11.4 Microplastics in Coastal Systems

Microplastic pollution in coastal ecosystems is significant, just like all other ecosystems around the world. Especially in coastal areas where tourism, aquaculture, fishing and port activities are heavy, this problem is more prominent. Due to various biological and chemical effects, complex hydrodynamic structures and different geographic conditions, the distribution of microplastics can be significantly different between various areas in these regions. In addition, there is a large variety of sources for microplastic pollution in coastal environments. The most important among these are land-based input through rivers, sewer systems and waste dumping, and marine-related sources such as fishing and shipping. Investigation of the source of microplastic pollution in coastal areas is considered an essential research area to determine microplastics' spatial and temporal distribution. This section focuses on microplastic pollution in sea meadows, beaches and estuarine regions in the coastal ecosystem.

11.4.1 Seagrass

Seagrass, one of the most productive ecosystems worldwide, cover 177.000 km^2 around all coasts of the world except Antarctica up to a depth of 40 m (Duffy et al. 2019). The only flowering plants in the marine environment there are 72 species of seagrass (Duffy et al. 2019). Seagrass offers invaluable benefits to marine life and people, some of which are listed below.

On soft surfaces, seagrass slows down water flow, holds inorganic and organic particles like a filter and helps these particles accumulate in and on the sediment, increasing sedimentation (Serrano et al. 2013).

It binds settling particles to the sediment, capturing them; it also slows shore erosion by reducing wave energy, produces O_2 by absorbing CO_2 and helps reduce global warming. In addition, seagrass beds serve as the nesting, feeding and hiding areas for many marine animals, primarily fish, and benefit fishing and tourism industries (de los Santos et al. 2021, 2020). However, this peerless ecosystem is vulnerable to anthropogenic effects like excessive nutritional element input, decreasing water quality, physical disruptions and sea pollution. As mentioned above, due

Table 11.1 Quantities of microplastics found in seagrass and unvegetated seabed in other parts of the world

Location	Seagrass MP abundance (MP/kg dw)	Unvegetated MP abundance (MP/kg dw)	Sources
China	196.7 ± 16.1–780.2 ± 147.0	93.3 ± 15.3–267.1 ± 60.5	Li et al. (2020)
Scotland	300 ± 30	110 ± 20	Jones et al. (2020)
England and Wales	215 ± 163	221 ± 236	Unsworth et al. (2021)
Spain (Santa Maria)	68–362	No information	Dahl et al. (2021)
Spain (Roquetas)	2173	No information	Dahl et al. (2021)
Spain (Agua Amarga)	3819	No information	Dahl et al. (2021)

to their characteristics as particle holders in marine environments, seagrass beds serve as a factor holding and accumulating microplastics in the sediment (de los Santos et al. 2021, 2020). However, according to studies, nothing suggests that seagrass sediment accumulates more microplastics than areas without vegetation. While some studies worldwide show higher microplastic amounts in sediments under seagrass beds than areas without vegetation, there are also studies where microplastic amounts in areas without vegetation are pretty high. Results of some studies from sediments under seagrass beds around the world are given in Table 11.1.

It can be seen here that in studies done in Wales/Britain and Portugal, higher amounts of microplastics were found in areas without seagrass. Studies show that the highest amount of microplastics can be found in the research done in China, while the lowest amount was found in Florida, USA. It was suggested that the low microplastic count found in the study from Florida was due to sampling from an area that was far from any river mouths and the industrial regions. It was also mentioned that the high microplastic amount found in the study in China might be due to some sampling areas being close to cities and the agricultural regions and the study using a more efficient separation method. It was even suggested that if previous studies used the separation method used in the study from China, the microplastic amount found in those might have been higher.

A few mechanisms were suggested to explain how seagrass holds floating microplastics and collects them in the sediment. For example, in the experimental study by Seng et al. (2020), seagrass and macroalgae were compared, and higher amounts of microplastic accumulation were found under seagrass. This was thought to be due to high quantities of epibiotic organisms in seagrass beds. There is a close relationship between the amount of microplastic accumulation and the amount of epibionts present.

Oberbeckmann et al. (2014) and Rummel et al. (2017) report that biofilms created by bacteria and epibiotic organisms affect microplastic accumulation. Another experimental study reported that the plastics' holding capacity depends on microplastic density and water movement (de los Santos et al. 2021). In addition, it was stated that structures called sea balls or Neptune balls created from dead leaves and roots by water movement also capture microplastics and cause accumulation in the sediment and beaches (Sanchez-Vidal et al. 2021).

When the most frequently found types of microplastics in studies conducted on areas with seagrass beds are checked, it can be seen that in areas with or without vegetation, the most frequently found type of microplastics is fibres, followed by filaments, films and foams. For example, a study in Britain found 92% fibres, while another study in Indonesia found 84% filaments (Unsworth et al. 2021). It was reported that synthetic microfibres make up 80% of all microplastics, and the primary vectors for these are high population cities and wastewater systems (Bessa et al. 2019). In conclusion, even considering seagrass's high particle capture capability, studies do not support the hypothesis of higher microplastic accumulation in areas with vegetation. The hypothesis that claims seagrass with longer leaves collects more microplastics, and when these leaves die and rot, the microplastics on them settled in the sediment was also not confirmed. A detailed study is required in this area as well. The discovery of high microfibre content in sediment samples taken from both vegetation and non-vegetation areas fits with the hypothesis that areas close to estuaries, high population density and wastewater systems would have a higher fibre density.

It is assumed that where microplastics accumulate in marine ecosystems depends on microplastics alongside the local physical and anthropogenic conditions. This might cause a higher accumulation of microplastics in seagrass sediments in some areas.

There are still some significant gaps in factual data that can explain the sources, distribution and accumulation dynamics of microplastics in the marine environment. It must be emphasized that to understand these dynamics fully, more detailed additional studies are necessary.

11.4.2 Microplastics in the Beach Sediments

A significant part of the coasts around the world (outside of the areas covered by glaciers) consist of sandy beaches (approximately 31%) (Luijendijk et al. 2018). As these are attractive areas, people want to use these areas more than other types of beaches. In addition, beaches are one of the unique ecosystems that support biodiversity. A single beach can host hundreds of invertebrates, including micro- and meiofauna. Beaches also provide essential ecological services such as filtering the seawater, ensuring the recycling of nutritional elements, supporting coastal fishing and providing critical habitats (nesting and feeding grounds) for endangered species such as turtles and birds. However, sadly, beaches worldwide are negatively affected by various factors such as population growth, global warming, erosion, sand extraction and plastic pollution.

First studies about microplastics accumulating on the beaches are done soon after floating plastic fragments were reported by Carpenter and Smith (1972). Carpenter and Smith (1972) reported the presence of nurdles in various beaches around the world (as the term microplastics was not used until 2004, these were reported as small particles).

In the 2000s, after discovering the ubiquity of microplastics in all marine ecosystem compartments and their harmful effects on all wildlife and humans, the number of studies to measure the presence of microplastics in the sediments covering the coastal areas between the sea and land increased. In these studies, primarily conducted in sandy beaches, the distribution, density, types, varieties and colours of the microplastics in the sediment were studied. However, there are difficulties in comparing these studies due to the differences in sampling methods, laboratory analysis methods and targeted microplastic sizes. Also, since factors such as the location and proximities of the studied beaches are different, it is difficult to make comparisons.

In this context, some studies from Asia, the Mediterranean and various countries worldwide were made, and the results are given in Table 11.2.

The high incidence of microplastics in Lebanese coasts is reported due to different microplastic separation methods and poor waste management in Lebanon. Microplastics found in Datça coasts were higher than the levels found in the rest of the Mediterranean but anywhere else until the studies conducted in 2019. As the region is far from large cities, industry and large ports, authors suggested that this is due to the population increase in the region during the tourism season, the yacht tourism in the area and the geological location of the region. In the study conducted in China, the high microplastic amounts were reported to be due to insensitivity towards the use of plastics and environmental protection and a fluorescence microscope that can detect plastics very well. In Vietnam, microplastic levels are the highest recorded so far. Authors suggest that this is due to the sampling location (Da Nang) being both a residential area and a tourism area, and wastewater effluents are flowing into the area. In addition, it is thought that the treated and untreated industrial wastewater that comes through Han and Cu De rivers is believed to affect microplastic pollution. In most of these studies, fibres are found in higher amounts compared to other microplastic types. The primary source of synthetic fibres is microfibres that come out of clothes washed in washing machines everywhere worldwide. A study suggests that approximately 700,000 fibres are released per wash (Napper and Thompson 2016; Özkan and Gündoğdu 2021). While some of these fibres are captured in facilities that treat residential wastewater, a significant amount reaches aquatic environments like seas, lakes and other water sources through rivers (Gündoğdu et al. 2018b). In addition, another reported source of fibres is fibres created by the degradation of ropes and nets used for fishing, which is thought to make up 18% of the marine waste (Lusher et al. 2017). While ratios vary, aside from fragments and fibres, film-, foam- and pellet-type plastics are also found in smaller amounts. The most reported colour for microplastics is blue, but black, white, transparent, green, red, yellow, pink and orange microplastics are also reported. Chemical compositions of microplastics are mainly polyethylene (PE),

Table 11.2 Quantities of microplastics found in beaches in other parts of the world

Location	MP items/kg dw	Sources
Hong Kong	5595	Fok and Cheung (2015)
China (Beibu Gulf areas)	5000–8714	Qiu et al. (2015)
China (Bohai Sea)	102.9–163.3	Yu et al. (2016)
China (Hong Kong)	16.8 ± 5	Lo et al. (2018)
Japan (Tokyo Bay)	1800	Matsuguma et al. (2017)
South Korea	(1–5 mm) 0–20, (0.02–1 mm) 1400–62,800	Eo et al. (2018)
Singapore	36.8	Ng and Obbard (2006)
Qatar	62	Veerasingam et al. (2020)
Belgium	52.8–213.4	Claessens et al. (2011)
Slovenia	213.2	Laglbauer et al. (2014)
Germany	13–532	Stolte et al. (2015)
Russia	1.3–36.3	Esiukova (2017)
Mexico	16–312	de Piñon-Colin et al. (2018)
Brazilian coast	2.4–17.4	Maynard et al. (2021)
Portugal	5–320	Chouchene et al. (2021)
Aegean Sea (Eastern Mediterranean)	275.75	Kaberi et al. (2013)
North Central Mediterranean (Gulf of Trieste)	155.6	Laglbauer et al. (2014)
Central Mediterranean Sea (Aeolian Archipelago)	151–678.7	Fastelli et al. (2016)
Aegean Sea (Dikili, İzmir, Turkey)	248	Lots et al. (2017)
Western Mediterranean	147	Lots et al. (2017)
Eastern Mediterranean	387	Lots et al. (2017)
Central Mediterranean (Eastern Adriatic Sea)	32.3–377.8	Blaskovic et al. (2016)
Central Mediterranean (Northern Adriatic Sea from Caorle (Italy)	137–703	Renzi et al. (2018)
Central Adriatic Sea (Silba Island)	180–526.7	Renzi et al. (2019)
Eastern Mediterranean (Lebanese coast)	2433	Kazour et al. (2019)
Aegean Sea (Datça Peninsula)	4617.6	Yabanlı et al. (2019)

polypropylene (PP), polyester (PEST), polyamide (PA), polystyrene (PS), polyethylene terephthalate (PET), polyvinyl chloride (PVC) and polystyrene (EPS). The most frequently reported polymer types are PE and PP, while the ratios of others vary from study to study.

11.4.3 Microplastics in Estuary Areas

Estuary areas, where freshwater from the rivers flow into the salty water of the sea ecosystem, are very dynamic, fertile areas that are of critical importance to other marine habitats. Around the world, 1200 main estuaries covering an area of 500,000 km^2 are defined (Adey and Loveland 2007). Estuaries are one of the most fertile areas among marine ecosystems. The main reason for this is the inflow from rivers and land drainage that enriches the area with nutritional salts. Estuary area, sediments and waters provide various direct and indirect ecosystem services. The services that the estuaries provide are breeding and feeding grounds for many organisms, including fishes. The area also provides recycling of nutrients; food to millions of people living around estuaries through fishing; for aquaculture, carbon capturing; and climate change reducing effects. In addition, estuaries act as a buffer zone protecting other ecosystems and habitats from events, e.g. tsunami and coastal erosion. This system also acts as a biological filtering system, reducing many chemical and organic pollutants to marine ecosystems.

Estuaries are also an accumulation area for microplastics. While significant amounts of microplastics come from marine sources like fishing and maritime activities, the most considerable source is rivers, which transport 1.15–2.41 million tonnes of plastics annually. For this reason, estuaries are one of the coastal areas where plastic pollution accumulates the most (Lebreton et al. 2017; Meijer et al. 2021).

There have been many studies that investigated the microplastic density in various estuaries around the world. Most of these studies examined the surface waters of estuaries, and microplastic densities reported in these studies vary greatly, just like other studies investigating beaches, sediments and surface waters. For example, Suteja et al. (2021) conducted in the Benoa Bay in Indonesia found an average microplastic density of 0.62 particles/m^3. However, while some studies conducted in some countries (Turkey (*Küçükçekmece Lagoon*), 33.000 particles/m^3 on average (Çullu et al. 2021); *China*, 930,000 particles/m^3 during the rainy season and 497,000 particles/m^3 during the dry season (Han et al. 2020); the *USA*, 940 particles/m^3 on average (McEachern et al. 2019); and *Argentina*, 139 particles/m^3 on average (Pazos et al. 2018)) found very high levels, some other studies (*UK Open Waters Northeastern Atlantic*, 0.14 particles/m^3 on average (Maes et al. 2017); *English Channel*, 0.27 particles/m^3 on average (Cole et al. 2014); and *Tamar Estuary* (*Southwest England*), 0.028 0.27 particles/m^3 on average (Sadri and Thompson 2014)) found very low levels. It can be said that the differences in methodology and the hydrodynamic characteristics of the water are the main reason for the difference in microplastic pollution levels. Regions close to the mouths of large rivers and lagoons naturally have a higher level of microplastic pollution. This is directly related to the 1.15–2.41 million tonnes of plastics transported to aquatic environments by rivers every year. As a result, estuaries are the hotspot for plastic accumulation (Lebreton et al. 2017). When these studies are examined, it can be seen that studies conducted in areas with high microplastic pollution were done at locations close to rivers that pass large and crowded cities, wastewater treatment facilities and landfills. It can be seen that these factors are less prominent or absent in areas with lower pollution levels. When sampling methods are investigated, it can be seen that in studies that found high levels, Niskin bottles, Van Dorn water samplers, rosettes or buckets were used. In contrast, studies that found low levels mostly used pumps, plankton nets manta nets. However, when the sizes of microplastics obtained for the studies were examined, it can be seen that studies uses water sampling samples and detects even the smallest plastic fragments since there is no pre-selection. However, when nets with a mesh size of 300 µm are used, microplastics smaller than 300 µm can easily escape, and as a result, only microplastics larger than 300 µm can be detected.

The sizes of microplastics found in the studies were also different. For example, microplastics smaller than 300 µm easily got out of nets with an eye size of 300 µm during fieldwork. However, when sampling is done via steel bucket, even the smallest microplastics can be sampled since no net is used. Afterwards, sampled microplastics are captured on 50, 100 and 200 µm filter papers during laboratory separation and sifting. For example, Han et al. (2020) found that particles smaller than 200 µm made up 87% of the sample. In most studies using nets with a mesh size of 300 µm, only microplastics have 300, 500 and 1000 µm sizes sampled. On the other hand, various researchers mentioned that other factors (e.g. the direction and speed of prevailing winds, biofouling and hydrodynamic characteristics such as currents, wave action and tides) are also affecting microplastics concentrations (Gündoğdu et al. 2018a; Leiser et al. 2021).

The types of microplastics reported in studies may also depend on the geographic locations of the estuaries and the presence of activities like fishing. For example, Suteja et al. (2021) reported the types of microplastics in the rainy season as 69.72% fragment, 18.35% foam, 8.26% fibre and 3.67% granules. On the contrary, they also reported 76.19% fragment, 15.86% foam, 4.76% fibre and 3.17% granules in the dry season (Suteja et al. 2021). In a study by Çullu et al. (2021) conducted in the Küçükçekmece Lagoon in Turkey, in the rainy season, fibres made up 49.71%, and fragments 41.41%, while in the dry season, fibres made up 44.92%, and fragments 41.78% of the sample. Cheung et al. (2018) reported a higher amount of foam for China (Hong Kong). Researchers suggest that this is due to the increased use of foam in storage, transportation and packaging in the area's fishing industry. The high amounts of fragments in the aquatic environment are due to fragmentation of macroplastic waste caused by all kinds of human activities and the burning activities in landfills (Barnes et al. 2009; Cheung et al. 2018; Cordova et al. 2019). It is also known that these plastics can be transported to aquatic environments by rivers and drainage from landfills (He et al. 2019). On the other hand, fibres are transported to the aquatic environments mostly by rivers receiving waters from wastewater treatment facilities and wind.

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Chapter 12
Modeling the Fate and Transport of Microplastics in Coastal Areas

Berna Ayat, Burak Aydoğan, and Sedat Gündoğdu

Abstract Numerical models are strong tools to understand the dynamics better and analyze the sources, transport, receptors, and consequences of microplastics in the coastal environment. Complex dynamics and interactions of biotic and abiotic components of aerial, terrestrial, aquatic, and benthic processes make the numerical modeling of microplastic transport challenging. In this chapter, we presented an overview of modeling aspects consisting of sources and sinks of microplastics, key processes affecting their transport and fate, types of coastal systems, physical properties of microplastics important for the numerical modeling studies, types of modeling approaches, data requirements, and tools for numerical simulations.

Keywords Modeling · Fate · Microplastic · Coastal areas

12.1 Introduction

Microplastic pollution in the marine environment is a growing concern. Microplastic particles floating or settled in the sea affect both biotic and abiotic components of the marine environment through absorption by biota, entanglement, and colonization. There is also strong evidence that microplastics support microbial growth (also

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known as plasticizers) and adsorb primary hydrophobic contaminants on their surfaces. The first addresses of floating microplastic particles are the coastal environment. According to Liubartseva et al. (2018), the relative contributions of floating plastic that arrive at the coastlines can be divided into three groups according to the origins of plastics. It appears that the majority of the plastics come from the closest terrestrial inputs (Gündoğdu et al. 2018; Liubartseva et al. 2018). For instance, in the model implemented by Liubartseva et al. (2018), the flux onto the coastlines is bigger than the flux to the bottom in the Mediterranean.

Modeling studies are essential in understanding the fate of microplastics in the marine environment. Supporting the modeling studies with actual field data is very important in having more detailed information about the distribution of microplastics. Thus, it is also crucial to reveal the sources of microplastics and prevent their leakage into the natural environment at the source.

This chapter is aimed to identify the sources, sinks, and the transport processes affecting the microplastics in coastal environments, to describe the general considerations in the selection of the proper modeling approaches and tools, and to identify the currently available modeling approaches and softwares in the modeling of the fate and transport of the microplastics in the coastal environments. The main objectives of the microplastic modeling are:

- To identify the possible sources, sinks, and affecting processes within the study area.
- To identify the knowledge gaps and to propose scientific methods to increase the understanding of the key processes to fill these gaps.
- To help the improvement of the monitoring programs.
- To provide data under different spatial and temporal scales as well as different climate conditions.
- To evaluate the impact of any developments or prevention programs on the spatiotemporal distributions of the microplastics within the study area.
- To predict the scale of microplastic pollution in the area over different temporal scales (days to centuries).
- To evaluate the effectiveness of the proposed management actions in reducing microplastic pollution.

12.2 General Aspects of the Modeling of the Fate and Transport of Microplastics in Coastal Environment

12.2.1 Sources and Sinks of Plastics in Coastal Areas

Microplastics found in coastal waters or transition land may be originated from different sources such as rivers, wind-driven transport, surface runoff, maritime traffic, fishing and aquaculture, coastal landfills, reclamations, recreational and tourism, construction and industrial effluent, wastewater treatment effluents, and sewage sludges. When defining the sources of the microplastics, it is also important to mention their formation, e.g., primary microplastics and secondary microplastics. Primary microplastics are produced at that size, and the secondary ones are produced by the fragmentation (or breakdown) of the plastic debris into macro- and mesoplastics and then turn into microplastics. Due to the difficulties in detecting and estimating primary or secondary microplastics separately, plastic pollution sources can be considered the sources for microplastics in general.

12.2.2 Key Processes in the Modeling of the Fate and Transport of Microplastics in the Coastal Environment

Microplastics in the coastal waters are subject to various processes identifying their fate in the coastal environment. The hydrodynamics of the coastal environment determines the advective, diffusive, and dispersive transport of the particles. These processes cover advection, diffusion, and dispersion. Advection defines the mechanical transport of particles by the flow. Hence the flow velocity determines the velocity of the particle transport. Diffusion determines the transport of the particle's random movements and occurs from the high concentration flow regions into the low concentration regions. Random movements of the particles may originate from Brownian motions or turbulence eddies. The first one is called molecular diffusion, and the latter is called "turbulent/eddy diffusion." In coastal waters, different scale eddies may arise due to the complex turbulent flow environment. Hence the eddy diffusion gains high importance in hydrodynamic modeling. Other biotic or abiotic processes affecting the fate and transport of microplastics are sedimentation, suspension, resuspension, accumulation, beaching, entrapment, turbulent mixing, turbulent entrapment, biofouling, embrittlement, fragmentation, heteroaggregation, and mineralization. Sedimentation can be accelerated by heteroaggregation with natural colloids, clays, and other high-density suspended particles which will lead to faster sedimentation of the plastic particles that are captured in the aggregate (Besseling et al. 2014; Besseling et al. 2017; Kooi et al. 2018). Key processes in the modeling of the fate and transport of microplastics in coastal areas are shown in Fig. 12.1.

Plastic litter goes under the fragmentation process via mechanical abrasion, biological decomposition, and photo-oxidation. Photo-oxidation causes the embrittlement of plastics by breaking the polymer chains at the sun-exposed surface generally penetrating about 100 μm. Ter Halle et al. (2016) reported that the smaller microplastics and cubic microplastics are fragmented much faster than the parallelepipeds as the latter one floats only on one surface limiting the sun exposure.

Another important process causing the change in shape, size, and weight of the particles is the aggregation during which the microplastics gathered together with other particles (organic or inorganic). These changes in particle properties affect its

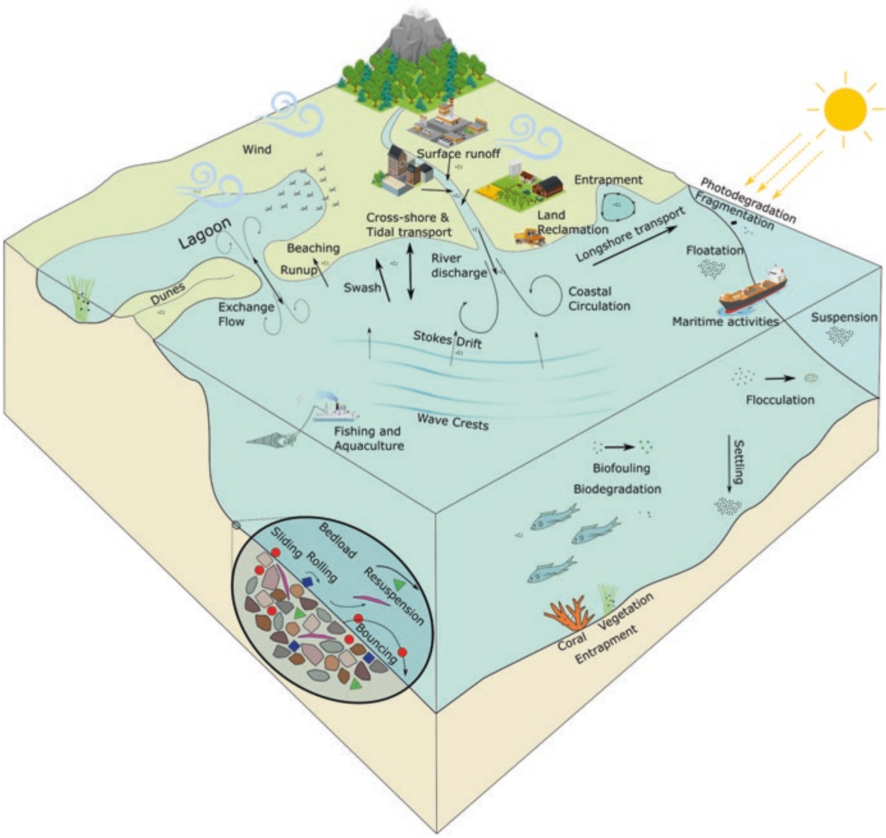

Fig. 12.1 Key Processes controlling the fate and transport of microplastics in coastal environments

transport behavior inside the water column or its settling properties, causing rapid sedimentation into the benthic sediments (Zhang 2017).

Settling of the microplastic particles is determined by the particle settling velocity, wp, and the vertical turbulent dispersion coefficient, KV. Settling velocity of any particle is a function of particle size, shape, roughness, and density and the flow properties represented by Reynolds number ($\mathrm{Re} = \frac{w_p D}{\nu}$). Settling velocity of a spherical particle under the low Re number (<1) flow conditions is defined as shown in Eq. (12.1) based on Stokes' law:

$$w_p = \frac{\Delta g D^2}{18\upsilon} \tag{12.1}$$

where g is the gravitational acceleration, D is the particle diameter, i is the kinematic viscosity of water, Δ is the submerged specific gravity ($\Delta = \frac{\rho_s - \rho_w}{\rho_w}$), and ρ_w and ρ_s are the specific density of water and particle, respectively. For greater Re number (>1) conditions, particle settling is resisted by the turbulent drag, and the settling velocity can be defined as shown in Eq. (12.2):

$$w_p = \sqrt{\frac{4\Delta g D}{3C_D}} \tag{12.2}$$

where C_D is the drag coefficient. For 1 <Re < 1000, C_D can be estimated based on Eq. (12.3):

$$C_D = \frac{24}{\text{Re}} + \frac{3}{\sqrt{\text{Re}}} + 0.34 \tag{12.3}$$

For Re > 1000 $C_D \cong 0.4$–0.5 for smooth spheres and $C_D \cong 1$ for disk shape particles. Considering the variability in the shape, size, and specific density of microplastic particles in the coastal environment, laboratory-based estimation of C_D will enhance the accuracy of the settling velocities.

Biofouling is another mechanism affecting the settling characteristics of microplastic particles. It is the accumulation of organic material on the particle surface, changing the physical properties of the particle (e.g., size, shape, and specific density). Biofouling may cause the settling of buoyant particles. It has been reported to occur within several days and causes the particles to sink within 7 weeks, and the process turns rapidly into defouling due to limited light penetration and temperature near the seabed (Ye and Andrady 1991; Kooi et al. 2017). Kooi et al. (2017) developed the theoretical model for the biofouling over the microplastics based on the settling, biofilm growth, and the ocean depth profiles for light, water density, temperature, salinity, and viscosity.

Settled microplastic particles are subject to benthic transport mechanisms over the seabed. Based on their physical properties, they slide, roll, or bounce over the seabed. Higher-density polymers tend to submerge in the seabed. Bedforms of different sizes, from ripples to dunes, are also expected to influence the benthic transport of microplastics.

12.2.3 Type of Coastal System

Coastal systems are the transition between land and sea. Morphological, sedimentary, and climatological properties of the coastal system determine the flushing and residence time and circulation pattern in the system and eventually determine the impact of the pollutant together with other variables inside the system. Flushing time can be defined as the time required to replace all the water that initially exists in the coastal system. Residence time is when a particle stays and moves inside the coastal system before leaving the system. Local (Hinata et al. 2017), regional (Politikos et al. 2020), and global scale (Liu et al. 2019; Bourgeois et al. 2016; Sharples et al. 2017) studies reported the residence times in different coastal systems such as bays (Meyers and Luther 2008; McEachern et al. 2019), estuaries (Sheldon and Alber 2002), lagoons (Cucco and Umgiesser 2006; Cavalcante et al. 2012), and oceans (Liu et al. 2019).

Microplastics are reported from different coastal ecosystems: mangroves (Deng et al. 2021), tropical coastal ecosystems (Lins-Silva et al. 2021), coastal beaches (Atwood et al. 2019; Gündoğdu and Çevik 2019; Bissen and Chawchai 2020), estuaries (Díaz-Jaramillo et al. 2021; Taha et al. 2021; Zaki et al. 2021), wetlands (Kumar et al. 2021), salt marshes (Weinstein et al. 2020), coral reef (Zhang et al. 2019; Huang et al. 2021), lagoons (Bayo et al. 2019; Chico-Ortiz et al. 2020; Quesadas-Rojas et al. 2021), and tidal flats (Wu et al. 2020).

12.2.4 Particle Properties

Physical properties of the plastic particles play an essential role in the modeling of microplastic transport in the coastal sea environment. The type of plastic is the key factor determining the properties of the plastics. Although many different types of plastics (polymers) have been produced with different properties, the most common types found in the marine environment are shown in Table 12.1.

The buoyancy of the microplastic particle determined by its physical properties determines its transport and fate in the coastal environment. Plastic particles with a specific density greater than the seawater are generally non-buoyant and deposited at the sediment layer over the seabed. Buoyant particles with specific densities less than the seawater are transported at the sea surface or suspended in the water column due to the highly turbulent flow environment in coastal seas. The specific density of widely found plastics in the marine environment varies widely, ranging between 0.8 and 1.4 g/cm^3. Other processes such as flocculation, chemical degradation, and biofouling may change the specific density of plastic particles. These changes are time-dependent and determine their transport over the coastal environment and the fate.

Although the specific density is the most important property in determining the transport and fate of the particles in the coastal environment, their sizes and shapes are also effective. Size distribution of plastics in marine and coastal environments varies widely from nanoplastics to macroplastics. Furthermore, the size of the plastics in the coastal environment varies via the mechanical effects, fragmentation, and degradation as a function of time. In that case, the residence time of plastics in the coastal environment gains importance. Each class of plastic sizes has different

Table 12.1 Common types of polymers found in the marine environment (adapted from (Plastics fact sheet 2009))

Name	Abbreviation	Specific density (g/cm^3)	Examples of debris
Polyethylene terephthalate	PETE, PET	1.30–1.40	Plastic bottles, food jars, ovenable and microwavable food trays, textiles (polyester), monofilament, carpet, and films
Linear low-density polyethylene	LLDPE	0.92–0.95	Bags, stretch wraps, toys, lids, pipes, cables, geomembranes
Low-density polyethylene	LDPE	0.92–0.94	Dispensing bottles, trays, all-purpose containers
High-density polyethylene	HDPE	0.90–0.99	Bottles (beverage, detergent, shampoo), bags, cereal box liners, extruded pipe, and wire and cable covering
Polyvinyl chloride	PVC	1.30–1.70	Packaging (clamshells, shrink wrap), pipes, siding, window frames, fencing, flooring, and medical products (blood bags, tubing)
Low-density polyethylene	LDPE	0.917–0.94	Bags (produce, dry cleaning, newspaper, and garbage bags), squeeze bottles, container lids, shrink wrap, toys, coatings for milk cartons and beverage cups, and wire and cable coverings
Polypropylene	PP	0.85–0.95	Yogurt and other food containers, medicine bottles, straws, bottle caps, fibers, appliances, and carpeting
Extruded and expanded polystyrene	PS	1.04–1.30	CD cases, yogurt containers, cups, plates, bowls, cutlery, hinged takeout containers (clamshells), electronic housings, building insulation, coat hangers, medical products, packing peanuts and other packaging foam, foamed coolers, and egg cartons

impacts on the ecosystem, i.e., micro- and nanoplastics may affect the biota mainly due to easily being ingested by the marine biota that can enter the food web.

Hence the polymer types and their physical properties obtained via the site surveys are important inputs for the numerical modeling studies, which aimed to simulate the transportation and accumulation patterns of the plastics in the coastal environment.

12.2.5 Fate and Transport Models in Coastal Areas

12.2.5.1 Eulerian Transport Models

In the Eulerian transport model, the movement of microplastics is computed overtime on a fixed grid. Transport equation can be written as given in Eq. (12.4) by assuming conservation of mass and incompressible flow:

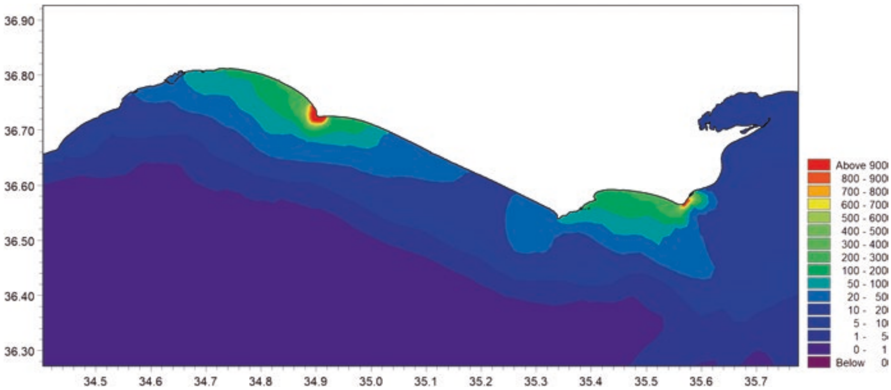

Fig. 12.2 Spatial distribution of microplastic concentrations from Eulerian transport model

$$\frac{\partial C}{\partial t}+u\frac{\partial C}{\partial x}+v\frac{\partial C}{\partial y}+\left(w+w_p\right)\frac{\partial C}{\partial z}=\frac{\partial}{\partial x}\left(K_H\frac{\partial C}{\partial x}\right)+\frac{\partial}{\partial y}\left(K_H\frac{\partial C}{\partial y}\right)+\frac{\partial}{\partial z}\left(K_V\frac{\partial C}{\partial z}\right)+G-L \tag{12.4}$$

where C is the microplastic concentration, t is the time, and u, v, and w are the flow velocities along the axis x, y, and z, respectively. K_H and K_V are the turbulent dispersion coefficients in horizontal and vertical directions, respectively. G and L are the gains and losses from source and sink locations in the model domain if they exist. The second, third, and fourth terms of Eq. (12.4) represent the advective transportation, and fifth, sixth, and seventh terms represent the dispersion. G accounts for the gained matter carried by the sources such as river discharges, wind transport, or ship-originated plastics. L shows the losses by the sinks, such as sedimentation and beaching. Seabed can be considered both as a source and sink via the erosion and deposition processes. Hence the bottom boundary condition ($z = -h$) can be written as seen in Eq. (12.5):

$$K_H\left.\frac{\partial C}{\partial z}\right|_{z=-h}=Q_e-Q_d \tag{12.5}$$

where h is the water depth and Q_e and Q_d are the erosion and deposition rates, respectively. An example of hydrodynamic model results showing the spatial distribution of microplastic particle concentrations within a modeling domain is shown in Fig. 12.2.

12.2.5.2 Lagrangian Transport Models

Lagrangian transport, a.k.a. particle tracking model, considers the plastics as passive particles with specific mass transported by the flow-through advection and dispersion. Lagrange methods usually use decomposed flow velocity into mean and fluctuating components. The mean flow velocity field corresponding to the advective transport is determined by the flow velocity field and interpolated into the particle locations. Next, the particle tracking model estimates the fluctuating component representing the dispersive transport by considering the particle inertia causing a relative motion of particles and the mean particle drift due to gravity. Hence, a particle moves on a three-dimensional domain with directions of x, y, and z as described in Eq. (12.6):

$$\frac{dx_x}{dt} = \overline{u} + u'; \frac{dx_y}{dt} = \overline{v} + v'; \frac{dx_z}{dt} = \overline{w} + w' \tag{12.6}$$

In the z direction, mean flow velocity should be estimated as a summation of the vertical velocity component and the settling velocity of the considered particle for non-buoyant particles ($\overline{w} = w + w_s$). Dispersive term in Eq. (12.6) is estimated by using stochastic approach which uses a random number (r) ranging between −1 and 1, and the amplitude of the motion for the dispersive term is the function of the dispersion coefficient in corresponding direction (K_H or K_V) as shown in Eq. (12.7):

$$u' = v' = 2.45r\sqrt{\frac{K_H}{dt}}; w' = 2.45r\sqrt{\frac{K_V}{dt}} \quad -1 < r < +1 \tag{12.7}$$

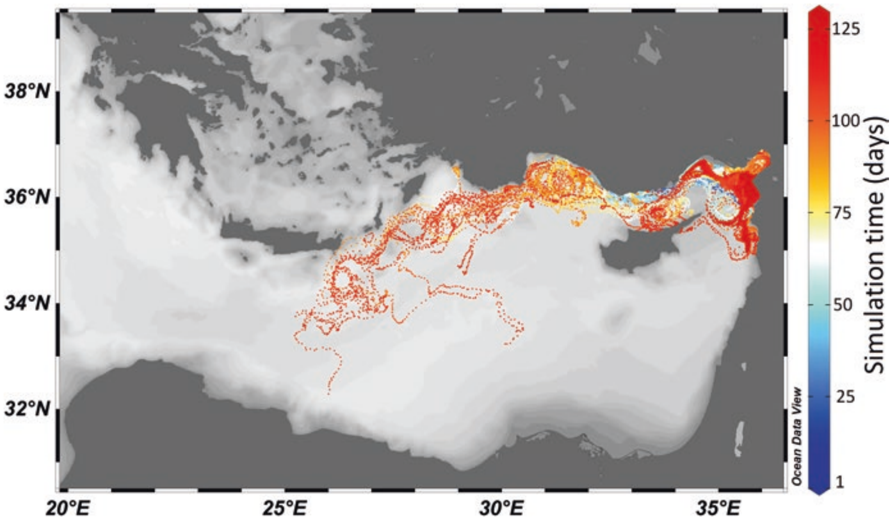

Fig. 12.3 Particle distribution from Lagrangian transport model

In particle tracking simulations, particles released from a known source are traced with the simulation time. An example of particle tracking model results showing that the spatial distribution of buoyant microplastic particle paths within a modeling domain is shown in Fig. 12.3 prepared by using the Ocean Data Viewer (Schlitzer 2020).

12.2.5.3 Source-Pathway-Receptor-Consequence Models

In cases where the information on the spatiotemporal properties of the coastal system and the knowledge of the transport processes is limited, one of the most important holistic approaches in the risk evaluation related to microplastic pollution is Source-Pathway-Receptor-Consequence (SPRC) models. SPRC models have been used in various research areas such as coastal flood risk management (Narayan et al. 2012), climate change adaptation of urban agglomerations (Zhao et al. 2020) to predict concentrations of specific chemicals in soil and shallow groundwater (Mallants et al. 2020), and investigation of heavy metals in urban road dusts (Chenery et al. 2020), since they were introduced in the environmental engineering field by Holdgate (1980). In the microplastic research, SPRC models were applied by Selvam et al. (2021) to evaluate the risk of heavy metal adsorption capacity of polymers in India. Waldschläger et al. (2020) presented a review on the application of SPRC models in the field of microplastic transport through the environment. Possible components of SPRC models and their interaction in coastal areas are shown in Fig. 12.4.

SPRC models describe the sources for the possible pollutant (microplastic in this context), define the pathways in which different mechanisms transport the pollutant from sources to the receptors, define the possible receptors, and evaluate the possible consequences of pollution. There can be different pathways for every single source, and similarly, several pathways may reach the same receptor and vice versa. The pollution concentrations from different sources may differ; some may contribute at higher rates. Also, the weight of the specific pathways in the transport of the pollutant may differ. Similarly, the receptor may be impacted at different levels by any specific pollutant. This makes the description of the quantities and the processes included in each pathway important. This overall holistic approach in the SPRC

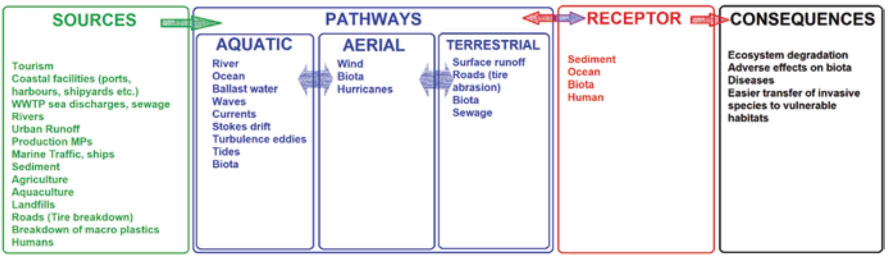

Fig. 12.4 SPRC approach in the microplastic modeling in coastal areas

modeling provides an overall evaluation of the risks at the receptor or the consequences of the pollution. In this way, decision-makers can use these models to prevent pollution at high-risk environments and to conduct prevention measures.

12.2.5.4 Data Requirements and Tools for Numerical Simulations

Due to the complex nature of the coastal area as a transition between land and sea and complicated interactions among the air, water, and sediment processes, including tides, currents, winds, and waves, significant amount of related data is needed to establish a proper numerical simulation. Satellite-based observations, in situ observations, and Argo floats are the primary tools for collecting different scale data globally. These data are used to create long-term continuous reanalysis data based on the specific data assimilation techniques. Widely used land and marine reanalysis datasets are Climate Forecast System Reanalysis (CFSR) provided by the US National Centers for Environmental Prediction (NCEP) of National Oceanic and Atmospheric Administration (NOAA) and ERA-5 provided by the European Centre for Medium-Range Weather Forecasts (ECMWF). The European Marine Observation and Data Network (EMODnet) funded by the European Union provides global coastline and bathymetric data with a grid resolution of 1/16×1/16 arc minutes. Copernicus services also provide a platform where a modeler can reach land, marine, atmosphere, and climate change data from different providers. Site surveys of microplastics in different compartments of coastal environment as well as different coastal systems and establishment of data sharing platforms are expected to enhance our knowledge about the dynamics of microplastic transport in coastal areas and let us further develop better process models. The engagement of volunteers via citizen science is expected to widen the data collection opportunities (Hardesty et al. 2017).

Different types (Eulerian or Lagrangian) of modeling tools for conducting a numerical simulation of microplastic transport in coastal environments can be used for different purposes. Some well-established ocean circulation models are conducted for modeling microplastic transport by different researchers. These modeling tools are listed together with the references of both developers and studies which are used for microplastic simulations in Table 12.2.

Although ocean circulation models listed in Table 12.2 are well-established and validated for modeling the hydrodynamics and simple transport mechanisms, their use in microplastic modeling requires further developments in means of complex processes described in Sect. 12.2.2.

Table 12.2 Currently available modeling tools and their types used in the fate and transport modeling of the microplastics in coastal areas

Name	Type	References	
		Developer	Conducted in
TUFLOW	Three-dimensional finite volume (FV) hydrodynamic model	BMT-WBM (2018)	He et al. (2021)
OceanParcels	Lagrangian particle tracking model	Lange and Van Sebille (2017); Delandmeter and Van Sebille (2019)	Lobelle et al. (2021)
HYbrid Coordinate Ocean Model (HYCOM)	Three-dimensional hydrodynamic model	Lebreton et al. (2012)	Teng et al. (2020)
PELETS-2D	Lagrangian particle tracking model	Callies et al. (2011)	Neumann et al. (2014)
ICHTHYOP	Three-dimensional individual-based Lagrangian tracking model	Lett et al. (2008)	Atwood et al. (2019)
The Second-generation Louvain-la-Neuve Ice-ocean Model (SLIM)	Three-dimensional hydrodynamic and Lagrangian transport model	Lambrechts et al. (2008)	Critchell et al. (2015)
The Regional Ocean Modeling System (ROMS)	Free surface, terrain-following, primitive equation oceanic model	Shchepetkin and McWilliams (2005)	Pereiro et al. (2019); Teng et al. (2020)
BSHcmod	Three-dimensional circulation model	Dick et al. (2001)	Neumann et al. (2014)
ECOM-si	Lagrangian particle tracking model	Blumberg (1994)	Zhang et al. (2020)

12.3 Final Remarks

The complex nature of coastal systems presents challenges for modeling microplastics in these environments. Knowledge is rapidly growing about the behavior of microplastics under different forcing and environmental conditions based on experimental studies and observations from different coastal systems globally. This allows numerical modelers to further develop their tools by validating their existent numerical tools and incorporating new transport processes. Although many research gaps remain, our understanding of dynamics and interactions between processes and the complex coastal environment based on-site surveys improves our numerical models. There is a need for data especially from the harsh environment such as benthos to develop better numerical tools. Better numerical tools would help the decision-makers implement robust management plans for the plastic solution.

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Chapter 13
Occurrence of Microplastic Pollution in Marine Water

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Abstract Plastic pollution has escalated during last 50 years. The estimated value of plastic content in marine environment is more than 250,000 tons. Microplastic (less than 500 mm in diameter) accumulation in seas and oceans is the new potential environmental concern because of its rapidly increasing concentration in marine water, sediments, and marine animals. These are introduced into the environment either through primary sources, which include plastic pellets, microbeads, and glitters, or secondary sources which are microplastic dust, water treatment plants, wear and tear from normal use, and large objects that produce secondary microplastic upon deterioration, etc. Particles as small as 0.01 mm to <5 mm have been found in oceans worldwide. They exist in different shapes like fiber, film fragment, granules, and spherules and consist of different polymers like polyethylene, polypropylene, polystyrene, low- and high-density polyethylene, thermoplastic polyurethane (TPR), nylon (NYL), polyvinyl chloride (PVC), ethylene propylene rubber (EPR), acrylonitrile, and styrene. Concentration of microplastic can be expressed in the number of particles per cm^2, m^2, m^3, or km^2. Maximum concentration of microplastic in terms of particles per meter cube is 15,560 recorded in Southeastern Sea,

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Korea, whereas, in terms of the number of items per km^2, it is 360,000 items at North Atlantic Ocean (subtropical gyre). Bohai Sea and Suva Harbour of Urban Coast of Fiji appear to be least contaminated with microplastic. Marine animals like mysids, mollusks, and fishes are ingesting microplastic and dying as a result of its accumulation in their stomach. MP particles also reach higher trophic levels through the process of biomagnification. Keeping these things in view, scientists either suggest imposing ban on the production of certain plastic type or formation of rules and regulation on the production, usage, and throwing off plastic items in order to control the ever-increasing levels of microplastic pollution in marine water.

Keywords Occurrence · Marine water · Microplastic · Pollution

13.1 Introduction

It is estimated that the world's oceans contain more than 250,000 tons of plastic due to anthropogenic activities in the last 50 years (Eriksen et al. 2014; Thompson et al. 2004a, b). This suggests that out of all anthropogenic debris in seas and oceans, about 86% consists of plastic (Laist 1987; Barnes et al. 2009; Ivar Do Sul and Costa 2014; Jambeck et al. 2015; Nelms et al. 2016). Plastic is one of the biggest threats to marine life since it is abundant with high mobility. It can develop interactions with all parts of the oceanic life through a number of different pathways (Derraik 2002; Cole et al. 2013; Gall and Thompson 2015; Nelms et al. 2016). This high dispersion of plastic all over the marine environment is responsible for the ingestion and entanglement of plastic by the aquatic organisms that may cause severe injury and ultimately death (Derraik 2002; Gall and Thompson 2015; Nelms et al. 2016; Duncan et al. 2017).

Plastics are made synthetically using a number of chemicals such as petrochemicals (obtained from oil, natural gas, or petroleum) that improve their performance (Costa et al. 2016). They are basically polymers which are derived from either organic or inorganic raw material and consist of hydrogen, carbon, oxygen, silicon, and chloride (Shah et al. 2007). High- and low-density polyethylene, polypropylene, polystyrene (PS), polyvinyl chloride (PVC), and polyethylene terephthalate (PET) are most widely used plastics these days and constitute 90% weight of the total plastic mass produced by the world (Andrady and Neal 2009). Plastics have some brilliant properties like temperature, light and chemical and water resistance (Andrady and Neal 2009; Cauwenberghe et al. 2015), good strength, and low manufacturing cost – all those that make them very suitable for use in a variety of applications. Anyhow, the real problem is that this very beneficial durability of plastic makes it extremely difficult to degrade (Sivan 2011). Plastic waste, like other common waste, is usually dumped into the landfills where it takes many years to decompose (Cole et al. 2011a, b), or it ultimately reaches to seas and oceans through different channels where it causes harmful impacts on the aquatic life and affects the marine contribution of primary productivity on a global scale

(Gregory 2009). Occurrence of plastic in marine ecosystems is attributed to low awareness to the environmental health, reduced recycling, and basic sanitation deficiencies throughout the time plastic is used and disposed (Li et al. 2016). The presence of plastic in natural systems is not only affecting ecological and socioeconomic factors but is also posing serious threats to human health (Antão-Barboza et al. 2018).

13.2 Microplastic

Plastics that have diameter less than 5 mm are commonly known as microplastic (MP) (Arthur et al. 2009; Moore 2008). Their presence in the environment was first reported in the 1970s (Carpenter and Smith 1972). Since then a number of organizations have conducted research and found that microplastics are abundantly present in the marine ecosystems and they are adversely impacting marine biota (Rands et al. 2010; Sutherland et al. 2010). After reaching the marine habitat, microplastics can show two types of behavior. Particles which are lighter than the water are likely to float on the surface, and they will spread all over the water. The amount of these floating microplastics varies from 93 to 268 kilotons (Eriksen et al. 2014; Setälä et al. 2014). Particles which are denser than water will settle below and accumulate at the bottom. Microplastic pollution has become a matter of immediate concern also because these tiny fragments are present in a wide variety of color, weight, form, and size and can easily absorb into the environment as well as into the bodies of organisms like crustaceans, fishes, mollusks, etc. causing physical damage or even mortality (Wright et al. 2013a, b; Antão-Barboza et al. 2018; Derraik 2002; Cole et al. 2011a, b; Andrady 2011; Ivar Do Sul and Costa 2014; Wang et al. 2016; Sun et al. 2017). Moreover, they can flow in the food web through bioaccumulation and biomagnification ultimately reaching higher-level organisms including human beings (Ríos et al. 2007; Andrady 2011; Kühn et al. 2015; Bennecke et al. 2016; Wang et al. 2016; Massos and Turner 2017).

Keeping in view the increasing levels of microplastic accumulation in the marine environment, the International Atomic Energy Agency has given projections of this increase in the Eastern Tropical Pacific Ocean till 2100, and the numbers are horrifying (IAEA 2020) (Fig. 13.1).

13.3 Types of Microplastic

Microplastic is normally categorized into two groups, i.e., primary and secondary microplastics.

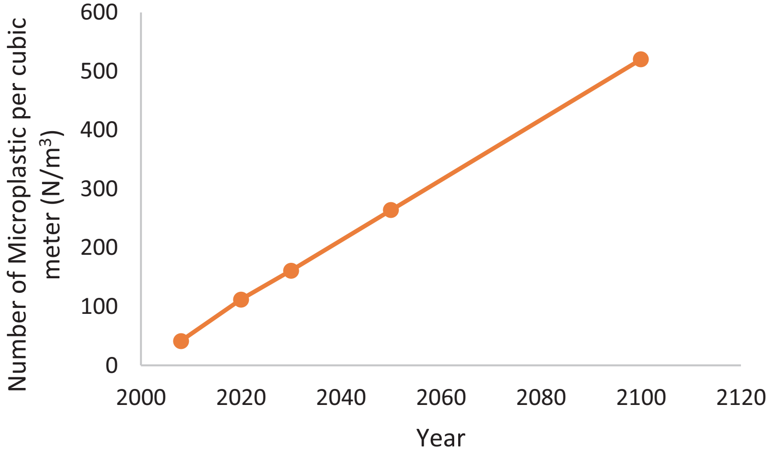

Fig. 13.1 Abundance of microplastic pollution in the Eastern Tropical Pacific Ocean (IAEA 2020)

13.3.1 Primary Microplastic

Microplastic is considered as primary microplastic if it is directly released into the environment. Primary microplastic is considered to constitute 15–31% of total microplastic of the marine habitat. It is released along with wastewater from laundry, vehicles' tire abrasion, and personal care products, like microbeads in the facial or hand cleansers. These sources contribute 35, 28, and 2% of total primary microplastic production, respectively (Parliament, European 2018). These fragments are also introduced into the environment when intermediate plastic feedstock or by-products are released from industry (GESAMP 2015). These tiny particles of microplastic are lipophilic which means they can quickly take up toxic substances, like dichlorodiphenyltrichloroethane (DDT) polycyclic aromatic hydrocarbons (PAHs) and polychlorinated biphenyls (PCBs). The presence of such substances on primary microplastic has been detected by the experiments (Cauwenberghe et al. 2015). It is estimated the 0.8–2.5 million tons of microplastic is globally released into the oceans every year. Ninety-eight percent of the total primary microplastic mass is released form land-based activities. Activities done on seas and oceans contribute to only 2%.

13.3.2 Secondary Microplastic

It is derived from the degradation of larger plastic items like plastic bottle, bag, and any other plastic object. It consists of 69–81% of microplastic in the marine habitat (Parliament, European 2018). Secondary microplastic is generated when plastic items are subject to breakdown through radiation exposure wind pressure or wave action (Rogers 2019). Radiation exposure is also known as photodegradation in which strong ultraviolet radiation from sun cleaves bonds between plastic particles (Barnes et al. 2009), whereas breakdown through wind or wave action is usually referred to as weathering (Arthur et al. 2009).

13.3.3 Nanoplastic

Microplastic is referred to as nanoplastic when as least two of its dimensions have a size less than 100 nm. It is also produced by the breakdown of larger plastic particles, e.g., when fragmentation occurs during the washing of clothes or when weathering occurs during an abrasion system (Costa et al. 2016). These tiny particles of plastic are very easily ingested by the aquatic organisms like phytoplanktons, zooplanktons, and corals, and this is where they make their way to the food chain. Like other microplastics, they also have the capability to adsorb persistent organic pollutants onto their surface and hence increase their harmful impact (Sharma and Chatterjee 2017).

13.4 Sources

Microplastic from different sources makes its way to the marine environment through four main pathways. If it is thrown on land, it will either flow along with water runoff or move with wind currents and reach seas and oceans. If it is present in wastewater, as a result of poor sanitation system and release of wastewater into natural bodies, it will ultimately reach into a river and finally in the ocean. The fourth way is the people throwing plastic objects and particles directly into the marine water because of lack of awareness (Fig. 13.2).

13.4.1 Primary Sources

Plastic *pellets* are a primary form of microplastic, and their diameter normally ranges from 2 to 5 mm. They are sent to the plastic transformers where they are used in making a number of plastic products. They can be released from different incidents, e.g., spilling, during the processes of manufacturing, recycling, and transporting (Essel et al. 2015). They are also known by other names, like as nurdles, nibs, or mermaid tears (Sundt et al. 2014).

Microbeads are used in personal care products, rinse-off cosmetics, and cleansing products. Estimation of microbeads in Hong Kong coastal water revealed that particles present in that region had a diameter of 0.332–1.015 mm. Life cycle assessment and FTIR spectroscopy revealed that the particles were similar to those

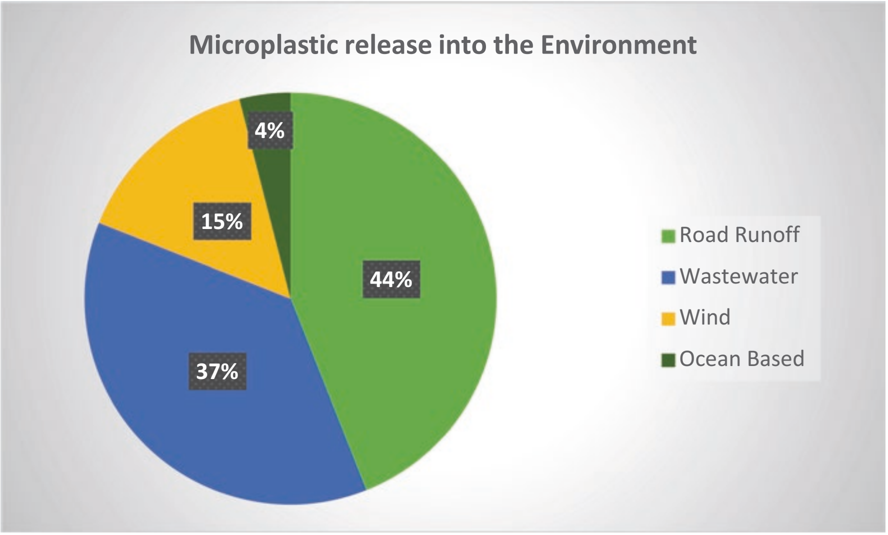

Fig. 13.2 Microplastic release into marine. Sources of microplastic release into the environment are explained below

found in a facial scrub from a local market in terms of color, shapes, and sizes (Cheung and Fok 2016).

Glitters are shiny substances which are made of polyethylene terephthalate (PET) polymer, acrylic, polyvinyl chloride (PVC), and/or polymethyl methacrylate (PMMA). These are commonly used in cosmetic and textile products, pre-school classroom settings, and different household applications and are available in a number of uniform shapes and sizes. These tiny particles after use find their way to the water streams along with the wastewater and finally reach seas and oceans (Yurtsever 2019).

13.4.2 Secondary Sources

13.4.2.1 Microplastic Dust

This includes release of plastic in the form of dust when different plastic products are cut or molded. This occurs at the plastic manufacturing industry as well as at repair shops/workshops. Plastic dust can also be released from the application of plastic paint and/or construction work that includes working with painted surfaces. Microplastic dust from burning activities can also be carried to the seas and oceans through wind currents and rainfall (Sundt et al. 2014).

13.4.2.2 Water Treatment Plants

Microplastic has been detected primary and secondary stages of water treatment. A number of these microplastic particles are capable of passing through filters being used in wastewater treatment plants (Cole et al. 2011a, b). About one particle per 1 liter of MP has been released back into the environment (Carr et al. 2016; Estahbanati and Fahrenfeld 2016; Mintenig et al. 2017).

13.4.2.3 Wear and Tear from Normal Use

Plastic fibers and other particles from laundry are washed from along with wastewater into water streams that ultimately reach marine environment. It is estimated that one garment in the laundry can release more than 1900 microplastic particles, and out of all, fleece releases the highest quantity of fibers (Katsnelson 2015; Grossman 2015). Wear and tear from aquaculture effluent (fishing net, ropes, etc.) and plastic particles from weathering household items like water pipelines also add to the microplastic accumulation into marine environment (Sundt et al. 2014).

13.4.2.4 Secondary Microplastic

Plastic waste is being dumped into the seas and oceans. Degradation and decomposition of these waste items from industrial waste and common waste (plastic bottles and other packaging materials like wrappers, plastic wraps, etc.) have potentially increased microplastic levels in the marine ecosystems (Mason et al. 2018; Carrington 2020). In the era of COVID-19 pandemic, face masks have been recognized as the new potential source of microplastic pollution. Face masks are usually made of polypropylene, polystyrene, polyacrylonitrile, and polyurethane. A number of these disposable face masks reach into the marine habitat through water streams where they break down into smaller particles (Fadare and Okoffo 2020).

All of the abovementioned sources are the releasing microplastic either into the atmosphere or water channels (streams, rivers, etc.) through which it is ultimately making its way to the seas and oceans (Fig. 13.3).

13.5 Occurrence

Research has been conducted worldwide in order to determine the concentration of microplastic in different oceans. This includes research in different regions like beaches, surface waters, and deep sea waters. China is considered among three top producers of plastic waste. Research has been carried out in order to determine the contamination of this plastic waste into the marine waters. Determination of suspended microplastic abundance in East China Sea and Yangtze Estuary, China, has

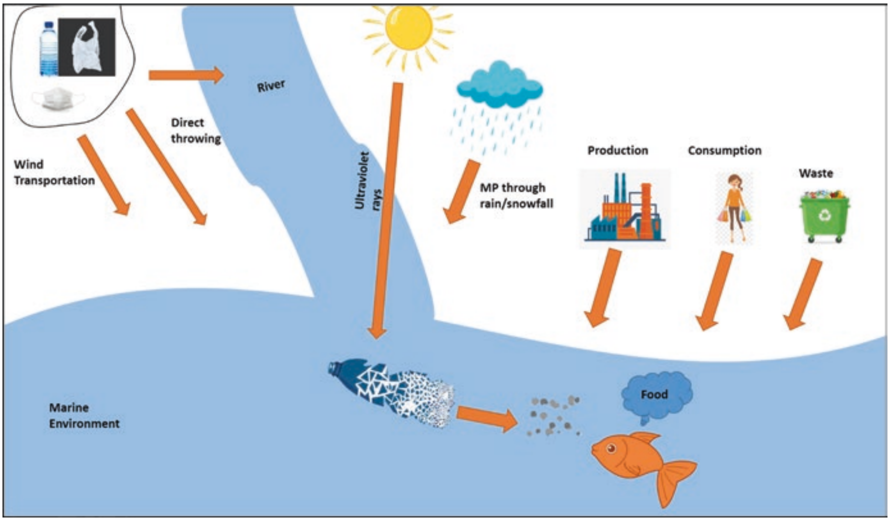

Fig. 13.3 Sources of microplastic (Eriksen et al. 2018; Mallavarapu 2021)

revealed that a maximum amount of particles were being carried out by the rivers into the sea. Particles were enumerated using a dissecting microscope. Different shapes of microplastic found in Yangtze Estuary include fibers (79.1%) being the most common shape, followed by granules (11.6%) and films (9.1%). In case of East China Sea (ECS), fibers constituted 83.2% of the total MP, followed by granules constituting 14.7%, films 2.1%, and spherules 0%. Spherules were the least common shape found in this study by Zhao et al. (2014). They suggest that this smaller amount can be attributed to the industrial initiatives being successful in reducing pellets from the environment.

Similarly, concentration of microplastic in coastal waters of Hong Kong has been determined. The abundance is represented as the amount of microplastic per 100 m^3 of water, and it has come up to be 32-1187 (particles per m^3) from June to July 2015, 46-1635 for November 2015, and 131-35,642 for March 2016. Particle shapes were line and line-like (0.2%), pellet and pellet-like (96.8%), fiber and fiber-like (0.4%), and fragment and fragment-like (2.7%) (Tsang et al. 2017). In the coastal waters of Japan, the amount of microbeads (smaller than 0.8 mm in diameter) was found to be 9.7% in comparison with the quantity of the total microplastic (Isobe 2016).

Research carried by on different estuaries in China suggests that most abundant shape of MP was fiber (83%) in these waters. In contrast, Bohai sea water contains only 3% fiber. This contrast is explained by the unequal distribution of different MP shapes in different sampling sites. Study conducted on Taiwan's Northern Coast has also revealed the presence of microplastic in different shapes. Kunz et al. (2016) have classified these shapes into different groups, viz., granular particles, elongated particles, and flat particles. Granular particles have resemblance with grains and are predominant in smaller-sized MP. Elongated particles are normally fragments that give appearance like chips. Flat particles are further divided into thick (can't be bent) and thin (can be bent easily) particles. Most common particles in the coastal water were fragments and thin shaped. Microbeads were only found in samples from Shalun and Waimushan beach. Plastic pellets were also discovered in these two beaches. Fibers were present in all samples. They were mainly white but occurred in different colors as well.

Water samples collected using trawls from Southeastern Sea, Korea, were analyzed to find that paint particles were the most abundant MP in them. Proportion of paint particles was 48.99%. Percentage of styrofoam was less than paint particles, i.e., 19.64%, whereas fibers were least of them, comprising 17.48% of the total MP particles. These figures refer to the sampling done in the month of May, while in July, concentration of styrofoam increased significantly. It became 51.74%, with paint particles remaining 19.16%. Particles of hard plastic comprised 17.8% of the total MP collected (Kang et al. 2015).

Experimentation on Southeastern Black Sea by Turkish researchers (Aytan et al. 2016) has revealed that Black Sea has become the hotspot for MP pollution. The concentration (600–1200 pieces per cubic meter) suggests that there is an urgent need of understanding the origins of these continuously accumulating particles, their modes of transportation, and their effect on the animals residing this polluted habitat. Primary shapes of MP particles from Black Sea were fragment, films, and fibers with a percentage value of 20%, 30.6%, and 49.4%, respectively.

Research done on North Sea and Celtic Sea however indicated relatively low abundance of microplastic in these surface waters (Table 13.1). Higher concentration was observed in some coastal and estuarine region, and few microplastics were found in the areas far away from potential land sources of microplastic. Scientists suggest that this small amount could be a result of atmospheric deposition of microplastic into the surface water (Maes et al. 2017; Dris et al. 2016). Different shapes of microplastic were found in these tow seas were line (5%), foam (8%), pellets (10%), film (14%), and fragments (63%) (Maes et al. 2017). In an experiment performed on North Atlantic Ocean in the region of Rockall Trough (Scotland, UK), water sample was taken from the depth of 2200 m in order to analyze the occurrence and properties of microplastic in that region. ATR-FTIR analysis identified 78 microplastics in 240 L of the sampled deep sea water. Out of these, 28 were identified as cellulose, 17 as synthetic microplastics, and others remained uncategorized because of unclear spectra (Courtene-Jones et al. 2017).

Desforges et al. (2014) came up with the conclusion that Northeastern Pacific Ocean had MP pollution 16 times greater than the North Pacific. The amount of microplastic content in its water was 27 times greater than in other oceans of the world, but recently with the figures being changed, Southeastern Sea, Korea, and subtropical gyre of North Atlantic Ocean appear to have the highest concentration of microplastic particles.

A number of studies have demonstrated that the presence of primary microplastic in marine environment is harmful for the aquatic organisms. Different vertebrates and invertebrates like bivalves, mussels (Van Cauwenberghe et al. 2013, 2015), crustaceans (Murray and Cowie 2011; Setälä et al. 2014), birds (Zhao et al. 2016; Holland et al. 2016), fish (Lusher et al. 2013; Neves et al. 2015), and even

Table 13.1 Abundance of microplastic in marine water

Region	Concentration (items per given area)	Size	Type of polymer	References
Southeastern Sea, Korea (Nakdong River Estuary)	210–15,560/m^3	<2 mm	Polyester, polyethylene, alkyd, expanded polystyrene, polypropylene, polyethylene + polypropylene	Kang et al. (2015)
Jiaojiang Estuary, China	955.6/m^3	<0.5–5 mm	Polypropylene, polyethylene, polytetrafluoroethylene, PVC	
Oujiang Estuary, China	680/m^3	<0.5–5 mm	Polypropylene, polyethylene, polytetrafluoroethylene, PVC	
Minjiang Estuary, China	1245/m^3	<0.5–5 mm	Polypropylene, polytetrafluoroethylene, polyvinyl chloride, polyethylene	
East Asian Seas, Japan	0.03–491/m^3	0.3–5 mm	NA	Isobe (2016)
Bohai Sea, China	0.01–1.23/m^3	0.3–5 mm	Polyethylene terephthalate (3%), polystyrene (16%), polypropylene (29%), polyethylene (51%)	Zhang et al. (2017)
Northern Coast of Taiwan	4–532/0.0152 m^3	0.25–4 mm	ABS (1%), polystyrene (12%), polypropylene (43%), polyethylene (44%)	Kunz et al. (2016)
Bay of Bengal, India	A few hundreds–20,000/km^2 (100,000 max.)	0.339–5 mm	NA	Eriksen et al. (2018)
Coastal Water, Hong Kong	51–27,909/100 m^3	(0.03–4.96) mm	Polypropylene (50.9%), low-density polyethylene (18.2%), high-density polyethylene (26.4%), styrene Acrylonitrile (0.9%), and polypropylene + ethylene propylene (3.6%)	Tsang et al. (2017)
Southeastern Black Sea	600–1200/m^3	0.2–5 mm	NA	Aytan et al. (2016)
Iskenderun Bay, Turkey	0.2254/m^2	2.7 mm (mean)	NA	Gündoğdu (2017)
Mersin Bay, Turkey	0.6827/m^2	3.01 mm (mean)	NA	Gündoğdu (2017)
Mediterranean Sea, Bay of Calvi	0.062/m^2	2–5 mm	NA	Collignon et al. (2014)

(continued)

Table 13.1 (continued)

Region	Concentration (items per given area)	Size	Type of polymer	References
North Sea, Celtic Sea, UK Channel	0–1.5/m^3	(1.0–2.79) mm	NA	Maes et al. (2017)
Rockall Trough, North Atlantic Ocean	70.8/m^3	0.4–5 mm	Acrylic (6%), polyester (65%), acrylic and cellulose (6%), polyethylene (6%), polyethylene terephthalate (17%)	Courtene-Jones et al. (2017)
Northeastern Pacific Ocean	279 ± 178/m^3	684 ± 953 μm	NA	Desforges et al. (2014)
West Coast Vancouver Island	1710 ± 1110/m^3	558 ± 521 μm	NA	Desforges et al. (2014)
Queen Charlotte Sound, New Zealand	7630 ± 1410/m^3	398 ± 376 μm	NA	Desforges et al. (2014)
Strait of Georgia	3210 ± 628/m^3	513 ± 494 μm	NA	Desforges et al. (2014)
South Pacific Ocean	Around 20,000/km^2 (>50,000 max.)	0.339–4.75 mm	NA	Eriksen et al. (2018)
East Greenland	2.38/m^3	0.77 mm (median)	Low- and high-density polyethylene (23%), polypropylene (10%), polyester (53%), nylon (3.3%), cellulose acetate (6.7%), polyvinyl chloride (3.3%)	Amélineau et al. (2016)
Beach water, South Africa	257.9 ± 53.36–1215 ± 276.7/m^3	0.080–5 mm	Polystyrene	Nel and Froneman (2015)
Ross Sea, Antarctica (Terra Nova Bay)	5–1705/m^2	(0.3–22)mm	Nylon, styrene-butadiene-styrene (SBS), polystyrene, thermoplastic polyurethane (TPR), ethylene propylene rubber (EPR), polyvinyl alcohol (PVA), polyvinyl chloride, polypropylene, polyethylene	Munari et al. (2017)
North Atlantic Ocean (subtropical gyre)	5000–360,000/km^2	≤ 300 μm	NA	Brach et al. (2018)

(continued)

Table 13.1 (continued)

Region	Concentration (items per given area)	Size	Type of polymer	References
Arctic Sea Ice	38–234/m^3	≤ 2 mm	Rayon (54%), polyester (21%), polyamide (nylon) (16%), polypropylene (3%), polyethylene + acrylic + polystyrene (2%)	Obbard et al. (2014)
Beibu Gulf	399–5531/m^3	500–1000 μm	Polypropylene, polyethylene	Li et al. (2020)
Urban Coast, Fiji • Vanua Navakavu. • Laucala Bay. • Suva Harbour	0.24 ± 0.07/m^3 0.09 ± 0.02/m^3 0.10 ± 0.02/m^3	≤125 μm	Polystyrene + latex + nitrile (17%), acrylonitrile butadiene styrene, polycarbonate, polyethylene, acrylic, polypropylene, polystyrene, polyvinyl chloride (8.3% each)	Ferreira et al. (2020)

marine mammals (Eriksson et al. 2013) can ingest these tiny particles and experience their adverse health effects like inflammation (Lu et al. 2016), teratogenicity (Nobre et al. 2015), low feeding rate (Bergami et al. 2016), and decreased energy storage (Wright et al. 2013a, b). It has also been revealed that *Ophiomusium lymani* had 1.153, *Hymenaster pellucidus* 1.582, and *Colus jeffreysianus* had 0.678 average microplastic pieces per 1 gram of the wet tissue weight. Particles consisted of polyester (17%), alkyd resin (50%), and acrylic and cellulose (33%) (Courtene-Jones et al. 2017). It is obvious that the sampling point was small and the results cannot be implied to the large scale.

Ingestion of microplastic by benthic holothurians has been studied. Four species, viz., *Holothuria floridana*, *Cucumaria frondosa*, *Thyonella gemmata*, and *H. grisea*, were found to have ingested (0.25–15 mm) more amount of MP than expected. The estimated number of fibers was up to 517 per individual (Graham and Thompson 2009). Ingestion of small fragments has also been observed in *Nephrops norvegicus*. Gut analysis showed that considerable population (83%) of these animals from Clyde Sea had MP in their bodies. Similarly, 100% of the seeded fish population had ingested nylon fibers (up to 5 mm) (Murray and Cowie 2011). In a laboratory study, it was demonstrated that ciliates in the marine environment mistake plastic microspheres (0.75 μm) with food particles. In another study, sea star, sea urchin, sea cucumber, brittle star, and sand dollar were observed to capture and ingest microspheres (10–20 μm) made of polystyrene divinylbenzene.

Researchers analyzed scat from captive grey seal and digestive tract of the wild fish (that seals used to eat) in order to understand the flow of microplastic in the food chain through biomagnification. The presence of microplastic particles was confirmed in both fish and scat samples. Researchers suggest that microplastic particles have been ingested by the fish prey which were eaten by the fish. Fish when

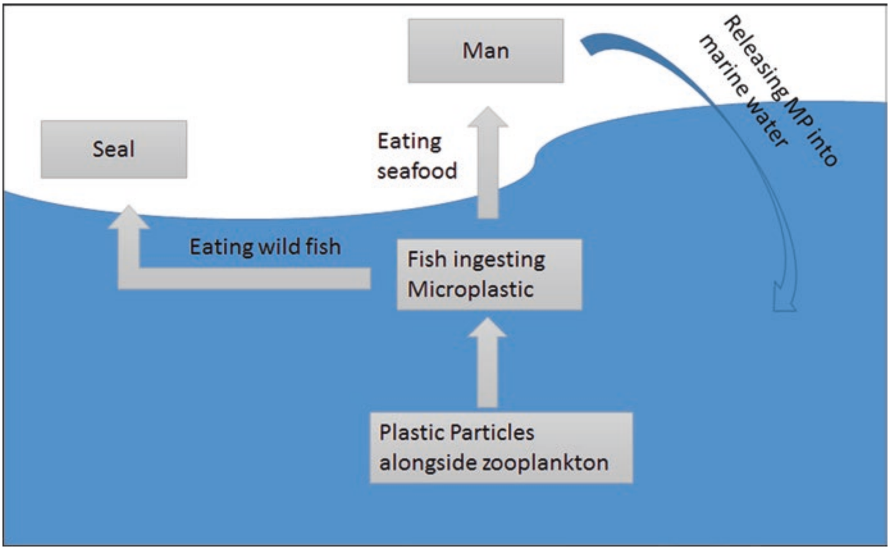

Fig. 13.4 Flow of MP in food web through biomagnification

eaten by the seal transferred these particles into the seals' bodies (see Fig. 13.4) (Nelms et al. 2018). Microplastic can also make its way to the bodies of human beings when they eat sea food, for example, shell fish that is found to have microplastic in its body (Murray and Cowie 2011; Nelms et al. 2018).

It is strongly suggested that we should ban the use of glitters since their concentration in the marine environment is increasing day by day. They are ingested by fish, amphibians, and other animals, and after being ingested, they are collected into their stomach and other organs. Accumulation of this microplastic into the stomach leads to the death of the animal because of starvation (Wildlife Artists).

13.6 Proposed Solution

As a universal solution, control at source is proposed as the best solution for reducing microplastic too from the environment and specifically from marine habitat. It integrates rethink and refuse; rethink implies for considering other options, and refuse suggests reduction of induction of single-use items into the environment. There is a need of further research for seeking alternatives to plastic products. Public community should also be made aware of the harmful outcomes of plastic usage and switching towards the green alternatives (Ivar Do Sul and Costa 2014).

Some researcher suggests that there should be a ban against the use of glitter in personal care products, while others say that a ban could be premature and that regulations should be made and enforced. However, different countries have worked in order to reduce glitter from the environment. For example, a cosmetic retailer in the UK has replaced plastic glitter made of PET with mineral and mica glitter.

The USA has imposed ban on the production of personal care products containing microbeads in 2016. Selling of cosmetic products and drugs that contain microbeads or any other plastic type has also been banned in 2018 and 2019, respectively. Canada has also banned the use of microbeads in 2016, while New Zealand has banned the use of glitters in pre-school (Parker 2017).

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Part III
Risk Assessment and Health Impact

Chapter 14
Microplastic Pollution and Contamination of Seafood (Including Fish, Sharks, Mussels, Oysters, Shrimps and Seaweeds): A Global Overview

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Abstract This chapter collected, collated, analysed, synthesised, interpreted and documented the last 15 years (2006–2021) of research investigations carried out on microplastic (MP) pollution impacts on seafood organisms including fish, sharks, oysters, mussels, shrimps, lobsters and seaweeds covering 36 locations or countries in the world (the Atlantic Ocean, Australia, the Baltic Sea, Bangladesh, Belgium, Brazil, Canada, Chile, China, Fiji, France, the Gulf of Mexico, India, Indonesia, Iran, Italy, Japan, Malaysia, the Mediterranean Sea, the Netherlands, North Pacific Central Gyre, North Pacific Subtropical Gyre, North Sea, Norway, Portugal, Saudi Arabia, Scotland, South Pacific Subtropical Gyre, Spain, Tanzania, Thailand, Turkey, the UK, the USA and Vanuatu). Elevated/high levels of MP ingestions (compared to other species investigated by researchers) were found in 47 seafood species (39 fish, 1 shark, 3 molluscs, 3 crustaceans and 1 seaweed). MP particles ingested by seafood organisms were highly variable and found related to feeding habits and habitats of the species. MP ingestion rate in seafood organisms varied between 3% and 100%. Higher ingestion (>30%) was reported from the Atlantic Ocean (fish), Australia (fish), Belgium (shrimp), Brazil (fish), Chile (fish), China (fish), China (seaweed), Fiji (fish), France (fish), India (fish), Italy (shark), Japan (fish), Malaysia (fish), North Pacific Central Gyre (fish), Portugal (fish), Scotland

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(lobster), South Pacific Subtropical Gyre (fish), Spain (fish), Thailand (fish), Turkey (fish), the UK (fish), the UK (shark), the USA (oyster) and Vanuatu (fish). Fibres were the major polymer (by shapes) ingested by seafood organisms (ingestion rate ranged from 33% to 99%). Black, blue, green, orange, purple, red and white were the coloured polymer ingested by various seafood organisms. The higher MP ingestion in seafood organisms may have occurred due to a number of reasons including the following: (1) the study might have been carried out in MP pollution 'hot spot' areas, (2) fish and other organisms that accidentally/mistakenly ingested MPs during their normal feeding activity (confusing MPs as prey/plankton/food) or (3) MP ingestion has occurred through trophic transfer from their prey species (as fish foods such as amphipods, copepods, decapods and euphausiids larvae are known to ingest microplastics). Polymers ingested by seafood organisms can adsorb persistent organic pollutants/priority pollutants (heavy metals, PAHs, DDT, PCBs). In addition, plastic additive chemicals (phthalates, bisphenol A, heavy metals, flame retardants) can leach out to the aquatic environment or ingested biota. Therefore, both adsorbed and additive chemicals may be transferred to humans via the consumption of contaminated seafood (fresh fish, whole fish, canned fish and dry fish). The possible human health effects of consuming MP-contaminated food and water include damage of both DNA and cells and inflammation reaction.

Keywords Microplastics · Seafood · Fish · Sharks · Oysters · Mussels · Shrimp · Lobsters · Seaweeds · Global contamination · Priority pollutants · Human health

14.1 Introduction

The common types of plastic are polyethylene (PE), polypropylene (PP), polystyrene (PS), polyethylene terephthalate (PET), polyamide (PA), polyester (PES) and polyvinyl chloride (PVC) (Kibria 2018; Kibria et al. 2021). Microplastics (MPs) are small plastic particles which are less than five (<5) millimetres (mm) in size (https://oceanservice.noaa.gov/facts/microplastics.html). Primary MPs include microbeads found in cosmetics, personal care products (soap, toothpaste and facial scrub), nail polish, lipsticks, microfibres (used in textiles), virgin pellets and cleaning products (Potter 2017; Germanov et al. 2018). Microbeads are commonly made from PE, PP, PET and PA. Secondary microplastics (resulting from environmental degradation of larger plastic particles/items as a result of weathering, wave action, wind abrasion, biodegradation and ultraviolet photo-degradation) include synthetic fibres (fishing gears, textiles), fragments (plastic bags, bottles, car tyres) and films (Kibria et al. 2021).

Low-density plastics are PP (density 0.925–0.959 g/cm^3) and PE (density 0.905 g/cm^3) which are dominant in the top/surface layers of the ocean (*note: the densities of freshwater and seawater are 1.0 g/mL and 1.025 g/mL, respectively*). High-density plastics are PVC (density 1.384 g/cm^3), PES (density 1.04–1.46 g/

cm^3) or PA (density 1.12–1.16 g/cm^3) which usually sink and accumulate to the sea bottom (Andrady 2011; Lusher et al. 2013; Cózar et al. 2015; https://amesweb.info/Materials/Density-of-Plastics.aspx). Thus, both low- and high-density MPs can be ingested inadvertently/accidentally through feeding by both pelagic (those live and feed at surface layers of the ocean) and demersal species (those live and feed near the bottom of the ocean) such as fish, sharks, oysters, mussels and prawns.

MPs can be directly ingested (called primary ingestion) by pelagic species (fish) and demersal species (fish, prawns) 'mistakenly or confusing' it as 'preys or food' while searching for food. The deep-sea sediment is a major sink for plastic debris. MPs can also be accidentally ingested by filter-feeding organisms such as mussels and oysters during their normal filter feeding. Due to the small particle size, MPs are reported to have been ingested by various seafood organisms including fish (Rochman et al. 2015; Wootton et al. 2021), sharks (Parton et al. 2020), mussels (De Witte et al. 2014), oysters (Ribeiro et al. 2020) and prawns and shrimps (Devriese et al. 2015; Ribeiro et al. 2020). Therefore, there is a likelihood of human exposure to MPs (as well as MPs that adsorbed chemical pollutants) via eating contaminated seafoods. Among the environmental pollutants in aquatic ecosystems, MPs are now recognised as emerging pollutants of a great concern and are considered a priority research topic (Barboza and Gimenez 2015). The objectives of this chapter are to:

- Collect, collate, analyse, synthesise, interpret and document MP pollution and contamination of seafood including fish, sharks, mussels, oysters, clams, squids, prawns, shrimps, lobsters and seaweeds across the globe.
- Document the impacts of MP pollution in the context of MP ingestion and the polymer shapes, types and colour ingested by seafood organisms.
- Assess the relationships between the MP ingestion and feeding habits and habitats of the seafood organisms.
- Assess/evaluate the possible human health effects from contaminations of seafood with MPs.

14.2 Microplastics (MPs) in Seafood (Fish, Sharks, Mussels, Oysters, Clams, Prawns, Shrimps and Seaweeds): A Global Overview

This section (Sect. 14.2) critically reviews research and investigations carried out for the last 15 years (2006–2021) with respect to MP pollution impacts on seafood organisms including fish, sharks, mussels, oysters, clams, prawns, shrimps and seaweeds covering 36 locations of countries in the world including the Atlantic Ocean, Australia, the Baltic Sea, Bangladesh, Belgium, Brazil, Canada, Chile, China, Fiji, France, the Gulf of Mexico, India, Indonesia, Iran, Italy, Japan, Malaysia, the Mediterranean Sea, the Netherlands, North Pacific Central Gyre, North Pacific Subtropical Gyre, North Sea, Norway, Portugal, Saudi Arabia, Scotland, South Pacific Subtropical Gyre, Spain, Tanzania, Thailand, Turkey, the UK, the USA and Vanuatu.

The following terminology has been used in this chapter: *bathypelagic*, the deep sea where the environment is dark, cold and deep, between 1000 and 3000 m; *benthic*, organisms that feed and live near or on the bottom sediments; *benthopelagic*, species that live and feed near or on the bottom as well as throughout the water column; *demersal*, those that live on or near the bottom and feed on organisms (plant or animal); *detrivore*, organism that feeds on dead and decomposing organic matter; *dw*, dry weight; *fragments*, broken pieces of hard plastic; *films*, thin soft plastic, such as plastic bags or paint chips; *fibres*, long and thin fibrous type of plastic; *GI*, gastrointestinal tract (stomach, intestine and the digestive tube of fish); *epipelagic*, the part of the ocean where there is enough sunlight for algae to utilise photosynthesis; *herbivores*, those that feed on plants, algae and phytoplankton; *mesopelagic*, fish and other organisms inhabiting in the intermediate depths of the sea, between about 200 and 1000 m depths; *MP*, microplastics; *neritic species*, those living in coastal areas; *oceanic species*, species living in open waters of the sea; *omnivores*, those feeding everything, both plant and animals including detritus; *pelagic*, those forage on organisms that live at the surface or throughout the water column; *planktivores*, those eating zooplankton and phytoplankton, including large zooplankton; *reef-associated species*, those living and feeding on or near coral reefs, and benthic or benthopelagic fishes that consistently associate with hard substrates of coral, algal or rocky reef fishes are those individuals that live on a coral reef (https://www.fishbase.se/glossary/Glossary.php?q=reef-associated). In this paper, we considered benthic as a synonym of demersal and vice versa.

14.2.1 Atlantic Ocean

14.2.1.1 MP-Contaminated Mesopelagic Marine Fishes

1. Around ten mesopelagic fish species (slender snipe eel, *Nemichthys scolopaceus*; the blunt snout smooth-head, *Xenodermichthys copei*; boa dragonfish, *Stomias boa*; glacier lantern fish, *Benthosema glaciale*; spotted barracudina, *Arctozenus risso*; lancet fish, *Notoscopelus kroyeri*; silvery light fish, *Maurolicus muelleri*; jewel lanternfish, *Lampanyctus crocodilus*; spotted lanternfish, *Myctophum punctatum*; silver hatchet fishes, *Argyropelecus* spp.) from the North Atlantic Ocean were investigated for MP contamination (research was carried out during 2013–2014). The highest MP ingestion was recorded in the following three species: glacier lantern fish, *B. glaciale* (0.33 plastic particles/fish) → spotted barracudina, *Arctozenus risso* (0.29 plastic particles/fish) → lancet fish, *Notoscopelus kroyeri* (0.16 plastic particles/fish) (analysis based on more than ten individual fishes). No plastics were found in the following three species: jewel lanternfish, *L. crocodilus* (0 plastic/fish); spotted lanternfish, *M. punctatum* (0 plastic/fish) and silver hatchet fishes, *Argyropelecus* spp. (0 plastic/fish). On average 11% of fish ingested MPs, 93% ingestion of MP was fibres and the most prominent colours of fibres were black (42%) and blue (34%). The study found

that the average proportion of fish ingesting plastic was higher at day time than during the night, though there were no significant differences between the proportions of individuals per species with plastic at different times of day (Lusher et al. 2016). The 11% MP ingestion by mesopelagic fishes in the North Atlantic Ocean (Lusher et al. 2016) are comparable to other areas such as 9.2% in the mesopelagic fishes of the North Pacific Subtropical Gyre (Davison and Asch 2011), whereas it was 36.5% (much higher) in the English Channel for pelagic and demersal fishes (Lusher et al. 2013). The above variation in MP ingestion could be due to the differences in feeding habits of species or MP loads in the surrounding areas.

2. Seven mesopelagic fishes (spotted lanternfish, *Myctophum punctatum*; glacier lanternfish, *Benthosema glaciale*; white-spotted lanternfish, *Diaphus rafinesquii*; rakery beacon lamp, *Lampanyctus macdonaldi*; stout saw palate, *Serrivomer beanii*; and scaly dragonfish, *Stomias boa* and *Gonostoma denudatum*) from the Northwest Atlantic were found contaminated with MPs (research was carried out during April–May 2015). The highest average number of MPs in the gut (stomach) contents was recorded in the following orders: stout saw palate, *S. beanii* (2.36 MP particles/fish) → Spotted lanternfish, *M. punctatum* (2.28 MP particles/fish) → *G. denudatum* (2.2 MP particles/fish). On average 73% of the fish ingested MPs, 98% of MPs ingested were fibres, polyethylene (PE) was the major polymer type identified in the fishes and black (67%) was the dominant colour of the fibres (Wieczorek et al. 2018). The much higher frequency of MP ingestion in fish in the study of Wieczorek et al. (2018) could be due to a number of reasons: (a) the study was carried out in an MP pollution 'hot spot' area (off the Newfoundland coast where MPs are highly concentrated), (b) the mesopelagic fishes that mistakenly ingested MPs as prey items during the night (since the studied seven species migrate to the surface only at night to feed) and (c) or it may have occurred through trophic transfer from their prey species (since the most common preys of mesopelagic fish were copepods, euphausiids and amphipods larvae). The decapods are known to ingest microplastics (Carpenter et al. 1972; Setälä et al. 2014; Desforges et al. 2015), thus facilitating the trophic transfer of MPs to organisms higher in the food chain (decapod is a prey to fish).

14.2.2 Australia

14.2.2.1 MP Contaminated Marine Fishes and Marine Prawns Including Different Trophic/Feeding Levels (Carnivores, Detrivores) and Habitats (Benthopelagic, Demersal, Reef-Associated).

1. Three benthic foraging fishes from the Sydney Harbour estuary (yellow fin bream, *Acanthopagrus australis*; sea mullet, *Mugil cephalus*; and silverbiddy, *Gerres subfasciatus*) were found contaminated with MPs (research was carried out during March–June 2015). On average 43% of the fish ingested MPs. Around

83% of MPs ingested were fibres (gastrointestinal tract or GI analysis). Trends of MP ingestion was in the following order: Sea mullet, *M. cephalus* (2.5 MP particles/fish) (benthopelagic) → yellow fin bream, *A. australis* (0.6 MP particles/fish) (demersal) → silverbiddy, *G. subfasciatus* (0.1 MP particles/fish) (demersal) (Halstead et al. 2018). The most prevalent polymers ingested were acrylic and polyester fibres, which could have originated from clothing since a single garment is known to produce in excess of 1900 fibres per wash (Browne et al. 2011).

2. Five fish species from fish markets of Brisbane and Sydney including two demersal (goatfish, *Upeneichthys lineatus* and *Parupeneus indicus*), two benthopelagic (paddle tail, *Lutjanus gibbus*; sea mullet, *Mugil cephalus*) and one reef-associated (common coral trout, *Plectropomus leopardus*) were found contaminated with MPs (research was carried out in 2019). About 61.6% of the fish ingested MPs, and 83.6% of ingested MPs were fibres (GI). Trends of MP ingestion were in the following order: paddle tail, *L. gibbus* (MP occurred in 75% fish) (benthopelagic, carnivore) → common coral trout, *P. leopardus* (MP occurred in 60% fish) (reef fish, carnivore) → goat fish, *U. lineatus* and *P. indicus* (MP occurred in 50% fish) (demersal, carnivore) → sea mullet, *M. cephalus* (MP occurred in 48% fish) (benthopelagic, detrivore). No trends were found among the fish sizes, or the weights of the GI and the amount of microplastics counted (Wootton et al. 2021). The above study (Wootton et al. 2021) demonstrates that MP contamination can spread across different trophic/feeding levels (carnivores, detrivores) and habitats (reef, benthopelagic, demersal). All of these fish species are also important fishery supporting the economy, employment and food sources in Australia. The higher MP ingestion by fish (61.6%) and high abundance of fibres (83.6% in fish stomachs) in coastal fish (Brisbane and Sydney) may be related to the higher population living in Australian coastal areas and consequences of generation of high volume of plastic into the environment.
3. Out of 21 fishes and 1 cephalopod (squid) investigated from the Southern Ocean (Tasmania), only 1 species (the Antarctic toothfish, *Dissostichus mawsoni*) was found contaminated with MPs. On average only 0.3% of the fish ingested MPs, and acrylic resin fragments were recovered from the Antarctic toothfish (Cannon et al. 2016). The low ingestion of MPs (0.3%) by the Antarctic toothfish could be related to less-populated coastlines and lower levels of marine plastic pollution in the sampled Southern Ocean (Tasmania).
4. Five different species of raw seafood including the Pacific oysters/giant cupped oysters (farmed) (*Crassostrea gigas*), brown tiger prawns (farmed) (*Penaeus esculentus*), blue swimmer crabs (wild) (*Portunus armatus*), Gould's flying squid (wild) (*Nototodarus gouldi*) and Australian pilchards (wild sardines) (*Sardinops neopilchardus*) (pelagic, planktivore) purchased from the local fish market were found contaminated with MPs (research was carried out in 2019). Sardines contained the highest total plastic mass concentration and squid the lowest total plastic mass concentration in the following orders: sardines, *S. neopilchardus* (2.9 mg/g tissue (pelagic) → blue crabs, *P. armatus* (0.34 mg/g tissue) (benthic) → tiger prawns, *P. esculentus* (0.07 mg/g tissue) (benthic) → oysters,

C. gigas (0.04 mg/g tissue) (benthic) → squid, *N. gouldi* (0.01 mg/g tissue) (pelagic-oceanic) (Ribeiro et al. 2020). The study found that the total concentration of MPs is highly variable among species. It is likely that the high concentration of MPs in sardines could be due to mistakenly ingesting MPs as 'prey/food' and another possibility of higher MPs in sardine could be the potential transfer of MPs from the gastrointestinal tract to flesh (muscle) during food processing and handling. In another study (Collard et al. 2017), sardines also ingested more MPs compared to herrings and anchovies, which the authors believed to be due to the highest filtration area and the closest gill rackers possessed by sardines (Collard et al. 2017).

14.2.3 Baltic Sea

14.2.3.1 MP Contaminated Demersal and Pelagic Marine Fishes

1. Five fishes including three demersal (Atlantic cod, *Gadus morhua*; common dab, *Limanda limanda*; and European flounder, *Platichthys flesus*) and two pelagic fish species (Atlantic herring, *Clupea harengus*, and Atlantic mackerel, *Scomber scombrus*) from the Baltic Sea were investigated for contamination with MPs (research was carried out in 2013). No MPs were detected in common dab, *L. limanda*, and Atlantic herring, *C. harengus*. On average 4.9% of the fish ingested MPs, and major MPs were of clear and white colour. Trends of MP ingestion were Atlantic mackerel, *S. scombrus* (30.8% MPs ingestion) (pelagic-neritic) → European flounder *P. flesus* (10% MP ingestion) (demersal) → Atlantic cod, *G. morhua* (1.4% MP ingestion) (benthopelagic) (Rummel et al. 2016). The presence of MPs in the fish gastrointestinal tract (GI) may reflect their occurrence in the environment where the species live. Moreover, the higher ingestion of MPs by mackerel (pelagic fish) may have occurred accidentally or mistakenly during normal feeding activity as prey (small fish).

14.2.4 Bangladesh

14.2.4.1 MP Contaminated Benthopelagic, Demersal, Pelagic Marine Fishes and Shallow and Offshore Marine Shrimps

1. Three fishes from the Bay of Bengal (pink Bombay duck, *Harpadon nehereus*; white Bombay duck, *H. translucens*; and goldstripe sardine, *Sardinella gibbosa*) were found contaminated with MPs (research was carried out during September 2017 to March 2018). Trends in MP ingestion were in this order: pink Bombay duck, *H. nehereus* (8.72 MP particles/g) (benthopelagic) → Bombay duck, *H. translucens* (5.80 MP particles/g) (demersal) → goldstripe sardine, *S. gibbosa*

(3.20 MP particles/g (pelagic-neritic). On average 53% of MPs ingested were fibres, and the dominant colour of the MPs was white (mean 51.33%) (Hossain et al. 2019). The three fish species may have been exposed to MPs inadvertently during feeding on the surface or at the bottom sediments while searching for food.

2. Two shrimp species (black tiger shrimp, *Penaeus monodon* (benthic), and brown shrimp/speckled shrimp, *Metapenaeus monoceros* (benthic)) from the Bay of Bengal (BoB) (research was carried out during September 2017 to March 2018) were found contaminated with MPs (GI). Brown shrimp/speckled shrimp, *M. monoceros*, had ingested more MPs (7.8 MP particles/shrimp) compared to tiger shrimp *P. monodon* (6.6 MP particles/shrimp). MP ingested was dominated by filaments and fibres and was of black colour. Filaments were 57% and 58% in tiger and brown shrimp, respectively; fibres were 32% and 57% in brown and tiger shrimp, respectively; and the colour of filaments and fibres were black (48% and 51% in tiger and brown shrimps, respectively) (Hossain et al. 2020). The reason behind the black coloured MPs found in shrimps is that almost all nets used in the BoB are made of black twines. The study demonstrates that both shallow-water shrimp (brown shrimp) and offshore deep-water shrimp (tiger shrimp) were contaminated with MPs. MPs detected in fish (Hossain et al. 2019) and shrimp species in Bangladesh (Hossain et al. 2020) are commercial species and support employment, economy and nutrition to local people. Some species

Fig. 14.1 Dry fish of the Ganges river sprat, *Corica soborna* or *kachki* are low-cost protein and mineral sources for poor communities in Bangladesh. Dried fish are consumed whole (without removing the gastrointestinal tract and gills); therefore, there is a significant risk of ingestion of MPs and MP-adsorbed high-risk chemical pollutants via consumption of dried fish. In Bangladesh, seafood provides about 60% animal protein supplies (Photo by Golam Kibria, July 2021)

such as Bombay duck and small brown shrimp are sold mainly in dried form without removing the digestive tract (locally called *shutki*, Fig. 14.1) and consumed whole. Dried fishes are considered low-cost protein sources in many developing countries such as Bangladesh where the annual dried fish consumption is reported to be 370 g/capita (Needham and Funge-Smith 2014). The consumption of whole dried Bombay duck and brown shrimp pose a risk to humans as it may be a pathway of MP transfer to humans (Kibria 2018; Kibria et al. 2021). The further risk posed by the contamination of dried fish (fish meal) with MPs is that they are used to feed farmed fishes and poultry in Bangladesh and can thus be transferred to livestock as well. For example, about 15% of marine fish and other organisms are converted to fish meal in Bangladesh for animal feed (principally poultry feed; *https://www.slideshare.net/worldfishcenter/lll--belton-b-mostafa-h-dried-fish-production-consumption-and-trade*). In addition, high-risk chemical pollutants (DDT, heavy metals, PAHs, PCBs) adsorbed in MPs can be transferred to human consumers via dried fish.

14.2.5 Belgium

14.2.5.1 MP Contaminated Molluscs (Mussels) and Crustaceans (Shrimps)

1. Both farmed blue mussels, *Mytilus edulis* (from bottom culture), and wild blue mussels, *M. edulis* (from open sea) (benthic, filter feeders), were found contaminated with MPs (research was carried out in 2013). Trends in MP ingestion were in this order: wild blue mussels, *M. edulis* (3.8 fibres/10 g mussels), and farmed blue mussels, *M. edulis* (3.5 fibres/10 g mussels). Orange fibres were the main MPs detected in wild mussels (De Witte et al. 2014). The higher MP contamination of wild mussels may be related to the higher MP pollution in the wild mussel harvesting area. The high orange fibre ingestion in wild mussels can be related to the abundance of such coloured nets and plastics in the local areas.
2. Common/brown shrimp, *Crangon crangon*, from Southern North Sea and Channel areas was found contaminated with MPs (research was carried out during March 2013–March 2014). On average 63% of the shrimp ingested MPs (1.23 MP particles/shrimp). Around 96.5% of MPs ingested were fibres, and purple-blue (43%) coloured fibres were the dominant MPs ingested by shrimps (Devriese et al. 2015; Barboza et al. 2018). The study reveals that common/brown shrimp is able to consume MPs which they have ingested either accidentally or as a colour preference. No MP particles were found in the tail muscle tissue (edible part), suggesting that MP particles are present in the digestive tract, the head or gills of the shrimp and not in the abdominal muscle tissue. High MP ingestion (fibres) by common/brown shrimp may reflect higher local plastic pollution levels sourced from local fishing activities, recreational boating, laundry and domestic wastewater. Another possibility is the accidental ingestion of fibres

by common/brown shrimp while searching and eating food in sediments. As it may not be possible to remove the digestive tract from the small-sized shrimp, therefore, there is a possibility that the MPs accumulated in the digestive tract could be transferred to humans via seafood consumption. To avoid MP consumption and MP transfer to humans, it is suggested to remove the digestive tract of the shrimp before cooking.

14.2.6 Brazil

14.2.6.1 MP Contaminated Freshwater Fishes Including Detrivores, Herbivores, Invertivores, Omnivores and Estuarine Catfishes

1. Thirteen fishes with different feeding habits from urbanised and non-urbanised streams (Ivaí River basin) were found contaminated with MPs (research was carried out in May 2018). These species were detrivores (where >60% of fish diet is detritus—*Hypostomus ancistroides*, *Hypostomus strigaticeps*, *Psalidodon fasciatus*), herbivores (where >60% fish diet is plants—*Psalidodon paranae*), invertivores (where >60% fish diet is invertebrates—*Characidium zebra*, *Piabarchus stramineus*, *Rhamdia quelen*), omnivores (where fish diet is macroinvertebrates (50%) and plants (50%)—*Bryconamericus iheringii*, *Psalidodon fasciatus*, *Rineloricaria pentamaculata*) (Garcia et al. 2020). The study found that fishes from urbanised streams had higher MP intake compared to non-urbanised streams. Trends of MP ingestion were urbanised fishes (37% MP ingestion) and non-urbanised fishes (15.5% MP ingestion). The higher MP intake in urbanised fish may indicate that urban streams are more polluted with plastics than the non-urbanised streams and can be related to the high population and high waste generated in urban areas. In addition, among detrivores, herbivores, invertivores and omnivores, only omnivorous fishes were found positively correlated with MP ingestion. This can be related to the foraging behaviour of omnivores to a wide range of resources and throughout the water column. A similar result was also found by Andrade et al. (2019) who reported that fish from omnivorous habit ingested a greater number of MPs (compared to herbivores, omnivores and carnivores) in the Amazon.
2. Three catfish species (epibenthophagous—Madamango sea catfish, *Cathorops spixii*, *Cathorops agassizii*; Pemecou sea catfish, *Sciades herzbergii*) from Goiana Estuary, Northeast Brazil, were found contaminated with MPs (research was carried out during January 2006 to August 2008). On average, 23% of fish ingested MPs, and 23% of MPs were blue nylon fragments. Trends in MP ingestion were in this order: *Cathorops agassizii* (33% MP ingestion) (benthopelagic) → Madamango sea catfish *Cathorops spixii* (18% MP ingestion) (demersal) → Pemecou sea catfish *Sciades herzbergii* (17% MP ingestion) (demersal) (Possatto et al. 2011). These catfish species are epibenthophagous (*epibenthophagous = organisms that live on or in or near or above the seabed, river, lake and*

prey on small animals living on the surface of the sediment) (Barletta and Blaber 2007). MP ingestion may have happened during normal feeding activity of the catfish or indirectly while on preying smaller fish that have been previously contaminated with plastics. Fishery activities could be the source of nylon fragments.

14.2.7 Canada

14.2.7.1 MP Contaminated Molluscs (Clams, Oysters, Farmed and Wild Blue Mussels)

1. Manila clams, *Venerupis philippinarum*, and the Pacific oysters, *Crassostrea gigas*, from shellfish farms and coastal areas of British Columbia were found contaminated with MPs (research was carried out in May–June 2016). Around 90.5% of MPs ingested were fibres, and fibres in clams/oysters were predominately of clear and blue coloured. Trends in MP ingestion were Pacific oysters, *C. gigas* (0.22 MP particles/oyster) and Manila clams (0.16 MP particles/clams) (Covernton et al. 2019). The study did not find any significant differences in MP concentrations between shellfish or their habitat (between shellfish aquaculture and non-aquaculture sites) in coastal British Columbia, Canada. This may suggest that the origin of MPs in the studied area was not from shellfish farms but rather either from sources like textiles or sewage or atmospheric fallout.
2. Farmed and wild blue mussels, *Mytilus edulis*, from Halifax Harbour were found contaminated with MPs (research carried out between August 2012 and May 2013). Microfibres (as microplastics or MPs) in farmed mussels (126.5 fibres/mussels) were higher than in the wild mussels (75 fibres/mussels) (Mathalon and Hill 2014). In this case, farmed mussels were grown on plastic polypropylene (PP) lines, which may have released MP fibres into the surrounding environment, thus possible contamination and accumulation of farmed mussels with MPs (fibres). It is therefore suggested that, as a preventive and safety measure, both farmed and wild mussels should be depurated in clean, plastic-free seawater before human consumption. Additionally, the accumulation of microfibres in farmed mussels compared to wild mussels can cause blockages in the digestive tract of mussels leading to a decrease in fitness (Wright et al. 2013). It is important to state here that microplastics have also been detected in higher trophic-level organisms. For example, ten species of fish (demersal and pelagic) from the English Channel had 36.5% microplastics in them (Lusher et al. 2013), and one third (35%) of fish (mesopelagic and epipelagic) caught from the North Pacific Central Gyre also had microplastics in them (Boerger et al. 2010) (see also Fig. 14.2).

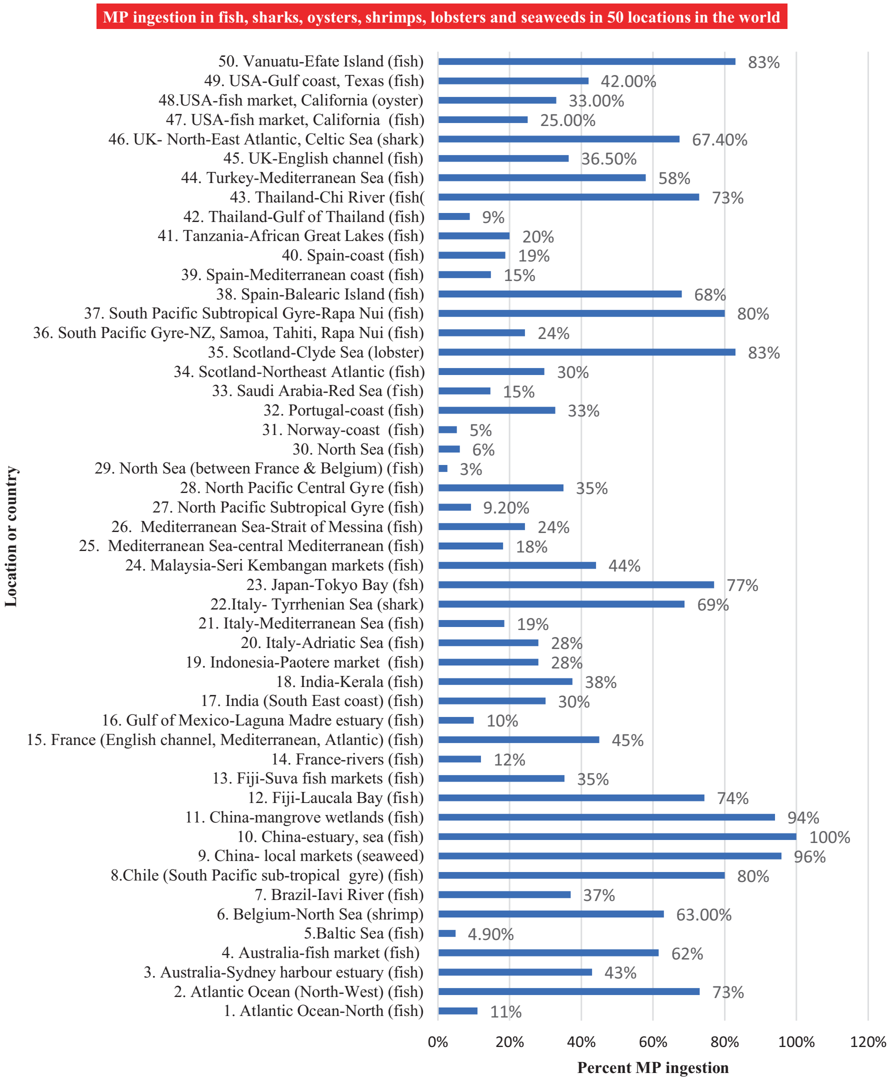

Fig. 14.2 MP ingestion (% of ingestion) in seafood organisms [*based on GI counts, numbers (1-–50) are references*: (1) Lusher et al. 2016; (2) Wieczorek et al. 2018; (3) Halstead et al. 2018; (4) Wootton et al. 2021; (5) Rummel et al. 2016; (6) Devriese et al. 2015; (7) Garcia et al. 2020; (8) Ory et al. 2017; (9) Li et al. 2020; (10) Jabeen et al. 2017; (11) Huang et al. 2020; (12) Ferreira et al. 2020; (13) Wootton et al. 2021; (14) Sanchez et al. 2014; (15) Collard et al. 2017; (16) Phillips and Bonner 2015; (17) Kumar et al. 2018; (18) Kripa et al. 2014; (19) Rochman et al. 2015; (20) Avio et al. 2015; (21) Romeo et al. 2015; (22) Valente et al. 2019; (23) Tanaka and Takada 2016; (24) Karbalaei et al. 2019; (25) Romeo et al. 2015; (26) Battaglia et al. 2016; (27) Davison and Asch 2011; (28) Boerger et al. 2010; (29) Foekema et al. 2013; (30) Rummel et al. 2016; (31) Bråte et al. 2016; (32) Neves et al. 2015; (33) Baalkhuyur et al. 2018; (34) Murphy et al. 2017; (35) Murray and Cowie 2011; (36) Markic et al. 2018; (37) Ory et al. 2017; (38) Nadal et al. 2016; (39) Compa et al. 2018; (40) Bellas et al. 2016; (41) Biginagwa et al. 2016; (42) Klangnurak and Chunniyom 2020; (43) Kasamesiri and Thaimuangphol 2020; (44) Güven et al. 2017; (45) Lusher et al. 2013; (46) Parton et al. 2020; (47) Rochman et al. 2015; (48) Rochman et al. 2015; (49) Peters et al. 2017; (50) Bakir et al. 2020]

14.2.8 Chile

14.2.8.1 MP Contaminated Marine and Coastal Fishes (Pelagic, Omnivore, Herbivore and Carnivore) and Crabs

1. Five fish species including greenfish, *Girella laevifrons* (omnivore, pelagic-neritic); *Scartichthys viridis* (herbivore, demersal); old black, *Graus nigra* (carnivore, demersal); *Helcogramoides chilensis* (carnivore, demersal); and *Auchenionchus microcirrhi*s (carnivore, demersal) from the central coast of Chile were found contaminated with MPs (research was carried out in summer 2015) (Mizraji et al. 2017). Around 99% of MPs ingested were fibres. Trends in MP ingestion were omnivorous fish (*G. laevifrons*) ingested a higher amount of MPs (fibres) (average 61 MPUs; *MPUs = microplastic units*) than herbivores (average 14 MPUs) and carnivores (average 10 MPUs) (Mizraji et al. 2017). The higher amount of MP fibres found in omnivorous fish *G. laevifrons* can be related to their feeding habit which feeds on a wide variety of food items, including red algae and invertebrate species (Mizraji et al. 2017). In particular, the higher percentage of red-coloured fibres found (79%) in their digestive tract may mean the fishes have confused red MPs as red algae (as food). Moreover, the ingested MP fibres can result in a number of sub-lethal or lethal effects, such as (a) gut blockage, false satiety sensation, physical injury in exposed fish (Browne et al. 2008; Wright et al. 2013); (b) hepatic stress due to bioaccumulation of chemical pollutants in plastics (Rochman 2013) and (c) effect on reproduction and growth as found with the European green crab, *Carcinus maenas* (Watts et al. 2015).
2. Amberstripe scad *Decapterus muroadsi* (pelagic-oceanic) from South Pacific Subtropical Gyre (Rapa Nui/Easter Island) was found contaminated with MPs (research was carried out in April 2018). Around 80% of fish ingested MPs (2.5 MP particles/fish), and 80% of ingested MPs were hard fragments. The majority of MP ingested were of blue colour (40%). Ingested blue microplastics resemble copepod prey (Ory et al. 2017). The higher ingestion (80% by *D. muroadsi*) rate could be due to the large ratio of microplastics to plankton in the highly oligotrophic waters of the Subtropical Gyres. According to Moore et al. (2001), the mass of plastic was six times than that of plankton in the North Pacific Central Gyre. In addition, fish may have mistakenly ingested MPs as natural prey (copepod) since they have ingested mostly blue MPs that are similar in colour to blue-pigmented copepod species that they commonly prey upon (Ory et al. 2017).

14.2.9 China

14.2.9.1 MP Contaminated Seaweed, Freshwater Fishes (Benthopelagic), Seawater Fishes (Pelagic, Benthopelagic, Benthic, Demersal, Carnivore) and Mussels

1. Twenty-four brands of commercial edible seaweed nori, *Pyropia* spp., from local markets (east coastal zone of China) was found contaminated with MPs (23 out of 24 seaweed nori or 95.8% of seaweeds nori brands were contaminated with MPs (research was carried out during January and February 2019)). Around 85.2% of MPs were fibres (1.8 MPs/g dw). Polyester fibres (18.9%) were the most dominant MPs detected, followed by rayon (6.6%), polypropylene (4.0%) and polyamide (1.8%). 41.4% of MPs were of blue-green colour (Li et al. 2020). 95.8% of seaweeds having MPs in them is much greater compared to 20% MPs in canned sprats and sardines (Karami et al. 2018). Moreover, the high frequency of MPs detected in seaweed nori (95.8%) shows that MPs can also be translocated through popular seafood species, and there is a possibility of transfer of hazardous chemicals adsorbed in MPs to biota and finally to humans (who use seaweeds as food). The fibres (85.2%) found in seaweed nori may come from clothes or of plastic lines used for the attachment of seaweed seedlings or fishing nets.
2. Twenty-one marine water fishes (consisting of five pelagic + four benthopelagic + 11 demersal + one benthic) from Yangtze estuary, East China Sea and South China Sea, and six freshwater fishes (benthopelagic) from Taihu Lake, China, were found contaminated with MPs (research was carried out during May to December 2015). The highest abundance of MPs was found in two marine species, *Thamnaconus septentrionalis* (7.2 MP particles/fish; demersal, carnivore), and was followed by three-lined tongue sole, *Cynoglossus abbreviates* (6.9 MP particles/fish; demersal, carnivore). On average, 100% of the sea fish and 95.7% the freshwater fish ingested MPs (which are globally highest; see Fig. 14.2 for MP ingested in other countries or locations). Around 61.4% of ingested MPs were fibres, and the dominant MPs ingested were transparent in colour. The abundance of plastics by items/individual was significantly higher in sea benthopelagic fishes (3.1 MP particles/fish) compared to freshwater benthopelagic fishes (2.4 MP particles/fish). The abundance of MPs was significantly higher in sea demersal fishes (4.7 MP particles/fish) than in sea pelagic fishes (3.18 MP particles/fish) (Jabeen et al. 2017). The higher ingestion of MPs by sea fishes may reflect the availability of various types of plastic particles in the marine environment compared to freshwater. Moreover, the higher ingestion by the demersal fish (bottom-living fish) may suggest the higher abundance of MPs in the sea bed/sediment (Woodall et al. 2014).
3. Wild and farmed blue mussels, *Mytilus edulis*, from coastal areas (12,400 miles or 2/3 of the total coastline) were found contaminated with MPs (research was carried out during July to October 2015). Wild mussels had more MPs (4.6 MP

particles/mussel) compared to farmed mussels (3.3 MP particles/mussel); 65% of ingested MPs by mussels were fibres (Li et al. 2016). The detection of MPs in both wild and farmed mussels may indicate that the microplastic pollution is ubiquitous; higher mussel contamination with MPs was found from the areas with intensive human activities.

4. Twenty-two benthic fishes (comprising 14 omnivores + eight carnivores) and ten pelagic fishes (all omnivores) from Zhanjiang mangrove, China wetlands, were found contaminated with MPs (research was carried out in March 2017). Around 94% of the fish was contaminated with MPs, and 70% of ingested MPs were fibres. MPs ingested were of different colours (transparent, white, black, blue, red, green, yellow and brown). The feeding habits of fish played an important role in MP ingestion in this study. For example, MP abundance was higher in benthic species (3.19 MP particles/fish) than in pelagic species (2.02 MP particles/fish) (benthic > pelagic). Hong Kong grouper, *Epinephelus akaara*, had the highest MP particles/fish (8 MP particles/fish) (benthic, carnivore) and was followed by Keel-jawed needle fish, *Tylosurus melanotus* (6.5 MP particles /fish) (reef-associated, omnivore) (Huang et al. 2020). The higher ingestion of MPs by benthic fish species can be due to higher contamination of the wetlands by MPs and abundance of benthic fish species in the wetlands. The higher ingestion of MPs in other demersal/benthic fish species was also reported from the East China Sea (Jabeen et al. 2017); western Mediterranean Sea (Battaglia et al. 2016); the coast of Portugal (Neves et al. 2015); Northeast Atlantic, Scotland (Murphy et al. 2017); and North Adriatic Sea (Avio et al. 2015). In contrast, higher MP contamination was found in pelagic species in the North Sea (Rummel et al. 2016), Baltic Sea (Rummel et al. 2016), English Channel (Lusher et al. 2013), South Pacific Subtropical Gyre (Markic et al. 2018) and the Mediterranean coast of Turkey (Güven et al. 2017) (see Fig. 14.2).

14.2.10 Fiji

14.2.10.1 MP Contaminated Demersal, Benthopelagic, Reef, Carnivore and Detrivore Marine Fishes

1. Fishes (bluestriped goatfish, *Upeneichthys lineatus*; common coral trout, *Plectropomus leopardus*; paddle tail, *Lutjanus gibbus*; sea mullet, *Mugil cephalus*; and yellowspot goatfish, *Parupeneus indicus*) from fish markets in Suva were found contaminated with MPs, and 35.3% of the fish ingested MP particles (research was carried out in 2019). Around 50% of the ingested MPs were films. Trends of MP ingestion were common coral trout, *P. leopardus* (1.5 MP particles/fish) (reef fish, carnivore) → paddle tail, *L. gibbus* (1.0 MP particles/fish) (benthopelagic, carnivore) → bluestriped goat fish, *U. lineatus* (0.5 MP particles/fish) (demersal, carnivore) → yellowspot goatfish *P. indicus* (0.5 MP particles/fish) (benthopelagic) → and sea mullet, *M. cephalus* (0.5 MP particles/fish)

(benthopelagic, detrivore) (Wootton et al. 2021). The above study (Wootton et al. 2021) demonstrates that MP contamination spread across the different trophic/feeding levels (carnivores, detrivores) and habitats (reef, benthopelagic, demersal). All of these fish species are also an important fishery supporting the economy, employment and food sources in Fiji.

2. Four fish species (*Mugil* spp., *Lethrinus* spp., *Lutjanus* spp., *Chanos* spp.) from the Laucala Bay were found contaminated with MPs. About 74.3% of the fish ingested at least one MP particle, and 60.2% of the ingested MPs were fibres (16.4 MP particles/fish) (Ferreira et al. 2020). The presence of MP in fish stomachs reflects MP pollution in the local environment where the fishes live. Plastic film dominated in the fish guts from Fiji that could have been originated from plastic bags and soft food packaging.

14.2.11 France

14.2.11.1 MP Contaminated Marine and Freshwater Fishes

1. Planktivorous fishes (European anchovy, *Engraulis encrasicolus*; Atlantic herring, *Clupea harengus*; European pilchard, *Sardina pilchardus*) from the English Channel, the Northwestern Mediterranean Sea and the Northeastern Atlantic (Bay of Biscay), were found contaminated with MPs (research was carried out during January–October 2013). Around 55% of the fish ingested MPs, and 55% of ingested MPs were fibres. Trends in MP ingestion (as fibres) were sardine, *S. pilchardus* (81% of fibre ingestion) (pelagic-neritic) → anchovy, *E. encrasicolus* (48% of fibre ingestion) (pelagic-neritic) → herring, *C. harengus* (38% of fibre ingestion) (benthopelagic) (Collard et al. 2017). Sardines ingested more MPs (fibres) than herrings and anchovies which could be related to the highest filtration area and the closest gill rackers found in sardines (Collard et al. 2017).
2. Gudgeons, *Gobio gobio* (benthopelagic), from French rivers were found contaminated with MPs (research was carried out in 2012). Around 12% of the fish ingested MPs (Sanchez et al. 2014). This work provides evidence that freshwater fish is also ingested MPs.

14.2.12 Gulf of Mexico

14.2.12.1 MP Contaminated Urbanised and Non-urbanised Fishes (Benthic, Pelagic)

The freshwater fish (from creeks, rivers) and marine fish (from an estuary, Laguna Madre) of the Gulf of Mexico were found contaminated with MPs (research was carried out between September 2013 and January 2014). Around 8.2% of the fish ingested MPs (8% freshwater fish and 10.4% marine fish had MPs). Polymers ingested by fish include polypropylene (PP), polyester (PET), polymethacrylate, polystyrene and nylon; urbanised fish ingested more MPs (29%) than fishes from non-urbanised streams (5%). The occurrence of MPs in fish by habitat (benthic, pelagic) and trophic guilds (herbivore/omnivore, invertivore, carnivore) was found similar (Phillips and Bonner 2015). The higher ingestion of urbanised fish may reflect urban waterways are more polluted with MPs compared to non-urbanized waterways. Nylon can be sourced from wastewater treatment facilities and terrestrial habitats where sewage sludge had been applied (Browne et al. 2011) to row crops.

14.2.13 India

14.2.13.1 MP Contaminated Marine Fishes and Mussels

1. Marine fish (anchovy, *Stolephorus commersonnii* from Punnapra in Alappuzha district, Kerala), was found contaminated with MPs (GSI) (research was carried out in August 2013). Around 37.5% of anchovies ingested MPs (Kripa et al. 2014). It is likely that anchovies may have ingested the MPs while feeding plankton or preying.
2. Green mussel, a commercially important bivalve, *Perna viridis*, from the Coastal waters/fishing harbour of Chennai was found contaminated with MPs (research was carried out in April 2017). Around 87% of ingested MPs were polystyrene (Naidu 2019). It is likely that mussels ingested MPs while filter feeding either accidentally or selectively and consumed sinking MPs from the waters. The filtration rate of a standard mussel is ~24 L/day (Clausen and Riisgård 1996). The ingested MPs can cause a range of negative consequences to the bivalves' health (Cole et al. 2013) and could transfer to the food web/food chain (Farrell and Nelson 2013) and ultimately to human consumers (Kibria 2018).
3. Marine fishes (Indian mackerel, *Rastrelliger kanagurta*, and Honeycomb Grouper, *Epinephelus merra*) from the southeast coast (Thirespuram and Punnakayal) were found contaminated with MPs. Around 30% of the fish ingested MPs, and 80% of ingested MPs were fibres. Ingested fibres were of black, red and translucent colour. Trends in MP ingestion were honeycomb grouper, *E. merra*, →Indian mackerel, *R. kanagurta* (Kumar et al. 2018). The ingestion of MPs may have occurred accidentally during normal feeding or through consuming MP-contaminated prey. The study shows a high intake of fibres by fish. Therefore, the risk of MP transfer to humans can be avoided if fishes are degutted (stomachs and intestine of fishes are removed) prior to consumption.

14.2.14 Indonesia

14.2.14.1 Ms Contaminated Marine Fishes

1. Eleven fish species from the Paotere fish market were investigated in 2014 for contamination with MPs. Those 11 species were Indian mackerel, *Rastrelliger kanagurta*; humpback red snapper, *Lutjanus gibbus*; oxeye scad, *Selar boops*; rabbitfish, *Siganus argenteus*, *S. fuscescens*, and *S. canaliculatus*; skipjack tuna, *Katsuwonus pelamis*; shortfin scad, *Decapterus macrosoma*; silver-striped round herring, *Spratelloides gracilis*; Nile tilapia, *Oreochromis niloticus*; and fish from the family of Carangidae (could not be identified to the genera level). Out of 11 species, six fish species including Indian mackerel, *R. kanagurta*; shortfin scad, *D. macrosoma*; silver-striped round herring, *S. gracilis*; fish from the family Carangidae; and two species of rabbitfish (*S. argenteus* and *S. canaliculatus*) were found contaminated (in GSI) with MPs. Around 28% of the fish ingested MPs, and 60% of ingested MPs were fragments. Trends in MP ingestion were fish of family Carangidae (5.9 MP particles/fish) (reef-associated) → shortfin scad, *D. macrosoma* (2.5 MP particles/fish) (reef-associated) → silver-striped round herring, *S. gracilis* (1.1 MP particles/fish) (pelagic-neritic) → Indian mackerel, *R. kanagurta* (1.0 M*P* particles/fish) (pelagic-neritic) → rabbitfish, *S. argenteus* (0.5 MP particles/fish (reef-associated) → *S. canaliculatus* (0.3 MP particles/fish) (reef-associated) (Rochman et al. 2015). MPs ingested by fishes were found related to habitats (reef, pelagic). The study demonstrates that fishes being sold for human consumption in the open market are contaminated with MPs. A number of priority pollutants (heavy metals, PAHs, DDT, PCBs) can be adsorbed in MPs (Kibria 2017; Kibria 2018). Therefore, there is a risk that such adsorbed pollutants could move up the food chain up to human consumers (from eating contaminated seafood such as fish and shellfish).

14.2.15 Iran

14.2.15.1 MP Contaminated Marine Fish (Benthic, Demersal, Pelagic, Reef-Associated, Carnivores), Molluscs (Snails, Clams, Oysters) and Crustacean (Green Tiger Prawn)

1. Four fish species (shrimp scad, *Alepes djedaba*; orange-spotted grouper, *Epinephelus coioides*; pickhandle barracuda, *Sphyraena jello*; and bartail flathead, *Platycephalus indicus*) from the Khark Island, Persian Gulf (a semi-enclosed water body) were found contaminated with MPs (in muscles) (research was carried out in November 2015). Around 50–65% (mean 56.25%) of the MPs were fibres, and the dominant ingested MPs were of black, transparent/white, blue and red/pink colour which fish may have mistaken as food. Trends in MPs in fish muscles were highest with bartail flathead, *P. indicus* (18.50 MP parti-

cles/10 g fish muscle) (benthic, reef-associated, carnivorous) → shrimp scad, *A. djedaba* (8 MP particles/10 g fish muscle) (pelagic, reef-associated, carnivorous) → orange-spotted grouper *E. coioides* (7.75 MP particles/10 g fish muscle) (benthic, carnivorous) → pickhandle barracuda, *S. jello* (5.66 MP particles/10 g fish muscle) (pelagic, carnivorous) (Akhbarizadeh et al. 2018). MPs can enter the fish body/muscles through either food web or direct exposure via the skin, gills or gastrointestinal tract (GSI) (Jabeen et al. 2017; Wright et al. 2013; Karami et al. 2017). The MPs found in muscles of fish is of great concern to humans' health. Nonetheless, the abundance of fibres in fish muscle may have originated from the fishing industry.

2. Five species of molluscs (two clams, one oyster and two snails) from the northern part of the Persian Gulf were found contaminated with MPs (soft tissues) (research was carried out during October and November 2016). The highest concentration of MPs was found in the snail, *Thais mutabilis* (17.7 MP particles/snail) → the clams, *Amiantis umbonella* (6.9 MP particles/clams) → clams, *A. purpuratus* (6.1 MP particles/clams) → pearl oyster, *Pinctada radiata* (4.5 MP particles/oyster) → mud snail, *Cerithidea cingulata* (3.0 MP particles/snail). Around 58% of ingested MPs were fibres. The MPs were black, white, transparent, red, pink and green coloured (Naji et al. 2018). Fibres (58%) detected in molluscs could have originated from fisheries, recreational boating, laundry, domestic wastewater and other human activities (Browne et al. 2011; Devriese et al. 2015; Murphy et al. 2017). Moreover, the contamination of molluscs (seafood) with MPs is a possible route of the transfer of MPs to humans.
3. Nine commercial fish species (northern pike, *Esox lucius*; European perch, *Perca fluviatilis*; zander, *Sander lucioperca*; Prussian carp, *Carassius gibelio*; common carp, *Cyprinus carpio*; Tench, *Tinca*; common bream, *Abramis brama*; vimba bream, *Vimba vimba*; common rudd, *Scardinius erythrophthalmus*) from Anzali Wetland were found contaminated with MPs (research was carried out from May to July 2018). MP particles were found in all fish tissues including GI, muscles and gonads, and the trend was as follows: 43.52% in the GI tract, 36.78% in the muscle and 14.5% in the gonads. Black, red and blue were the dominant colours of MPs ingested by fish. The higher abundance of MP particles was recorded in omnivorous species (common carp, *C. carpio* (2.0 MP particles/fish) (benthopelagic), and Prussian carp, *C. gibelio* (1.5 MP particles/fish) (benthopelagic), than in carnivorous species (northern pike, *E. Lucius* (0.7 MP particles/fish) (pelagic), and European perch, *P. fluviatilis* 0.45 MP particles/fish) (demersal). Omnivorous fish species had around 2.26 MP particles/fish compared to carnivores' fish species having 1.10 MP particles/fish (Rasta et al. 2021). The higher abundance of MP particles observed in omnivorous fish species could be linked to their (a) feeding habits (omnivores may have ingested MPs from sediment and plant surfaces while feeding on benthic organisms (Rasta et al. 2020) and (b) the possession of a longer intestine in omnivores (compared to carnivores) allowed to retain MPs for a longer period of time in stomachs (Karachle and Stergiou 2010). Further, the occurrence of MPs in the fish muscle (36.78%) is alarming as there

is a possibility of transfer of adsorbed chemical contaminants in MPs to humans via the seafood consumption.

4. Four fish species (bartail flathead, *Platycephalus indicus* (demersal, reef-associated); greater lizardfish, *Saurida tumbil* (demersal, reef-associated); northern whiting, *Sillago sihama* (pelagic, reef-associated); the tongue sole, *Cynoglossus abbreviates* (demersal)) and crustacean (the green tiger prawn, *Penaeus semisulcatus* (benthic) from the Musa estuary and the Persian Gulf) were found contaminated with MPs in different organs (skin, muscle, gut, gills, liver) (research was carried out in June 2015). On the basis of MPs/individual, the highest MPs was found with bartail flathead, *P. indicus* (21.8 MP particles/fish) (demersal) → northern whiting, *S. sihama* (14.1 MP particles/fish) (pelagic) → greater lizardfish, *S. tumbil* (13.5 MP particles/fish) (demersal) → tongue sole, *C. abbreviates* (12 MP particles/fish) (demersal) → green tiger prawn, *P. semisulcatus* (7.8 MP particles/prawn) (benthic) (Abbasi et al. 2018). The study confirms the accumulation of MPs in different parts (skin, muscle, gut and gills, liver) of commercial fishes and prawns. The highest MP concentration was found in demersal fish (bartail flathead, *P. indicus*) which search their food in the sediment. Sediments are known as major sinks for MPs, where most of the denser MPs stay.

14.2.16 Italy

14.2.16.1 MP Contaminated Marine Fishes (Benthopelagic, Pelagic and Benthic/Demersal

1. Commercial marine fishes (European pilchard, *Sardina pilchardus*; European hake, *Merluccius*; picked dogfish, *Squalus acanthias*; red mullet, *Mullus barbatus*; and tub gurnard, *Chelidonichthys lucernus*) from the Adriatic Sea were found contaminated with MPs (research was carried out in March 2014). Around 28% of fish ingested MPs; 57% of MP ingested were fragments. Trends of MPs in stomachs were as follows: European pilchard, *S. pilchardus* (1.78 MP particles/fish) (pelagic) → red mullet, *M. barbatus* (1.57 MP particles/fish) (benthic) → European hake, *M. merluccius* (1.33 MP particles/fish) (benthopelagic) → picked dogfish, *Squalus acanthias* (1.25 MP particles/fish) (benthopelagic) → tub gurnard, *C. lucernu*s (1.0 MP particles/fish) (benthic). Around 65% of polymer ingested by fish was polyethylene (PE), followed by 19% polyethylene terephthalate (PET) (Avio et al. 2015). It can be expected that the high-density MP particles (such as PET) that sink in the sediments of the sea might have been ingested by demersal/benthic organisms, whereas low-density MP particles (that floats) on the surface (such as PE) ingested by pelagic/benthopelagic organisms (Brandao et al. 2011; Wright et al. 2013). Based on this study, it can be concluded that PET and PE consumed by fish may indicate the presence of plastic bags and bottles in the Adriatic Sea.

2. Three large pelagic fishes (swordfish, *Xiphias gladius*; bluefin tuna, *Thunnus thynnus*; albacore, *T. alalunga*) from the central Mediterranean Sea (Aeolian Islands, Strait of Messina) were found contaminated with MPs (research was carried out in 2012 and 2013). Around 18.2% of the fish ingested plastics. MPs ingested were of transparent, white, yellowish, grey, blue, red coloured. The trends in MP contamination were as follows: bluefin tuna, *T. thynnus* (32.4% of fish ingested plastics) (pelagic) → Albacore, *T. alalunga* (12.9% of fish ingested plastics) (pelagic) → swordfish, *X. gladius* (12.5% of fish ingested plastics) (pelagic) (Romeo et al. 2015). The study demonstrates the widespread presence of plastics in the Mediterranean water column. Plastic particles may have been ingested by these three large fishes during the predation or chasing of schools of small prey in shallow water where plastic fragments are more abundant (Romeo et al. 2015). The study also illustrates the contamination of large seafood items (swordfish, bluefin tuna, albacore) by MPs. It is, therefore, urgently needed to carry out further research of the possible transport of toxic chemicals via the MP contamination of seafood to other biota such as birds, whales and humans (i.e. transport of persistent, bioaccumulative and toxic (PBT) substances including DDT, PAHs, heavy metals and plastic additive toxic chemicals) (additives are added to enhance the performance of the plastics, such as phthalates, nonylphenol, bisphenol A, brominated flame retardants).

14.2.17 Japan

14.2.17.1 MP Contaminated Pelagic Marine Fish (Anchovy)

1. Japanese anchovy, *Engraulis japonicus* (pelagic-neritic) from the Tokyo Bay, was found contaminated with MPs (research was carried out in August 2015). Around 77% of the fish ingested MPs (GI); 87% and 7.3% of ingested MPs were fragments and microbeads, respectively. Japanese anchovy, *E. japonicus*, is a pelagic fish; therefore, the proportion of ingested MPs may reflect the MP pollution in the local environment (Tanaka and Takada 2016). Like this study (77% MP ingestion), there are other several places where higher MP ingestion was also recorded which might be MP pollution ‘hot spots’ (based on MP ingestion rate by fish). For example, 73% MP ingestion was recorded from the Atlantic Ocean (Wieczorek et al. 2018); 74.3% MP ingestion from Laucala Bay, Fiji (Ferreira et al. 2020); 80% MP ingestion from South Pacific Subtropical Gyre, Chile (Ory et al. 2017); 94% MP ingestion from mangrove wetlands, China (Huang et al. 2020); and 100% MP ingestion from China Sea, China (Jabeen et al. 2017) (Fig. 14.2 for global scale of MP ingestion in fish and other seafood organisms).

14.2.18 Malaysia

14.2.19 MP Contaminated Dried Eviscerated Fishes and Pelagic, Pelagic-Neritic, Benthopelagic Demersal Marine Fishes

1. Four commonly consumed dried and eviscerated fishes (fish from which internal organs such as gastrointestinal tract has been removed) were found contaminated with MPs (research was carried out in 2014). These fishes were Indian mackerel, *Rastrelliger kanagurta* (pelagic-neritic); spotty-face anchovy, *Stolephorus waitei* (pelagic-neritic); greenback mullet, *Chelon subviridis* (demersal); and belanger's croaker, *Johnius belangerii* (demersal). The highest MP loads were found in greenback mullet, *C. subviridis* (24 MP particles/fish), and lowest in spotty-face anchovy, *S. waitei* (2 MP particles/fish). Polymer polypropylene (PP) and polyethylene (PE) were found significantly higher in greenback mullet, *C. subviridis*, and belanger's croaker, *J. belangerii* (Karami et al. 2017). The study demonstrates that evisceration did not reduce or eliminate the risk of MP contamination to consumers. The eviscerated fish might have been contaminated with MPs during handling on the fishing vessels or during salting. Recently sea salt has been found contaminated with plastic fragments, fibres, filaments and films (Barboza et al. 2018). During drying processes MPs may have been translocated to fish flesh (muscle) from the alimentary tract. Since dried fishes are consumed as a whole, therefore, there is a likelihood that dried seafood consumers are more vulnerable to MP exposure and there is a need to assess the edible fish and seafood tissues for MP presence.
2. Eleven commonly consumed marine fish species collected from Seri Kembangan fish markets were investigated for MP contamination. They comprised two pelagics (torpedo scad, *Megalaspis cordyla*, and orange-spotted grouper, *Epinephelus coioides*), four pelagic-neritic (Indian mackerel, *Rastrelliger kanagurta*; Kawakawa, *Euthynnus affinis*; longtail tuna, *Thunnus tonggol*; and threefinger threadfin, *Eleutheronema tridactylum*), two benthopelagics (African catfish, *Clarias gariepinus*, and cachama, *Colossoma macropomum*) and three demersal fishes (Delagoa threadfin bream, *Nemipterus bipunctatus*; grass carp, *Ctenopharyngodon idella*; and oxeye scad, *Selar boops*). On average 44.44% of fish ingested MP; 67.4% and 16.35% of polymers (by shapes) ingested by fish were fragments and fibres, respectively; and polyethylene (88.4%) and polypropylene (9.3%) were the most dominant polymers (by types) ingested. Threefinger threadfin, *E. tridactylum*, and African catfish, *C. gariepinus*, showed high MP ingestion rates of 100% and 90%, respectively. Trends of MP ingestion were threefinger threadfin, *E. tridactylum* (10 MP particles/fish) (pelagic-neritic) → African catfish, *C. gariepinus* (9 MP particles/fish) (benthopelagic) → cachama, *C. macropomum* (5 MP particles fish) (benthopelagic) → Indian mackerel, *R. kanagurta* (5 MP particles/fish) (pelagic-

neritic) → grass carp, *C. idella* (4 MP particles/fish) (demersal) → orange-spotted grouper, *E. coioides* (4 MP particles/fish) (pelagic) → longtail tuna, *T. tonggol* (3 MP particles/fish) (pelagic-neritic) → torpedo scad, *M. cordyla* (2 MP particles/fish) (pelagic) → Delagoa threadfin bream, *N. bipunctatus* (1 MP particles/fish) (demersal) (Karbalaei et al. 2019). The variation in MP ingestion could be due to different feeding habits of the above-investigated fish species. MPs can be transferred to humans via direct fish consumption (fresh and dried) and indirectly via feeding the viscera and bones as feed to farmed livestock (fishmeal is used as an ingredient of food in poultry and farmed fish) (Hantoro et al. 2019).

14.2.20 Mediterranean Sea

14.2.20.1 MP Contaminated Marine Large Pelagic Fish

1. Pompano, *Trachurus ovatus*, a pelagic-neritic fish from the Strait of Messina (central Mediterranean Sea), was found contaminated with MPs (research was carried out in May and November 2012). Around 24.3% of fish ingested MPs (stomach analysis); ingested MPs were of different colours (hyaline/glassy, white, yellow, pinkish and blue) (Battaglia et al. 2016).

14.2.21 The Netherlands

14.2.21.1 MPs Contaminated Molluscs (Blue Mussel, Pacific Oyster, Periwinkle and Sand Hopper)

1. Five benthic species comprising common shore crab, *Carcinus maenas*; and hopper, *Gammarus* spp.; periwinkle, *Littorina littorea*; blue mussel, *Mytilus edulis*; and Pacific oyster, *Crassostrea gigas* from the Netherlands (Dutch coast, Rhine estuary, Port of Rotterdam and the coast near Ter Heijde), were investigated for MPs (research was carried out between 2012 and 2013). In crabs, no MP particles were detected. The higher MP concentrations were observed in filter feeders (oysters, mussels). Trends in MP ingestion were blue mussel, *M. edulis* (62 MP particles/g dw) → Pacific oyster, *C. gigas* (58.5 MP particles/g dw) → periwinkle, *L. littorea* (20 MP particles/g dw) → sand hopper, *Gammarus* spp. (11 MP particles/g dw) (Leslie et al. 2017). The higher MP concentrations observed in Pacific oysters and mussels reflect that people could be exposed to MPs via eating contaminated oysters and mussels.

14.2.22 North Pacific Central Gyre

14.2.22.1 MP Contaminated Mesopelagic and Epipelagic Fish Species

1. Five mesopelagic fish species (bigfin lanternfish, *Symbolophorus californiensis*; golden lanternfish, *Myctophum aurolanternatum*; Laura's lantern fish, *Loweina interrupta*; Reinhardt's lantern fish, *Hygophum reinhardtii*; Indo-Pacific snaggletooth, *Astronesthes indopacificus*) and one epipelagic fish Pacific saury (*Cololabis saira*) from the North Pacific Central Gyre were found contaminated with MPs (research was carried out during February 11 to 14, 2008). Around 35% of the fish had ingested plastics (2.1 MP particles/fish); 94% of ingested plastic were fragments. White (58.2%), clear (16.7%) and blue (11.9%) coloured plastic fragments were most abundant in fish (these coloured MPs are similar to the colours of plankton in the area). The highest number of MPs was found in Bigfin lanternfish, *S. californiensis* (7.2 MP particles/fish), and was followed by golden lanternfish, *M. aurolanternatum* (6.0 MP particles/fish) (Boerger et al. 2010). The study found that the larger fish had more pieces of plastic on average in their stomach than smaller fishes.

14.2.23 North Pacific Subtropical Gyre

14.2.23.1 MP Contaminated Mesopelagic Fish Species

1. Twenty-seven mesopelagic fish species from the North Pacific Subtropical Gyre were investigated for contamination with MPs (research was carried out in August 2009). MPs were found only in eight species (all are bathypelagic) (diaphanous hatchet fish, *Sternoptyx diaphana*; highlight hatchetfish, *Sternoptyx pseudobscura*; Pacific blackdragon, *Idiacanthus antrostomus*; Andersen's lanternfish, *Diaphus anderseni* and *Diaphus fulgens*; Bolin's lanternfish, *D. phillipsi*; Cocco's lanternfish, *Lobianchia gemellarii*; pearly lanternfish, *Myctophum nitidulum*). On average, 9.2% of the fish ingested MPs, and ingested MPs were mainly fragments (57%) and fibres (36%). Yellowish-white, blue, green, black and transparent plastics were the dominant colours of the plastics recovered from fish stomachs (Davison and Asch 2011). The MP ingestion of 9.2% by fishes of this area is much lower compared to other places studied (see Fig. 14.2).

14.2.24 North Sea

14.2.24.1 MP Contaminated Benthopelagic, Demersal and Pelagic Marine Fishes

1. Five fishes including three demersal (Atlantic cod, *Gadus morhua*; common dab, *Limanda limanda*; and European flounder, *Platichthys flesus*) and two pelagic fishes (Atlantic herring, *Clupea harengus*, and Atlantic mackerel, *Scomber scombrus*) from the North Sea were investigated for contamination with MPs (research was carried out during June–September 2013). No MPs were detected in Atlantic cod, *G. morhua*; European flounder, *P. flesus*; and Atlantic herring, *C. harengus*. On average, 6.1% of the fish ingested MPs, and fragments were major plastic particles. Trends in MP ingestion were Atlantic mackerel, *S. scombrus* (13.2% MP ingestion) → common dab, *L. limanda* (5.4% MP ingestion). Polyethylene (PE) (40%) and polyamide (PA) (22%) were the main polymers detected; clear and white were the most dominant colours of polymers in fish (Rummel et al. 2016). The higher ingestion of MPs in Atlantic mackerel, *S. scombrus* (pelagic fish), may have occurred accidentally during normal feeding activity or were mistakenly ingested MPs as prey (small fish). The detected coloured fibres probably originated from the fishing gears of commercial fisheries.
2. Seven common North Sea fish species (Atlantic mackerel, *Scomber scombrus*; Atlantic cod, *Gadus morhua*; grey gurnard, *Eutrigla gurnardus*; haddock, *Melanogrammus aeglefinus*; Atlantic herring, *Clupea harengus*; horse mackerel, *Trachurus trachurus*; and whiting, *Merlangius merlangus*) were found contaminated with MPs. On average, 2.6% of fish ingested MPs (research was carried out during January and February 2011). Trends in MP ingestion (% individual with plastics) were Atlantic cod, *G. morhua* (13% MP ingestion) (benthopelagic) → haddock, *M. aeglefinus* (6.2% MP ingestion) (demersal) → whiting, *M. merlangus* (5.7% MP ingestion) (benthopelagic) → Atlantic herring, *C. harengus* (1.4% MP ingestion) (benthopelagic) → grey gurnard, *E. gurnardus* (demersal) → horse mackerel (<1% MP ingestion) (Foekema et al. 2013). It is probable that fish having different feeding habits and diets might have caused differences in MP ingestion. For example, Atlantic herring and horse mackerel are planktivorous and collect their food primarily by filtering seawater, whereas Atlantic cod, haddock and whiting are primarily piscivorous. Therefore, MP ingestion might have occurred by coincidence or inadvertently confusing MPs as food (prey).

14.2.25 Norway

14.2.25.1 MP Contaminated Atlantic Cod Fish

1. The fish, Atlantic cod, *Gadus morhua* (benthopelagic a common and economically important marine fish in Norway), from the Norwegian coast was found contaminated with MPs. Around 5.3% of the fishes ingested MPs (indicating a low level of MP ingestion); out of the six sites investigated, MPs were found only in fishes of the two sites. Bergen City Harbour site was identified as a 'hot spot' for plastic ingestion (polyester) in the Atlantic cod, *G. morhua* (Bråte et al. 2016). The MP ingestion in cod fish (5.3%) is comparable to 2.6% of MP ingestion from the North Sea fish (Foekema et al. 2013), 4.9% of MP ingestion from the Baltic Sea (demersal and pelagic fishes) (Rummel et al. 2016) and 6.1% of MP ingestion from the North Sea (demersal and pelagic fishes) (Rummel et al. 2016) (see Fig.14.2 for MP ingestion in other fishes in the world).

14.2.26 Portugal

14.2.26.1 MP Contaminated Bathydemersal, Benthopelagic, Demersal, Pelagic-Neritic Marine Fishes and Sharks

1. Twenty-six commercial fish species (including seven benthopelagic, four bathydemersal, one bathypelagic, nine demersal and five pelagic-neritic) off the coast of Portugal was investigated for MP contamination. Out of 26 species, no MPs were detected in eight species. Around 32.7% of fish ingested MPs, and 65.8% of ingested MPs were fibres. Pelagic fish ingested more MP particles, and benthic fish ingested more fibres. MP ingestion rate was higher with benthic fishes (63.5%) compared to pelagic fishes (36.5%). Benthic fishes also ingested more fibres. The following species had the highest MP ingestion: chub mackerel, *Scomber japonicus* (0.57 MP particles/fish) (pelagic–neritic) → Atlantic mackerel, *Scomber scombrus* (0.46 MP particles/fish) (pelagic–neritic) → European hake, *Merluccius merluccius* (0.34 MP particles/fish) (demersal) → piper gurnard, *Trigla lyra* (0.26 MP particles/fish) (bathydemersal) → small-spotted catshark, *Scyliorhinus canicula* (0.39 MP particles/fish) (demersal) → bogue, *Boops boops* (0.09 MP particles/fish) (demersal) → Atlantic horse mackerel, *Trachurus trachurus* (0.07 MP particles/fish (pelagic-neritic) → blue jack mackerel, *Trachurus picturatus*) (0.03 MP particles/fish (benthopelagic) (Neves et al. 2015). Benthic fish ingested more fibres which may indicate an ample presence of fibres in the coastal sediments. These fibres may have originated from fishing activities. The synthetic fibres ingested can cause a number of problems including preventing food ingestion and blocking the functioning of different organs (Derraik 2002). The plastic ingestion has also caused internal bruising (injury) and inflammatory responses in mussels (Browne et al. 2008; von Moos et al.

2012). Furthermore, it was found that fish sampled close to the mouth of the river had the highest percentage of ingested MP. This fact can be related to the presence of high quantities of plastics in the river mouth and the closeness of the site to a highly populated area.

14.2.27 Saudi Arabia

14.2.27.1 MP Contaminated Marine Fish (Coral Reef, Demersal, Pelagic and Seagrass Habitat Species)

1. Twenty-six commercial and non-commercial fish species from four different habitats (13 coral reef-associated, 8 demersal, 2 seagrasses and 3 mesopelagic) were sampled along the Saudi Arabian coast of the Red Sea to assess MP contamination (research was carried out in 2011, 2012, 2016 and 2017). No MPs was detected in eight species; 14.6% of the fish ingested MPs; and 98% of the ingested MPs were fibres. The highest prevalence of microplastic ingestion was reported in the Rosy dwarf monocle bream, *Parascolopsis eriomma* (3 MP particles/fish) (demersal) (Baalkhuyur et al. 2018). This study reveals that mean microplastic particles (per individual fish) from demersal, seagrass and coral reef habitats were significantly higher than that in the mesopelagic habitat. It is, therefore, likely that feeding strategies (feeding habits and habitats) played a potential role in MP ingestion in demersal, seagrass, coral reef and pelagic fishes in the Red Sea.

14.2.28 Scotland

14.2.28.1 MP Contaminated Marine Fish (Coral Reef, Seagrass, Demersal and Pelagic Fish Species)

1. The uptake of MPs by seven demersal species (plaice, *Pleuronectes platessa*; flounder, *Platichthys flesus*; common dab, *Limanda limanda*; pollock, *Pollachius pollachius*; ling, *Molva molva*; halibut, *Hippoglossus hippoglossus*; megrim, *Lepidorhombus whiffiagonis*) and five pelagic fish species (blue whiting, *Micromesistius poutassou*; greater argentine, *Argentina silus*; horse mackerel, *Trachurus trachurus*; black scabbard, *Aphanopus carbo*; round nose, grenadier, *Coryphaenoides rupestris*) in the Northeast Atlantic around Scotland (East and West coast) was investigated for MP ingestion (research was carried out in 2013 and 2014). Out of 12 species, 7 species didn't ingest MP (pollock, ling, halibut, blue whiting, horse mackerel, black scabbard and round nose grenadier). On average 29.7% of the fish ingested plastics; 82.1% of MPs ingested were fibres; and the dominant-coloured polymers ingested were black (43.0%) followed by

clear (21.9%), blue (13.2%), red (11.4%) and green (9.6%). Polyamide (65.3%), polyethylene terephthalate (14.4%) and acrylic (14.4%) were the major polymer types ingested. Trends in MP ingestion were common dab, *L. limanda* (1.3 MP particles/fish) (demersal) → plaice, *P. platessa* (0.9 MP particles/fish) (demersal) → flounder, *P. flesus* (0.8 MP particles/fish) (demersal) → megrim, *L. whiffiagonis* (0.1 MP particles/fish) (demersal) → greater argentine → *A. silus* (0.1 MP particles/fish) (pelagic). Demersal fish species ingested significantly higher amounts (45%) of MPs than the pelagic species (6.7%). Similarly, coastal fish ingested significantly higher amounts (47.7%) of MPs than the offshore species (8%) (Murphy et al. 2017). The results demonstrate that both demersal and pelagic fishes have ingested MPs; and coastal fish ingested more MPs that could indicate that the coastal sites were highly polluted with plastics. In this study, fibre (82.1%) was the most dominant type of plastic found in fish. High fibre content was also reported in fish, shrimps, oysters, mussels, clams and seaweeds from across the globe (see Fig. 14.3).

2. The Norwegian lobster (a crustacean), *Nephrops norvegicus* (benthic) from the Clyde Sea (around the Isles of Cumbrae), was found contaminated with MPs (research was carried out in May–June 2009). Around 83% of Norwegian lobster, *N. norvegicus*, ingested MPs, and 83% of ingested MPs were filaments. This study shows that a high proportion of the decapod crustacean, *N. norvegicus*, contains plastic in their stomachs and that this plastic has the potential to accumulate within these crustaceans (Murray and Cowie 2011). The lobster (*N. norvegicus*) might have ingested MPs as they feed on prey (MPs may have accumulated in prey) or they ingested MPs via sediment as they feed on the fauna associated with sediments. *N. norvegicus* is omnivorous and consumes a wide variety of benthic fauna. Filaments ingested can block different organs of the lobster and provide a false satiation (no desire to eat) effects (Auman et al. 1998).

14.2.29 South Pacific Subtropical Gyre

14.2.29.1 MP Contaminated Marine Fish (Benthopelagic, Benthic and Pelagic Species)

1. Thirty-four fish species from four locations in the South Pacific Subtropical Gyre region (Auckland, Samoa, Tahiti and Rapa Nui) were investigated for MP contamination (research carried out in September 2015 and October 2016). Out of 34 species, 33 species ingested MPs. On average, 24.3% of the fish ingested MPs. Fragments were 49%, whereas fibres were 33%. Black (22%), blue (18%) and white (17%) were the dominant colours of ingested MP by fish. The major polymers ingested were polyester (28%), polyethylene (26%), rayon (17%), polypropylene (9%), polyvinyl chloride (7%), polyamide (4%), polyurethane (3%), acrylic (3%), rubber (2%) and styrene acrylonitrile copolymer. The

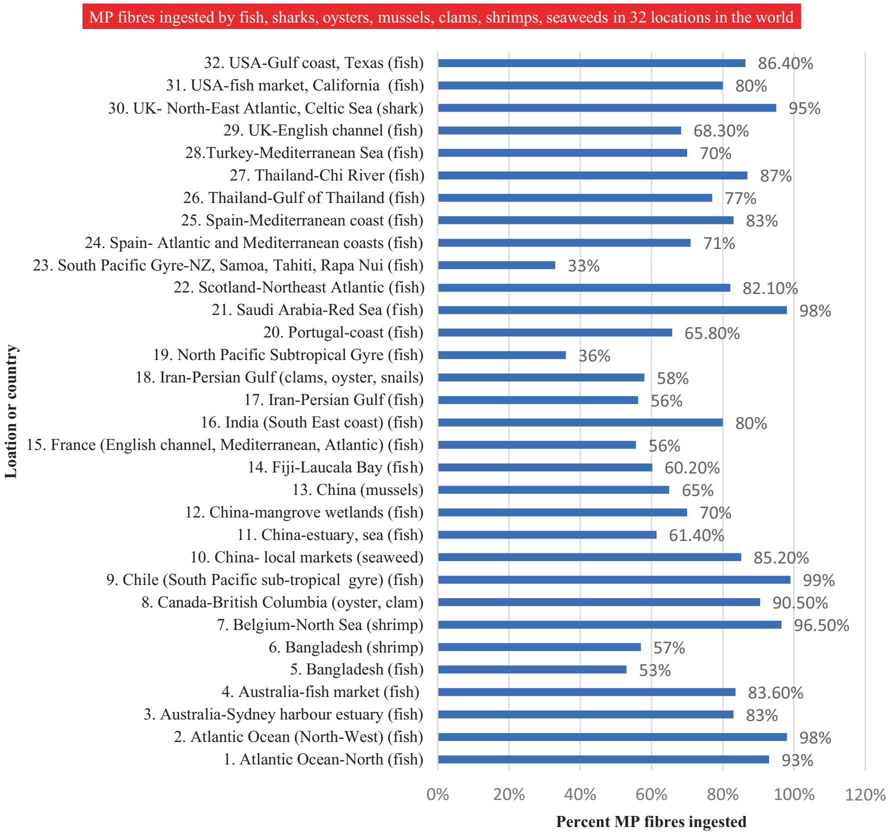

Fig. 14.3 MP fibres ingested by seafood organisms [numbers (1–32) are references: (1) Lusher et al. 2016; (2) Wieczorek et al. 2018; (3) Halstead et al. 2018; (4) Wootton et al. 2021; (5) Hossain et al. 2019; (6) Hossain et al. 2019; (7) Devriese et al. 2015; (8) Covernton et al. 2019; (9) Mizraji et al. 2017; (10) Li et al. 2020; (11) Jabeen et al. 2017; (12) Huang et al. 2020; (13) Li et al. 2016; (14) Ferreira et al. 2020; (15) Collard et al. 2017; (16) Kumar et al. 2018; (17) Akhbarizadeh et al. 2018; (18) Naji et al. 2018; (19) Davison and Asch 2011; (20) Neves et al. 2015; (21) Baalkhuyur et al. 2018; (22) Murphy et al. 2017; (23) Markic et al. 2018; (24) Bellas et al. 2016; (25) Compa et al. 2018; (26) Klangnurak and Chunniyom 2020; (27) Kasamesiri and Thaimuangphol 2020; (28) Güven et al. 2017; (29) Lusher et al. 2013; (30) Parton et al. 2020; (31) Rochman et al. 2015; (32) Peters et al. 2017]

benthopelagic fish ingested significantly more plastic (31.4% MP ingestion rate) compared to the pelagic (18% MP ingestion rate) and benthic/demersal fish (16% MP ingestion rate).The benthopelagic species feed on wider areas (bottom as well as throughout the water column) which could have caused higher exposure to plastics while searching for food. Two species had the maximum MP ingestion rate. Those were luderick/parore, *Girella tricuspidata* (70% MP ingestion rate) (benthopelagic), from New Zealand and yellow fin-tuna, *Thunnus albacares* (70% MP ingestion rate) (pelagic-oceanic), from Rapa Nu. The average

MP ingestion rates by fish at different locations were as follows: highest in Rapa Nui (49.2%), Tahiti (25%) and Samoa (17.9%) and lowest in Auckland, New Zealand (15.8%). The highest MP ingestion by fish in Rapa Nui can be related to high plastic accumulation at subtropical gyre and low availability of food (in oligotrophic waters); basically, the low food availability at Rapa Nui may have caused higher ingestion of plastics (Markic et al. 2018). There are several other places where MP ingestion by fish was much higher (see Fig. 14.2) compared to Rapa Nui (Easter Island, where 49.2% fish ingested MPs).

14.2.30 Spain

14.2.30.1 MP Contaminated Marine Fish (Demersal and Pelagic Species)

1. Three commercial and demersal fish species (lesser spotted dogfish, *Scyliorhinus canicula*; the red mullet, *Mullus barbatus*; and the European hake, *Merluccius merluccius*) from the Spanish Atlantic and Mediterranean coasts were found contaminated with MPs (research was carried out in 2014). On average 17.5% of fish ingested MPs (18.8% in red mullets, 16.7% in hakes and the lowest 15.3% in dogfish). Around 71% of the MPs ingested were fibres, and the predominant coloured MPs ingested were black (51%), red (13%) and grey (12.7%). Trends of MP ingestion were red mullet, *M. barbatus* (1.75 MP particles/fish) (demersal) → lesser spotted dogfish, *S. canicula* (1.2 MP particles/fish) (demersal) → European hake, *M. merluccius* (1.0 MP particles/fish) (demersal) (Bellas et al. 2016). The Mediterranean Sea is considered as a great accumulation zone of plastic debris. Therefore, fibres may have sourced from fishing activities, textiles and hygiene and cosmetic products. Mullet, hake and dogfish are demersal fishes that may have ingested MPs (fibres) mistakenly while feeding at the bottom of the sea.
2. Two commercially important small pelagic fish species (pilchard, *Sardina pilchardus* (pelagic-neritic), and European anchovy, *Engraulis encrasicolus* (pelagic-neritic)) from the Spanish Mediterranean coast was found contaminated with MPs (research was carried out in 2015). Around 14.8% of fish ingested MPs, and 83% of ingested MPs were fibres. The dominant colours of ingested MPs were blue (45.8%) and transparent (20.8%). Pilchard, *S. Pilchardus* (pelagic-neritic), had the highest MP ingestion (15.24% MP ingestion) compared to the European anchovy, *E. encrasicolus* (14.28% MP ingestion) (Compa et al. 2018). Microfibres were the dominant plastic ingested by fish which may have been sourced from washing machines (e.g. about 700,000 fibres could be released from an average of 6 kg wash load of acrylic fabric (Browne et al. 2011; Napper and Thompson 2016). The ingestion of MPs has reduced the growth and food consumption in the crab *Carcinus maenas* (Watts et al. 2015).
3. The semi-pelagic fish (bogues, *Boops boops*) from around the Balearic Islands (Mediterranean Sea) was found contaminated with MPs. Around 68% of the fish

ingested MPs. Filaments were the only MPs ingested (Nadal et al. 2016). Bogues could have ingested MPs while feeding on the organisms attached to plastics. MP filaments may have originated from synthetic garments (washing of clothes) through sewage outfall.

14.2.31 Tanzania

14.2.31.1 MP Contaminated Freshwater Fish (Tilapia)

1. Two commercial fish species (Nile perch, *Lates niloticus* (demersal), and Nile tilapia, *Oreochromis niloticus* (benthopelagic)) from the African Great Lakes (within Lake Victoria) were found contaminated with MPs (research was carried out in March 2015). About 20% of both species ingested MPs (55% in Nile perch and 35% in Nile tilapia). Polymer ingested by the fish includes polyethylene (PE), polypropylene (PP), polyurethane (PUR) and polyester (PES) (Biginagwa et al. 2016). Polymer ingested (PE, PP, PUR, PES) by fish might have been sourced from plastic bags, plastic bottles, packaging materials and textiles. Further, chemical pollutants (DDT, PAH, PCBs, heavy metals) adsorbed onto MPs can be transferred to humans via the consumption of these two important fishes. Since these two species (Nile perch and Nile tilapia) are heavily consumed as food by local people, therefore, in addition to assessing MP contamination in seafood, the concentration of high-risk chemicals (POPs) in different parts of fish organs (skin, gills, stomachs, intestine and muscles) should also be investigated to safeguard public health.

14.2.32 Thailand

14.2.32.1 MP Contaminated Pelagic and Demersal/Benthic Marine and Freshwater Fishes

1. Fifteen marine fish species (eight demersal and seven pelagic) from the Gulf of Thailand were investigated for MP contamination (research was carried out during June 2018–February 2019). Out of 15 species, no MPs were detected in two demersal species. On average, 8.84% of fish ingested MPs. MP ingestion was higher with pelagic fish (9.92% ingestion) compared to 7.76% ingestion in demersal. Around 77% of ingested MP was fibres which might have been sourced from fishing and tourism activities. The highest ingestion was recorded in the following species: Indian mackerel, *Rastrelliger kanagurta* (0.40 plastic/fish) (pelagic) → goldstripe sardinella, *Sardinella gibbosa* (0.29 plastic/fish) (pelagic) → yellowstriped goatfish, *Upeneus vittatus* (0.22 plastic/fish) (demersal) → bigeye scad, *Selar crumenophthalmus* (0.18 plastic/fish) (pelagic) (Klangnurak and

Chunniyom 2020). The MP ingestion rate of 8.84% in the above study is much lower compared to other fishes from Asia. Previous studies reported MP ingestion of 30% from India (Kumar et al. 2018), 44.44% ingestion from Malaysia (Karbalaei et al. 2019), 55% ingestion from Indonesia (Rochman et al. 2015) and 100% ingestion in sea fish from Coastal waters of China (Jabeen et al. 2017) (also see Fig. 14.2). Nonetheless, one of the most important reasons for high MP ingestion in Asian fish (such as in China, Indonesia, India) could be due to poor management of plastic wastes resulting in high MP pollution in waterways (rivers, oceans). China, Indonesia, the Philippines, Vietnam, Sri Lanka, Thailand, Malaysia, Bangladesh, India, Pakistan, Myanmar and North Korea are among the top 20 countries mismanaging high quantities of plastic wastes (Jambeck et al. 2015; Kibria 2017).

2. Eight freshwater fish species from Chi River were investigated for MP contamination (research was carried out in October 2018). On average 72.9% of fish ingested MPs (1.73 MP particles/fish), and 86.95% of ingested MPs were fibres (which may have originated from fishing gears such as nets and from clothing). Blue (56.9%) was the dominant colour followed by red (15.3%) and black (10.9%) MPs ingested by fishes. The higher MP ingestion was found in the following freshwater fishes: Smiths barb, *Puntioplites proctozystron* (86.7% MP ingestion (benthic, omnivore) → *Laides longibarbis* (83.3% MP ingestion) (benthic, detrivore) → *Labeo chrysophekadion* (75% MP ingestion) (benthic, detrivore) → *Mystus bocourti* (73.3% MP ingestion) (benthic, carnivore) → *Henicorhynchus siamensis* (71.4% MP ingestion) (omnivore) → *Cyclocheilichthys repasson* (70.4% MP ingestion) (benthic, omnivore) → *Labiobarbus siamensis* (50% MP ingestion) (benthic, carnivore (Kasamesiri and Thaimuangphol 2020). The average MP ingestion rate of 72.9% in the above study (Kasamesiri and Thaimuangphol 2020) is comparatively much higher compared to 8.84% MP ingestion of marine fishes from Thailand (Klangnurak and Chunniyom 2020), around 10% from the Gulf of Mexico (Phillips and Bonner 2015); 12% of fishes from French rivers (Sanchez et al. 2014); 14.8% of fishes from Spain (Compa et al. 2018); 36.5% of fishes from the English Channel, UK (Lusher et al. 2013); 37.5% of fishes from Kerala, India (Kripa et al. 2014); 15.5–37% of fishes from the Streams of Brazil (Garcia et al. 2020); and 44% of marine fishes from Malaysia (Karbalaei et al. 2019) (see also Fig. 14.2).

14.2.33 Turkey

14.2.33.1 MP Contaminated Benthopelagic, Demersal, Pelagic, Pelagic-Neritic and Reef-Associated Marine Fishes

1. Twenty-eight fish species (comprising of 7 benthopelagic, 12 demersal, 3 pelagic-neritic, 1 pelagic-oceanic, 5 reef-associated) from Turkish territorial waters of the Mediterranean Sea were investigated for MP contamination

(research was carried out in July and August 2015). Out of 28 species, MPs was not detected in three species, 58% of fish ingested MPs (2.36 MP particles/fish), 70% of ingested MP were fibres and the majority of fibres was of blue colour (50.5%). The MP ingestion was higher in pelagic fish compared to demersal. The following species had the highest MP ingestion: chub mackerel, *Scomber japonicus* (9.4 MP particles/fish) (pelagic-neritic) → golden grey mullet, *Liza aurata* (7.47 MP particles/fish) (pelagic-neritic) → dusky spinefoot, *Siganus luridus* (3.62 MP particles/fish) (reef-associated) → annular seabream, *Diplodus annularis* (benthopelagic) (2.85 MP particles/fish) → meagre, *Argyrosomus regius* (benthopelagic) (2.47 MP particles/fish) → axillary seabream, *Pagellus acarne* (benthopelagic) (2.46 MP particles/fish) (Güven et al. 2017). In this study, fish habitat types showed some effect on the MP ingestion as evidence by MP particles/fish in the stomach. For example, fish from the pelagic-neritic zone on average ingested slightly more microplastic particles than fish from other habitats.

14.2.34 The United Kingdom (UK)

14.2.34.1 MP Contaminated Pelagic and Demersal Marine Fishes and Sharks

1. Five pelagic (whiting, *Merlangius merlangus*; blue whiting, *Micromesistius poutassou*; Atlantic horse mackerel, *Trachurus trachurus*; poor cod, *Trisopterus minutus*; John Dory, *Zeus faber*) and five demersal fish species (red gurnard, *Aspitrigla cuculus*; dragonet, *Callionymus lyra*; red band fish, *Cepola macrophthalma*; solenette, *Buglossidium luteum*; thickback sole, *Microchirus variegate*) from the English Channel (coastal waters 10 km southwest of Plymouth) were found contaminated with MPs (research was carried out during June 2010 and July 2011). Around 36.5% of the fish ingested MPs (1.90 MP particles/fish), 68.3% of ingested MPs were fibres and 45.4% of MPs were black colour. Trends in MP ingestion were pelagic (38%) and demersal (35%). The MP ingestion was highest with John Dory, *Z. faber* (2.7 MP particles/fish) (benthopelagic). Rayon (57.8%) and polyamide (PA) (35.6%) were the main polymers ingested by the fishes (Lusher et al. 2013). Pelagic/planktivorous fish may have ingested more fibres because fibres were the same colour as prey items (Boerger et al. 2010) (see also Fig. 14.3 for a comparison of fibres ingested by fish and other seafood organisms worldwide). MP ingestion may have occurred while feeding or searching for food at the surface or throughout the water column (in the case of pelagic fishes) and at the bottom of the sea (in the case of demersal fishes). The identified rayon (57.8% of MPs ingested) might have been originated from sewage outfall containing clothing, furnishing, female hygiene products and nappies, whereas polyamide (35.6% of MPs ingested) might have originated from the fishing industry (Lusher et al. 2013).

2. Four shark species (small-spotted catshark, *Scyliorhinus canicula*; spiny dogfish, *Squalus acanthias*; starry smooth-hound, *Mustelus asterias*; and bull huss, *Scyliorhinus stellaris*) from the Northeast Atlantic and Celtic Sea (Cornwall) were investigated for MP contamination. Around 67.4% of sharks ingested MPs. Trends in MP ingestion were starry smooth-hound, *M. asterias* (75% MP ingestion) (demersal) → bull huss, *S. stellaris* (70% MP ingestion) (reef-associated) → small-spotted catshark, *S. canicula* (66.6% MP ingestion) (demersal) → spiny dogfish, *S. acanthias* (58% MP ingestion) (benthopelagic). About 95% of ingested MP was fibres, and the majority of fibres was blue (88%) in colour (Parton et al. 2020). The possible sources of fibres could be fishing nets and ropes, automotive tyre wear and clothing and textiles. Sharks may have ingested MPs via food (crustaceans and molluscs which have been contaminated with MPs) or direct engulfment of MPs while feeding on sediments for the targeted prey species. It is known that MPs in the marine environment ultimately sink to the seafloor and are lost in the sediment (Maes et al. 2017; Martin et al. 2017). Shark species have also been contaminated with MPs from Greece (Ionian Sea: pelagic stingray, *Pteroplatytrygon violacea*, 50% MP ingestion; Anastasopoulou et al. 2013a, b), Italy (Tyrrhenian Sea: blackmouth catshark, *Galeus melastomus*, 78.1% MP ingestion; Valente et al. 2019), Spain (Balearic Islands: blackmouth catshark, *Galeus melastomus*, 17% MP ingestion, Alomar and Deudero 2017) and the UK (Northeast Atlantic: starry smooth-hound, *Mustelus asterias*, 75% MP ingestion; Parton et al. 2020). Sharks are used as human food (meat and soup in Asia, Europe, the USA, Africa and Australia), in the industry (skin, shark liver oil) and also for medicinal purposes (vitamin A, cancer cure). Shark meat is known as 'flake', sold mostly in fish and chip shops in developed countries; they are also used for preparing fish meal as livestock feed (Haroon and Kibria 2021). Therefore, there is a risk to consumers if the shark muscle or shark liver oil gets contaminated with MPs. High-risk chemical pollutants (DDT, PCBs, PBDEs, PAHs) adsorbed in MPs can also be transferred to top consumers from eating contaminated shark muscle (meat) or shark live oil.

14.2.35 The United States of America (USA)

14.2.35.1 MP Contaminated Pelagic and Demersal Marine Fishes and Oysters

1. Twelve fish species and one oyster species from a fish market in Half Moon Bay, California, USA, were investigated for MP contamination (research was carried out during August through November 2014). Out of 12 fish species, 8 species were found ingested MPs. Twenty-five percent of fish ingested MPs, and 80% of ingested MPs were fibres. Trends in MP ingestion were jacksmelt**,** *Atherinopsis californiensis* (1.6 MP particles/fish) (pelagic-neritic)→ Pacific sanddab, *Citharichthys sordidus* (1.0 MP particle/fish) (demersal) → striped bass, *Morone*

saxatilis (0.9 MP particles/fish) (demersal) → Pacific oyster, *Crassostrea gigas* (0.6 MP particles/fish) (benthic) → Pacific anchovy, *Engraulis mordax* (0.3 MP particles/fish) (pelagic-neritic) → yellowtail rockfish, *Sebastes flavidus* (0.3 MP particles/fish) (demersal) → Chinook salmon, *Oncorhynchus tshawytscha* (0.25 MP particles/fish) (benthopelagic) → blue rockfish, (0.2 MP particles/fish) (demersal, reef-associated) → lingcod, *Ophiodon elongatus* (0.1 MP particles/fish) (demersal) (Rochman et al. 2015). MP particles ingested by fishes were found related to habitats (pelagic, demersal). Detected fibres might have originated from textiles. As the fish collected from fish markets were found contaminated by MPs, it can, therefore, be concluded that anthropogenic debris (plastics) has infiltrated marine food webs and can transfer chemical pollutants adsorbed in MPs to humans via food webs (e.g. a number of priority pollutants (heavy metals, PAHs, DDT, PCBs) can be adsorbed in microplastics; Kibria 2017; Kibria 2018). Therefore, there is a risk that such adsorbed pollutants could move up the food chain to human consumers (from eating contaminated seafood such as fish and oysters).

2. Six fish species from Texas Gulf Coast (research was carried out during September 2014 to September 2015) were investigated for contamination with MPs. Overall, 42% of fish ingested MPs (range was 26.8%–46.6%) (1.93 MP particles/fish). Around 86.4% of ingested MPs were fibres, and purple/blue (35.5%) and purple (23.0%) were the major colours of MP fibres. Trends in MP ingestion were Atlantic spadefish, *Chaetodipterus faber* (46.6% MP ingestion) (reef-associated) → pinfish, *Lagodon rhomboides* (46.5% MP ingestion) (demersal) → Atlantic croaker, *Micropogonias undulates* (45.2% MP ingestion) (demersal) → sand trout, *Cynoscion arenarius* (43.2% MP ingestion) (demersal) → Southern kingfish, *Menticirrhus americanus* (35.3% MP ingestion) (demersal) → grunt, *Orthopristis chrysoptera* (26.8% MP ingestion) (demersal) (Peters et al. 2017). The lower MP ingestion with grunt (26.8% of MP ingestion) may be related to the most selective foraging behaviour of the species, which feed almost exclusively on benthic invertebrates. In contrast, higher MP ingestions (35.2–46.6%) with other five fishes (spade fish, croaker, pin fish, trout, king fish) may be related to their generalist foraging behaviour of foraging throughout benthic and water column habitats including the inclusion of piscivory (those that eat primarily fish) as diet and suction feeding to capture prey (Peters et al. 2017).

14.2.36 Vanuatu

1. Yellow fin-tuna, *Thunnus albacores*; red claw crab, *Cardisoma carnifex*; and reef fishes (chocolate surgeonfish, *Acanthurus pyroferus*; convict surgeonfish, *A. triostegus*; lined surgeonfish, *A. lineatus*; dark capped parrotfish, *Scarus oviceps*; *Chlorusus* spp.; titan triggerfish, *Balistoides viridescens*; and *Carangidae* spp.) from the South Pacific Ocean (Efate Island) were found contaminated with MPs

(research was carried out in 2018). The highest MP contamination was found in the following orders: yellow fin-tuna, *T. albacares* (83% of fish ingested MP; 4.3 MP particles/fish) (pelagic-oceanic) → red claw crab/land crab, *C. carnifex* (57% of crab ingested MP; 1.71 MP particles/crab) (benthic) → and reef fishes (35% of reef fishes ingested MP; 2.9 MP particles/fish) (Bakir et al. 2020). The study found that edible seafood including crabs, fish and yellow fin-tuna are contaminated with MPs. MPs have adsorbed hydrophobic pollutants like organic compounds (DDT, PAHs, PCBs) and heavy metals from the surrounding water, and there is a potential for transfer of these pollutants to seafood organisms following ingestion (Bakir et al. 2014; Brennecke et al. 2016; Kibria 2018) and ultimately to the human consumer from eating contaminated seafood. The possible human health effects of consuming MP-contaminated food and water include damage of both DNA and cells and inflammation reaction (Vethaak and Legler 2021).

14.3 Conclusion

This chapter collected, collated, analysed, synthesised, interpreted and documented the last 15 years (2006–2021) of research investigations carried out on microplastic (MP) pollution impacts on seafood organisms including fish, sharks, oysters, mussels, shrimp lobsters and seaweeds covering 36 locations or countries in the world (the Atlantic Ocean, Australia, the Baltic Sea, Bangladesh, Belgium, Brazil, Canada, Chile, China, Fiji, France, the Gulf of Mexico, India, Indonesia, Iran, Italy, Japan, Malaysia, Mediterranean Sea, the Netherlands, North Pacific Central Gyre, North Pacific Subtropical Gyre, North Sea, Norway, Portugal, Saudi Arabia, Scotland, South Pacific Subtropical Gyre, Spain, Tanzania, Thailand, Turkey, the UK, the USA and Vanuatu).

Elevated/high levels of MP ingestions (compared to other species investigated by researchers; Sect. 14.2) were found in 47 seafood species (40 fishes including 39 fish and 1 shark, 3 molluscs, 3 crustaceans and 1 seaweed). Based on feeding habitats, the 40 fish species comprised 11 benthopelagic, 10 pelagic-neritic, 9 demersal/benthic, 4 pelagic-oceanic, 4 reef-associated and 2 bathypelagic (Table 14.1).

MP ingestion rate in seafood organisms varied between 3% and 100% (Fig. 14.2), and higher ingestion (> 30%) was reported from the Atlantic Ocean (fish), Australia (fish), Belgium (shrimp), Brazil (fish), Chile (fish), China (fish), China (seaweed), Fiji (fish), France (fish), India (fish), Italy (shark), Japan (fish), Malaysia (fish), North Pacific Central Gyre (fish), Portugal (fish), Scotland (lobster), South Pacific Subtropical Gyre (fish), Spain (fish), Thailand (fish), Turkey (fish), the UK (fish), the UK (shark), the USA (oyster) and Vanuatu (fish). The higher ingestion of MPs by seafood species may reflect the availability of various types of plastic particles in the environment or waterways and local pollution with MPs. Fibres were the major polymer (by shapes) ingested by seafood organisms (range 33–99%) (Fig. 14.4). Fibres in the environment can be from fishing activities (nets, lines, ropes), washing machines, textiles, sewage outfall and atmospheric fallout. Black, blue, green,

Table 14.1 List of fish, sharks, mussels, oysters and seaweeds which ingested high levels of MP and have high commercial, economical, nutritional and livelihood values

Fish *(freshwater and marine fishes and sharks)*
1. African catfish, *Clarias gariepinus* (benthopelagic) (Malaysia) (Karbalaei et al. 2019)
2. Atlantic cod, *Gadus morhua* (benthopelagic) (North Sea) (Foekema et al. 2013)
2. Atlantic cod, *Gadus morhua* (benthopelagic) (Norway) (Bråte et al. 2016)
3. Atlantic mackerel, *Scomber scombrus* (pelagic-neritic) (North Sea) (Rummel et al. 2016)
3. Atlantic mackerel, *Scomber scombrus* (pelagic-neritic) (Baltic Sea) (Rummel et al. 2016)
4. Atlantic spadefish, *Chaetodipterus faber* (reef-associated) (the USA) (Peters et al. 2017)
5. Bartail flathead, *Platycephalus indicus* (reef-associated) (Iran) (Abbasi et al. 2018)
5. Bartail flathead, *Platycephalus indicus* (reef-associated) (Iran) (Akhbarizadeh et al. 2018)
6. Bigfin lanternfish, *Symbolophorus californiensis* (pelagic-oceanic) (NPCG) (Boerger et al. 2010)
7. Bluefin tuna, *Thunnus thynnus* (pelagic-oceanic) (Italy) (Romeo et al. 2015)
8. Chub mackerel, *Scomber japonicus* (pelagic-neritic) (Portugal) (Neves et al. 2015)
8. Chub mackerel, *Scomber japonicus* (pelagic-neritic) (Turkey) (Güven et al. 2017)
9. Common carp, *Cyprinus carpio* (benthopelagic) (Iran) (Abbasi et al. 2018)
10. European hake, *Merluccius merluccius* (benthopelagic) (Italy) (Avio et al. 2015)
11. Flounder, *Platichthys flesus* (demersal) (Scotland) (Murphy et al. 2017)
12. *Girella laevifrons* (pelagic-neritic) (Chile) (Mizraji et al. 2017)
13. Glacier lanternfish, *Benthosema glaciale* (pelagic-oceanic) (Atlantic Ocean) (Lusher et al. 2016)
14. Golden lanternfish, *Myctophum aurolanternatum* (bathypelagic) (NPCG) (Boerger et al. 2010).
15. Greenback mullet, *Chelon subviridis* (demersal) (Malaysia) (Karami et al. 2017)
16. Honeycomb grouper, *Epinephelus merra* (reef-associated) (India) (Kumar et al. 2018)
17. Indian mackerel, *Rastrelliger kanagurta* (pelagic-neritic) (Thailand) (Klangnurak and Chunniyom 2020)
18. Jacksmelt, *Atherinopsis californiensis* (pelagic-neritic) (the USA) (Rochman et al. 2015)
19. Japanese anchovy, *Engraulis japonicus* (pelagic-neritic) (Japan) (Tanaka and Takada 2016)
20. John Dory, *Zeus faber* (benthopelagic) (the UK) (Boerger et al. 2010)
21. Luderick/parore, *Girella tricuspidata* (benthopelagic) (New Zealand) (Markic et al. 2018)
22. Nile perch, *Lates niloticus* (demersal) (Tanzania) (Biginagwa et al. 2016)
23. Nile tilapia, *Oreochromis niloticus* (benthopelagic) (Tanzania) (Biginagwa et al. 2016)
24. Paddle tail, *Lutjanus gibbus* (benthopelagic) (Australia) (Wootton et al. 2021)
24. Paddle tail, *Lutjanus gibbus* (benthopelagic) (Fiji) (Wootton et al. 2021)
25. Pilchard, *Sardina pilchardus* (pelagic-neritic) (Spain) (Compa et al. 2018)
26. Pink Bombay duck, *Harpadon nehereus* (benthopelagic) (Bangladesh) (Hossain et al. 2019)
27. Prussian carp, *Carassius gibelio* (benthopelagic) (Iran) (Abbasi et al. 2018)
28. Smiths barb, *Puntioplites proctozystron* (benthic) (Thailand) (Kasamesiri and Thaimuangphol 2020)
29. Red mullet, *Mullus barbatus* (demersal) (Spain) (Bellas et al. 2016)
30. Red-spotted grouper, *Epinephelus akaara* (benthic) (China) (Huang et al. 2020)
31. Rosy dwarf monocle bream, *Parascolopsis eriomma* (demersal) (Saudi Arabia) (Baalkhuyur et al. 2018)

(continued)

Table 14.1 (continued)

32. South American pilchard, *Sardinops neopilchardus* (pelagic-neritic) (Australia) (Collard et al. 2017)
33. European pilchard, *Sardina pilchardus* (pelagic-neritic) (France) (Collard et al. 2017)
34. Sea mullet, *Mugil cephalus* (benthopelagic) (Australia) (Halstead et al. 2018)
35. Shortfin scad, *Decapterus macrosoma* (reef-associated) (Indonesia) (Rochman et al. 2015)
36. Starry smooth-hound shark, *Mustelus asterias* (demersal) (the UK) (Parton et al. 2020)
37. Stout saw palate, *Serrivomer beanii* (bathypelagic) (Atlantic Ocean) (Wieczorek et al. 2018)
38. *Thamnaconus septentrionalis* (demersal) (China) (Jabeen et al. 2017)
39. Threefinger threadfin, *Eleutheronema tridactylum* (pelagic-neritic) (Malaysia) (Karbalaei et al. 2019)
40. Yellow fin-tuna, *Thunnus albacares* (pelagic-oceanic) (Vanuatu) (Bakir et al. 2020)
40. Yellow fin-tuna, *Thunnus albacares* (pelagic-oceanic) (Rapa Nu/Easter Islands) (Markic et al. 2018)
Molluscs *(oysters, mussels, snails)*
41. Blue mussel, *Mytilus* edulis (benthic) (Belgium) (De Witte et al. 2014)
41. Blue mussel, *Mytilus edulis* (benthic) (China) (Li et al. 2016)
41. Blue mussel, *Mytilus edulis* (benthic) (the Netherlands) (Leslie et al. 2017)
42. Hermit crab snail, *Thais mutabilis* (benthic) (Iran) (Naji et al. 2018)
43. Pacific oyster, *Crassostrea gigas* (benthic) (the Netherlands) (Leslie et al. 2017)
43. Pacific oyster, *Crassostrea gigas* (benthic) (the USA) (Rochman et al. 2015)
43. Pacific oyster, *Crassostrea gigas* (benthic) (Canada) (Covernton et al. 2019)
Crustaceans *(shrimps, lobsters)*
44. Common shrimp, *Crangon crangon* (benthic) (Belgium) (Devriese et al. 2015)
45. Speckled/brown shrimp, *Metapenaeus monoceros* (benthic) (Bangladesh) (Hossain et al. 2020)
46. Norway lobster, *Nephrops norvegicus* (benthic) (Norway) (Murray and Cowie 2011)
Seaweeds
47. Seaweed nori, *Pyropia* spp. (China) (Li et al. 2020)

(Note: each species has been categorised based on their feeding zones or habitats); *NPCG:* North Pacific Central Gyre

orange, purple, red and white were the various coloured polymers ingested by various seafood organisms. Ingestion of different coloured polymers may indicate that fishes have confused MPs as food (MP may often resemble copepod, algae or small fish).

MP particles ingested by seafood organisms were found highly variable and related to the feeding habits and habitats of the species. For example, pelagic/surface feeders (fish and other aquatic animals) may have ingested MPs mistakenly or confusing it (MPs) as food particles (plankton or prey). Several benthopelagic fishes have ingested higher levels of MPs (Table 14.1). This could be related to the fact that benthopelagic species feed on wider areas (in the bottom as well as throughout the water column) which could have caused higher exposure to plastics while searching for food. Higher levels of MP ingestion also occurred in demersal/benthic fish and prawns/shrimps (Table 14.1). Demersal (benthic/bottom) feeders (fish, prawns and other aquatic animals) while searching for food in seabed sediments may accidentally ingest MPs (the ocean sediments are a sink for MPs), whereas filter feeders such as mussels and oysters may inadvertently consumed/ingested/sucked MPs along with algae/microparticles during their normal filter feeding (Fig. 14.4). Several omnivores also ingested high levels of MPs (see the section under Chile, China, Iran and Thailand). Omnivores forage in a wide range of resources throughout the water column and may have ingested MPs from sediment and plant surfaces while feeding on benthic organisms. In summary, the higher MP ingestion in seafood organisms may have occurred due to a number of reasons including the following: (a) the research investigation may have been carried out in MP pollution 'hot spot' areas, (b) fish and other organisms might accidentally or mistakenly ingested MPs during their normal feeding activity (confusing MPs as prey/plankton/food) or (c) MP ingestion has occurred through trophic transfer from their prey species (as fish foods such as amphipods, copepods, decapods) which are known to have ingested microplastics (see Sect. 14.2).

Plastic ingestion can cause both physical and chemical effects in fish and other marine biota including blockage, rupture, abrasion and lesions, satiation and starvation. The ingestion of MPs can reduce the growth and food consumption in the fish.

MPs can accumulate in fish skin, gills, stomachs, liver, intestine and muscles. Hence, the consumption of MP-contaminated fresh fish, whole fish, canned fish or dried fish poses risks to humans as it may be a pathway of MP transfer to humans. Furthermore, polymers (PE, PP, PES) ingested by seafood organisms can adsorb persistent organic pollutants/priority pollutants (heavy metals, PAHs, DDT, PCBs). In addition, plastic additive chemicals (phthalates, bisphenol A, heavy metals, flame retardants) can leach out into the aquatic environment or biota which ingested MPs. Therefore, both adsorbed and additive chemicals may be transferred to humans via the consumption of MP-contaminated seafood (Fig. 14.4) (note: some of the above chemicals are highly toxic, bioaccumulative, endocrine-disrupting and carcinogenic to humans; Kibria et al. 2021). Salting of fish for preservation or drying fish with salt could bring additional contamination of seafood with MPs as sea salt has also been found contaminated with MPs. MP-contaminated fish meal used to feed poultry or farming fish/shrimps/crabs, etc. could be another indirect pathway to the exposure of MP to humans. It is therefore suggested that, as a preventive and safety measure, both farmed and wild seafoods should be depurated in clean, plastic-free seawater before human consumption. The risks of transfer of MPs to humans can further be avoided if fishes are degutted (stomachs and intestine of fishes are removed) prior to consumption. The possible human health effects of consuming MP-contaminated food and water include damage of both DNA and cells and inflammation reaction.

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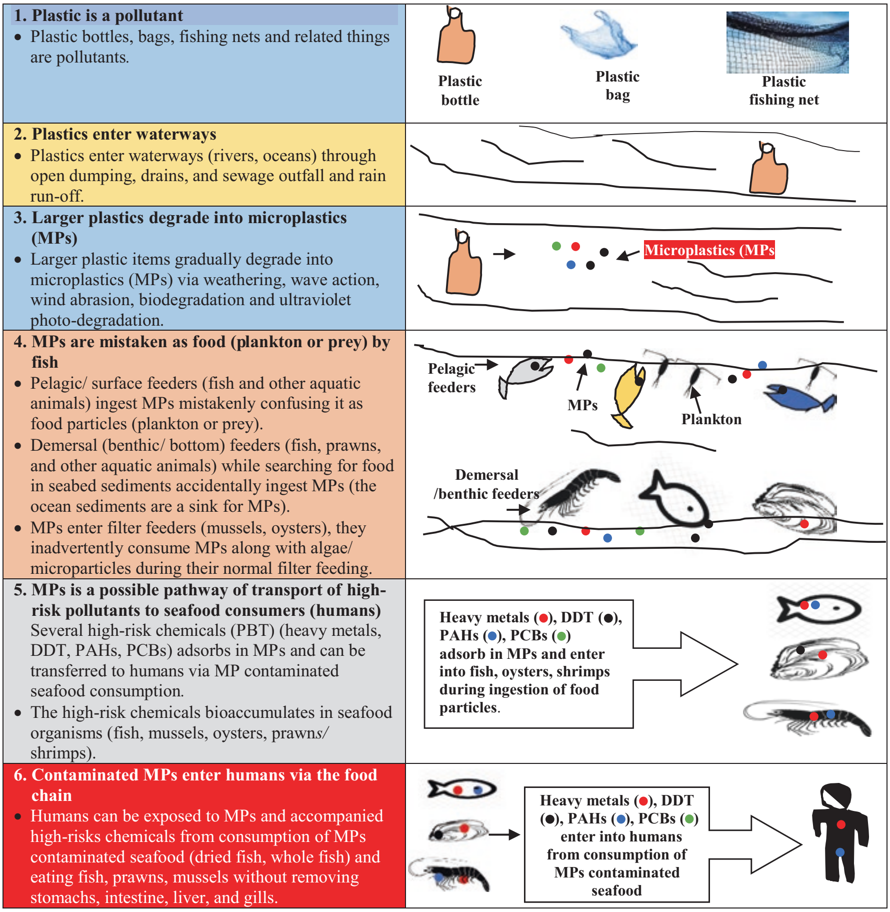

Fig. 14.4 Sources, transports, sinks and ingestion of microplastics (MPs) by pelagics/surface feeders, demersal/benthic feeders and filter feeders (mussels) in waterways (rivers, oceans) and possible transport pathways of chemical pollutants to humans [*DDT: dichlorodiphenyltrichloroethane; PAH: polycyclic aromatic hydrocarbons; PBT: persistent, bioaccumulative and toxic (PBT) chemicals; PCB: polychlorinated biphenyl*]

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Chapter 15
Impacts of Plasticizers on Riverine Ecological Integrity in Context to Sustainability Challenges

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Abstract Rapid urbanization and industrialization have introduced a variety of organic pollutants in the environmental matrices. Some of these chemicals are resistant to degradation and are termed as persistent organic pollutants (POPs). Humans and other living organisms have become highly susceptible to these environmental contaminants. For example, the semi-volatile organic compounds such as bisphenol A (BPA), phthalate esters (PAEs), and styrene monomers (SM) are extensively used in industrial production and served as intermediate complexes in different products used in daily activities. They have been classified as endocrine disruptors; therefore, exposure to these toxins creates various complications in humans and other living organisms in various ecosystems. Consequently, a balance is disturbed in the ecosystems, disintegrating the dependence of organisms on abiotic and biotic factors. This article aims to provide an overview of the commonly used plasticizers and their classification and applications, fate and transport, metabolism and mechanism of action. Subsequently, pollution load in different matrices of the riverine ecosystem has been targeted with special emphasis on the shift in ecological integrity. In addition, concerns over the use of these chemicals and their exposure have been

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highlighted that reflect a dire need to restrict and minimize their use to retrieve the ecological equilibrium.

Keywords Impact · Plasticizers · Riverine ecology · Microplastic · Sustainability

15.1 Introduction

Environmental contamination has been increased in recent years due to rapid urbanization that has introduced various natural and synthetic chemicals into different environmental matrices (Ortiz-Colón et al. 2016). Their application and discharges are directly linked with the industries and population growth (Li et al. 2019). Some of these chemicals are organic in nature that are resistant to environmental degradation. For example, bisphenol A (BPA), phthalate esters (PAEs) and alkylphenols (APs) are semi-volatile organic compounds, extensively used in industrial production and served as intermediate complexes in plastics, bindings, and filling materials. These materials are also called plasticizers due to their physicochemcal properties and ability to increase the flexibility, elasticity, durability, workability, and transparency of the plastic polymer (Fasano et al. 2012; Bocqué et al. 2016; Notardonato et al. 2019; Paluselli et al. 2019; Muobom et al. 2020; Bastiaensen et al. 2021).

Plasticizers are mainly used in the polymer industry and are described as low molecular weight additives used to manufacture several consumer products (Muobom et al. 2020). They impart commercially and mechanically conducive properties in plastic materials such as low glass temperature, increased workability, and processability (Bocqué et al. 2016). Consequently, there has been a tremendous increase in plasticizers' research and synthesis in the last few decades. Currently, most of the plasticizers are phthalate esters but it is expected that industries will eventually drift to green compounds (Vieira et al. 2011). For example, thermoplastic starch employs urea as a plasticizer (Correa et al. 2017). Over the years, research and synthesis of plasticizers have resulted in the production of a massive number of compounds tested for plasticization, approximately 30,000 (Godwin 2017). However, only 50 to 60 of them with varied chemical basis are in use commercially (Bocqué et al. 2016). Most commonly plasticized polymers include polyvinyl chloride (PVC), polyvinyl butyral (PVB), and polyvinyl acetate (PVAc). Among these categories, PVC has a highest consumption rate throughout the world. Hence, 90% plasticizers are used for the production of PVC related products, while other polymers that include minute quantities of these plasticizers are polyvinylidene chloride, PVB, polyolefins, acrylic polymers, nylon, fluoroplastics, and polyurethanes (Godwin 2017; Groh et al. 2019).

Commercially available plasticizers include a variety of chemical compounds in which ortho-phthalates are the most widely and traditionally used plasticizers. However, due to its neurotoxicity, scientist are trying to replace this hazardous plasticizer (Pecht et al. 2018; Engel et al. 2021). There are several other types of plasticizers that are in use, such as trimellitates (Godwin 2017; Altindag and Akdogan 2021), adipates (Caldeirão et al. 2021), and sebacates (Navarro et al. 2017).

The increased demand for flexible plastic polymers influenced the production of plasticizers. About 8.4 million tons of plasticizers are produced globally, and 70% of them are phthalates (Jamarani et al. 2017). According to estimates, their demand will increase to about 9.75 million tons in 2024. Moreover, a recent report suggests that the Asia Pacific region has a 59% share in global plasticizer demand, whereas Western Europe and North America have 12.1% and 11.5% shares, respectively (Pritchard 2017). With respect to global consumption pattern of plasticizers, China is far ahead with 42%, followed by countries of Western Europe 14%, the USA 11%, and India 5%. Moreover, Japan and Eastern Europe have the same percentage of consumption with 4%, South Korea and Middle East with 3.6%, Brazil and Central America with 1.6%, Africa and Thailand with 1.4%, and Indonesia with 1.3%, and other countries accounted for 5.5% cumulatively (Pritchard 2017; Plasticizers 2018). The plastic industry in Pakistan has a great history of success and its products are considered as fourth largest items of imports. The industry has surpassed all other industrial sectors in terms of development and representing annual growth rate of 15% to 17%. The annual per capita consumption in the country is estimated as 6.5 kg, quite less than average consumption of 38 kg (Ahmed 2019).

Excessive utilization of the plasticizers has been confirmed through different compartments of the environment (surface water, groundwater, air, suspended solid sediments, leachate, effluent, sludge, and biosolid) in a bundle of studies that are conducted in the USA, Canada, Japan, China, the Netherlands, Italy, Germany, South Korea, Belgium, Australia, Spain, etc. (Flint et al. 2012; Graziani et al. 2019). The highest level in South Asia has been detected that is considered due to burning of plastic waste (Fu and Kawamura 2010). These compounds do not only pollute the urban areas due to industrial manufacturing; conversely, they have enhanced the soil contamination in rural land due to excessive use of agricultural plastics (Gao and Wen 2016).

15.2 Classification of Plasticizers

There are two main categories on the basis of which plasticizers are classified, internal and external, depending upon their interaction with the polymer molecule. The former one follows the principle of copolymerization and forms primary bonds when added to any polymer. The prominent categories are vinyl acetate and vinylidene chloride. However, external plasticizers when added to polymer cannot make chemical bonds, are loosely attached via nonbonding interactions, and are considered as low volatile substances. The secondary interactions sustain them in the medium and therefore, they can be lost easily into the environment (Byun et al. 2014; Bocqué et al. 2016). External plasticizers are also classified as primary and secondary plasticizers based on some essential criteria. For a plasticizer to be classified as primary, it should be completely soluble in the polymer even at very high concentrations, not leach out quickly, and can be used solely. In other words, primary plasticizers do not require any additional additives and are responsible for making the polymer flexible. Examples include sulfates and esters of alcohols, phenols, and alkylsulfite acids (Oxoplast 2016). However, secondary plasticizers possess slow gelation, they are incompatible with the polymer, and in most cases, they are made compatible with the presence of primary plasticizers. This category is used to improve the properties and to decrease the cost of product. The eminent types are aliphatic, cycloaliphatic, and partially chlorinated paraffins (Daniels 2009; Bocqué et al. 2016).

15.3 Theories of Plasticization

There are three theories regarding the plasticizers' working mechanism: the lubricity theory, the gel theory, and the free volume theory (Bocqué et al. 2016; Muobom et al. 2020). Generally, the mechanism of action is termed plasticization, and it is accepted that plasticizers function by reducing the intermolecular interactions between the polymer molecules, thus reducing their rigidity. Consequently, desirable properties such as flexibility, durability, high processability, and workability are introduced into the polymers. The plasticizer is mixed with the plastic (polymer) until incorporated into the matrix during plasticization. This is done by simultaneously heating and thoroughly mixing until the dissolution of either the resin or the plasticizer results in a homogenous consistency. Therefore, the material obtained can be molded into desired shapes and commercial products.

15.3.1 Lubricity Theory

The lubricity theory explains this phenomenon by stating that the plasticizers diffuse into the polymer matrix or the molecule's three-dimensional structure and effectively reduce the friction between the polymer chains during heating. This results in the reduction of polymer-polymer interactions or the van der Waals forces. It can also be described as the alternating layers of polymer and plasticizer, similar to a lubricant; it decreases the friction and allows easy slippage, increasing its flexibility as mentioned in Fig. 15.1a (Daniels 2009).

15.3.2 Gel Theory

The gel theory, as described in Fig. 15.1b, employs the idea that plasticizer molecules are bonded with polymer molecules through secondary forces such as hydrogen bonding and van der Waals interactions. The plasticizer breaks the

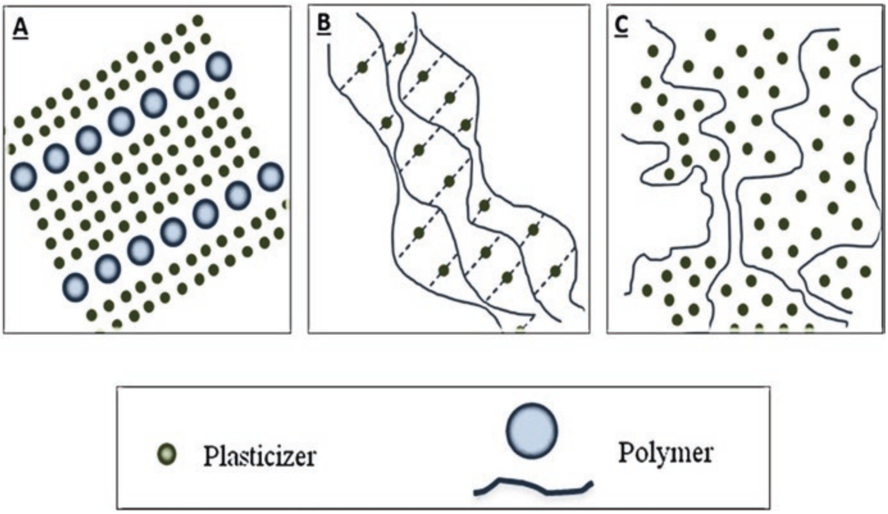

Fig. 15.1 Theories of plasticization; (**a**) Lubricity theory, (**b**) Gel Theory, (**c**) Free Volume Theory

polymer-polymer bonding and prevents their reformation by inserting into the intermolecular spaces between them, or simply plasticizers reduce the number of interacting sites between the polymers. According to this theory, the plasticized polymer is an intermediate state between solid and liquid; therefore, the only forces maintaining its structure are secondary (Chandola and Marathe 2008; Daniels 2009).

15.3.3 Free Volume Theory

Lastly, the free volume theory assumes that a rigid polymer has very little free space between adjacent polymer chains, and plasticizers increase free volume. Consequently, the polymer becomes more rubbery, and this consistency is maintained even after cooling, therefore allowing the free movement of polymer chains or creating a separating force limiting their interaction or increasing free space between them. The free volume theory is most widely accepted as it can be used to predict the behavior of most plasticizers (Chandola and Marathe 2008). The free volume is increased by adding side chains or end groups to the backbone of the plasticizer; this lowers Tg and allows polymer molecules to move freely, given in Fig. 15.1c.

15.4 Applications of Plasticizers

Plasticizers are most commonly used in the plastic industry or as binding and filling materials; however, most recently, their use has intruded almost every chemical industry. Automobiles, pharmaceuticals, medical equipment, child care products, stucco, wires and cables, concrete, and energetic materials are some of the most common consumers of plasticizers (Bocqué et al. 2016; Muobom et al. 2020). Application of PAEs in industries is based on their molecular weight. Low molecular weight PAEs are used to manufacture hygiene and personal care products, such as diethyl phthalate (DEP) and dibutyl phthalate (DBP). However, high molecular weight PAEs are used in toys, food packaging, and vinyl flooring and some PAEs are also used as intermediate lubricants (Notardonato et al. 2019).

15.5 Problems Associated with Plasticizers

The chemicals present in the plastic may be released during production, consumption, disposal, and recycling and enter into different products and environmental matrices (Groh et al. 2019) where they can resist the chemical, biological, and photolytic degradation for a longer time and can easily accumulate in the living biota. Therefore, they are also referred to as persistent organic pollutants (POPs) and are substances of international concern (Duong et al. 2014; Qureshi et al. 2016; Al-Saleh et al. 2017). Consequently, the demand for plastic-based goods has made humans and other living organisms vulnerable to its toxicity, through inhalation, ingestion, and dermal contact (Duong et al. 2014; Rowdhwal and Chen 2018; Plichta et al. 2019). Plasticizers are not only an urban concern anymore; owing to excessive use of agricultural plastic, rural areas are also affected as increasing concentrations have been detected in soils (Gao and Wen 2016). Various studies on the fate of plasticizers globally confirmed their accumulation in surface water, groundwater, suspended solids, sediments, leachate, effluent, and biomass (Groh et al. 2019).

The plasticizers have proved their toxicity and are suspected of being mutagenic and carcinogenic in nature (Qureshi et al. 2016). These compounds have gained great attention in previous years due to their interference with natural hormonal system and are called endocrine disruptors (Pallotti et al. 2020). The increased level of these plasticizers in human beings has been linked with multiple diseases and medical conditions. For instance, diabetes, cardiovascular issues, altered enzyme functions, tumors in mammary glands, and development of obesity are coupled with their exposure (Calafat et al. 2008; Vom Saal et al. 2012).

15.6 Phthalate Esters (PAEs)

Phthalate esters (PAEs) are alkyl aryl esters that are formed by double esterification of phthalic acid (aromatic dicarboxylic acid) with alcohols such as methanol and ethanol in the presence of a catalyst (Anne and Paulauskiene 2021). The application of PAEs in the industrial processing is based generally on molecular weight. Ortho-phthalate and terephthalates are the common phthalate plasticizers; however, ortho-phthalates are often referred to as phthalates due to their widespread use and production. Ortho-phthalates are classified based on their molecular weight as high molecular weight (HMW) and low molecular weight (LMW). HMW ortho-phthalates have 7–13 carbon atoms in their backbone, whereas LMW phthalates have 3–6 carbon atoms. Their working mechanism is similar to any other plasticizer elaborated earlier. The use of phthalate esters in industries also depends upon the alkyl chain; however, in most cases, 40% of final plastic product is PAEs (Wittassek et al. 2011).

The covalent bonding is not so strong in these compounds; therefore, they can migrate under thermal or mechanical stress. For instance, PVC contain a significant amount of DEHP that is a source of PAEs contamination into the environment. The exposure to sunlight and microbial activity are other threats to their degradation, and PAEs try to adsorb themselves on humus, sediments, and soil particles to get secured from degradation. Hence, their lifetime is believed to be hours in the atmosphere and months and years in soil and sediments, respectively. Moreover, phthalate esters are lipophilic and consequently, they can accumulate in the adipose tissue and be linked with several diseases Compounds with lesser molecular weights are quickly degraded as compared to higher molecular weight compounds. They can easily bioaccumulate in different organisms such as fish, invertebrates, and plants; however, their accumulation in complex organisms resulted in metabolization followed by excretion (Müllerová and Kopecký 2007; Net et al. 2015; Paluselli et al. 2019; Shu et al. 2019).

15.6.1 Metabolism of Phthalate

Phthalates can quickly be metabolized in the human body, and their metabolism takes place in two steps. In the first step, hydrolysis of diester phthalates occurs that results in the production of monoester metabolites. This process utilizes lipase and esterase enzymes and takes place in the small intestine and parenchyma. Phthalates that are classified as small and branched can be excreted through urine. In comparison, long-branch phthalates are bio-transformed by oxidation into excretable compounds and eventually excreted. The transformation of long-chain phthalates occurs in the second step, termed conjugation. This step is catalyzed by uridine 59-diphosphoglucuronyl transferase. Conjugation produces hydrophilic gluconeride conjugates that can be easily excreted. Each phthalate undergoes the same two steps before it is excreted. Several studies have been conducted on detecting phthalates in urine that have served as the basis for studies on their toxicity and exposure (Frederiksen et al. 2007; Wang et al. 2019a, b).

15.6.2 Exposure to Phthalate

PAEs display a migration phenomenon that contributes significantly to phthalate exposure along with metabolism. There are several ways of phthalate exposure; broadly, humans can be exposed by direct contact, ingestion, and inhalation (Duong et al. 2014; Rowdhwal and Chen 2018; Plichta et al. 2019). Various studies on the migration phenomena have shown that LMW phthalate can easily migrate into environmental matrices and accumulate. Blanco-Zubiaguirre et al. used food simulants to study the migration of phthalates and found that their migration to foodstuff increases from paper or cardboard by increasing temperature (Blanco-Zubiaguirre et al. 2021).

Another study investigated the occurrence of six potentially harmful PAEs in pickles packaged in polyethylene terephthalate (PET) containers. It was reported that the concentrations of DEHP and DnBP were exceeding their threshold concentrations. Moreover, the concentration of PAEs was found to depend on storage time and temperature positively; therefore, their concentration in pickled vegetables shows one of the many ways of phthalate exposure, especially from foodstuff (Cheshmazar et al. 2021). Research showed its presence in packed food; therefore, fast-food eaters are more exposed to phthalates than other community (Edwards et al. 2021). Its accumulation in the body can cause severe damage to the liver and reproductive system. For example, the samples of marine turtles, collected from the Sicilian coasts, depicted the presence of PAEs (DEHP, DBP, DEP, and BBP) in the liver and gonads at a significant concentration. In addition, tests have also showed various negative effects on different rodents (Mikula et al. 2005; Savoca et al. 2018). The exposure to pregnant mice resulted in the masculinization of a brain part that is responsible for cyclic gonadotropin release in their female offsprings (Rubin et al. 2006), and parental exposure caused delayed hatching of zebrafish (Shi et al. 2015). Furthermore, studies conducted on cows and pigeons as special groups to investigate the presence of phthalates in food, urine, and dunk have found their high concentrations. The ingestion with feed is the principal root of exposure (Tao et al. 2021).

Clothing can also be a significant source of phthalate exposure because there has been an increase in the use of phthalates in clothing. Cloths provide a primary root for direct contact with PAEs. These are semi-volatile organic compounds (SVOCs); therefore, the skin can easily be exposed to them depending upon two factors, such as clothing concentration (amount of SVOCs in cloths) and skin-to-cloth contact (Li et al. 2021). They are present in almost all matrices of environment (water, soil, and sediment), and due to multiple exposure routes, the daily intake may reach up to 70 μg/kg (Net et al. 2015).

15.6.3 Toxicity

It has already been ascertained that PAEs are hazardous. Most of the low molecular weight PAEs are pollutants and are responsible for several types of pollution, such as indoor pollution (Li et al. 2021), water, air, and soil. Consequently, concerns over effects on human health and animals have been increasing. Phthalate esters are high-risk compounds for the reproductive system; di-(2-ethylhexyl)-phthalate (DEHP) is particularly notorious for its disrupting effects on testicles and consequently affects sperm production. A recent study conducted on rats showed that DEHP decreases sperm motility, viability, maturation of nucleus, and sperm count. It also increases DNA breakage in the sperm nucleus (Karimpour et al. 2020). Studies on several other rodents have found that PAEs (DEHP) and their metabolite (MEHP) can quickly be passed from mother to infant by breastfeeding and can also move through the placenta to an infant in the uterus. Studies have shown that human breast milk contains a high concentration of hydrophobic phthalates such as DEHP and DiNP (Frederiksen et al. 2007). Therefore, it is a threat to the well-being of infants and children as it can induce several alterations in the placenta and impair the fetus's normal development. Furthermore, DEHP is also linked with several pregnancy disorders in women (Pallotti et al. 2020; Martínez-Razo et al. 2021).

15.6.4 Occurrence in Water

A recent study in China has proved DBP, DEHP, and DIBP to be major pollutants in Taihu lake water. Regardless of the season, DEHP possess high risks for fish and aquatic life. In comparison, BBP, DIBP, DHP, and DPhP were also linked to risk for aquatic life during the rainy season (Luo et al. 2021). Wang et al., in Tianjin, China, concluded that BBP, DBP, and DEHP were present in all the water samples collected; moreover, they found their highest concentration in tap water, followed by bottled water and barreled water. The study also linked the migration of phthalates and their carcinogenic effects with high-temperature storage of water in bottles (Wang et al. 2021).

Recent studies have shown that the concentrations of DEHP, DBP, and DIBP are significantly high in surface sediments and pore water. Although there are notable differences between urban distribution and nonurban distribution of PAEs, high concentrations of DEHP and DBP are linked to ecological risks in surface sediments. In contrast, DEHP, DBP, and DIBP can also be associated with ecological risks in pore water (Liu et al. 2020). Therefore, phthalate esters have been classified as one of the major categories of pollutants in all types of surface water that includes rivers, lakes, canals, glaciers, and sediments, and even bottled water (Annamalai and Vasudevan 2020; He et al. 2020; Kingsley and Witthayawirasak 2020; Li et al. 2020; Zhang et al. 2020). A study conducted on Santa Catarina River, Mexico, depicted highest concentration of PAEs in water samples with a value of 60 µg/L

(Cruz-López et al. 2020). Different compounds of phthalates have been detected in all sampling sites of Okavango Delta (Bartsch et al. 2019). Songhua River, along with its tributaries, has also represented seven types of PAEs in the water samples ranging from 1.153 to 7.867 μg/L, and DEHP remained the major contributor (Wen et al. 2018).

15.6.5 Occurrence in Sediments

According to Duong et al. (2014), phthalates were detected with high frequency in river sediments with a range between 0.13 and 0.4 mg/kg (Tiwari et al. 2016). The urban channel of Ria and coast of Campeche, Mexico, depicted PAEs in the sediments with a range of 18.292–21.702 μg/g dw (Ramirez et al. 2019). Moreover, research conducted on Spanish Iberian continental shelf also revealed PAEs concentration with a range of 16–4974 ng/g dw (León et al. 2020).

15.6.6 Occurrence in Soil

Vegetable and crop soils are one of the significant sources of phthalate pollution. The data collected in various studies show regional and geographical differences; however, both vegetable soils and crop soils require further investigation. The risk associated with soil comes from the subsequent ingestion of crops by animals and humans. DEHP is the primary soil pollutant (Zhou et al. 2021). Scientists at the Department of Botany and Microbiology, College of Science, King Saud University, Riyadh, have linked DEHP uptake in forms of biomass with a high concentration of citric acid in the soil. 20 m/kg of DEHP was accumulated in the shoots at a concentration of 200 mM citric acid (Mustafa et al. 2021). Moreover, DEHP is the major PAE in the soil and air of greenhouses for vegetable growth. It is especially dangerous in closed greenhouse environments; therefore, greenhouses should be ventilated. The high concentration of PAEs in greenhouse air and soil is linked with the plastic polymer used for building the greenhouses (Blanco-Zubiaguirre et al. 2021).

The presence of microplastics (MPs) and their accumulation in the soil is also linked with high phthalate concentration. Q. Li et al. quantified the size, type, and the number of MPs by using the direct infrared method. The scientist correlated MP concentration with PAEs and found a positive relationship in Xuzhou city, whereas no such relation was found in Shouguang city (Tao et al. 2021). Moreover, a high concentration of MPs was found in greenhouse soil as compared to non-greenhouse soil. Another study in the Huang-Huai-Hai region of China showed similar results and linked DEHP with risks associated with phthalate exposure from soil (Zhou et al. 2021).

15.7 Bisphenol A

Bisphenol A, generally written as BPA, is an organic compound belonging to a group of bisphenols and diphenylmethane. Bisphenols have several other analogues, but BPA is widely used in the preparation of epoxy resins and is considered as a precursor in the formation of plastics, mostly the polycarbonate plastic (Liao et al. 2012; Bittner et al. 2014; Boonlert-uthai et al. 2019). Just like other phenols, it is converted into esters, ethers, and salts and represents better solubility in fats and organic solvents but lesser in water (Corrales et al. 2015).

The plastic that is prepared from BPA appeared as clear, tough, lightweight, and heat resistant; hence, it is used in a variety of products. Due to widespread applications of polycarbonate plastics in the packaging of food and drinks, the utilization of BPA as an additive in commercial products has been increased. For example, water and infant bottles, medical devices, safety equipment made from plastic and plastic products, compact discs, etc. are some examples of BPA-based products. In addition to it, epoxy resin that is taken from BPA has also many applications in linings and coatings of water pipes, bottle tops, and food cans (Gibson 2017). It is also used in the production of thermal paper for sales receipt (Pivnenko et al. 2015), and it is confirmed from a study in which its highest concentration was found in the sludge of paper and textile industries (Lee et al. 2015). Furthermore, composites and dental sealants may also contain BPA (Bagley et al. 2021), and most recently, they have been used in pavement engineering (Xiang and Xiao 2020).

Polycarbonate is identified as an essential thermoplastic polymer, and due to its desirable performance, it has been used in engineering. For example, it is used in aircraft, automotive, electrical appliances, telecommunication, and data storage (Kausar 2018). Polycarbonate plastics have construction (Agarwal and Gupta 2017) and optical product applications (Mishra et al. 2018). The demand for petroleum-based polycarbonates and their subsequent harmful impacts on the environment have also resulted in the production of bio-based polycarbonates; however, there is still room for research and their potential applicability in the industry (Cui et al. 2019).

15.7.1 Metabolism of BPA

After ingestion of BPA in humans, it is easily absorbed from the gastrointestinal tract and metabolized into BPA-glucuronide in the liver and intestine. In this process, BPA is conjugated with UDP-glucuronic acid with the help of the uridine diphosphate-glucuronosyltransferase enzyme. Subsequently, all the metabolites of BPA are excreted through urine and feces (Ma et al. 2019). Based on the available literature, it can be ascertained that various biological samples such as blood (Owczarek et al. 2018), urine, hair, tissue, and amniotic fluids can be used to monitor and study BPA in the human body.

15.7.2 Exposure to BPA

Humans and other biotas can be exposed to BPA by several routes. However, it has been reported that the primary source of exposure in humans is diet. Furthermore, water, air, and dust are the secondary sources of BPA exposure. In contrast, BPA exposure via food and beverage products accounts for the majority of BPA ingestion in humans (Almeida et al. 2018). The mechanism of BPA exposure is similar to that of phthalate esters. BPA leach into food or drinks from epoxy resin coatings of canned food or polycarbonate consumer products. The leaching of BPA from polycarbonate bottles or epoxy resins has been associated with the temperature of the content or the bottle. The leaching of BPA can be significantly enhanced by contamination in the manufacturing process (Holmes et al. 2021).

Children can be exposed to BPA by breastfeeding and baby bottles. Studies have shown that breast milk contains BPA (Niu et al. 2021). BPA has been reported to have high concentrations in the sludge of paper and textile industries (Lee et al. 2015). Several studies on urine have shown high concentrations of BPA (Braun et al. 2011). Therefore, humans and other living organisms can be exposed to BPA due to their leaching into the surrounding environment, including water, sediments, and sewage networks. In addition to this, several factors such as concentration of dissolved oxygen in water, location, effluents, urbanization, and use of consumer products containing BPA are essential determinants.

Although the major route of BPA exposure is ingestion via food and water, other important routes also contribute significantly. Inhalation is considered to be the second most important contributor to BPA exposure. It is estimated that around 78% of total BPA exposure is caused by indoor pollution due to epoxy and polycarbonate products. The prolonged use of these products, such as circuit boards and adhesives, can volatilize BPA. Subsequently, BPA can be inhaled, ingested, or absorbed into human body from indoor dust (Sánchez-Piñero et al. 2020). A recent study has concluded that BPA is ubiquitous in the indoor environment; the study was conducted in kinder gardens and primary schools (Deng et al. 2018).

15.7.3 Toxicity

BPA is classified as a chemical pollutant, and it has been linked with several toxic effects on the environment, particularly aquatic ecosystems and human health. Vertebrates and invertebrates are prone to the toxic effects of BPA in water (Liu et al. 2021). In humans, BPA has been linked with inflammatory bowel diseases and other digestive complications (DeLuca et al. 2018). BPA is an endocrine disruptor and has been detected in quantities more than permit-able in body fluids such as urine, blood, and plasma (Beg and Sheikh 2020). The quantities of BPA and its endocrine-disrupting effects have also been studied in marine and aquatic ecosystems. Pengyu Chen et al. (2021) studied the endocrine-disrupting effects of BPA in association with graphene oxide and concluded that BPA causes severe endocrine disruption to zebrafish larvae (Chen et al. 2021). BPA has been reported to cause infertility; it has been shockingly evident that BPA exposure is inevitable, causing hormonal imbalances and reproductive defects in humans (Castellini et al. 2020). The toxic effects of BPA are also reported in the uterus and embryo (Nelson et al. 2020; Pivonello et al. 2020). In embryos, BPA can cause several abnormalities such as the feminization of male fetuses, variation in prostrate size, and changes in the adult sperm parameters such as motility, density, and sperm count.

Moreover, men exposed to BPA can experience erectile dysfunction and a reduction in the concentration of libido (Meli et al. 2020). BPA also hinders the normal development and functioning of the thyroid gland. A study on rats showed the effects of BPA on the production of thyroid gland hormones and their metabolism, whereas the same research indicated that ginger extract could lower the disrupting effects of BPA (Mohammed et al. 2020).

In women, BPA has been linked with changes in morphology and functions of various sex organs such as oviducts, vagina, ovaries, and uterus (Pivonello et al. 2020). Long-term exposure to BPA can cause endocrine disruption in other higher mammals. Another study showed that these adverse effects are prevalent in women who become pregnant by copulation or by in vitro fertilization (Radwan et al. 2020). Therefore, it can be ascertained that BPA is harmful for the reproductive health of animals in all the ecosystems and not just humans.

BPA and PAEs have been found to lower the levels of vitamin D in bloodstream. Vitamin D deficiency has been linked to several diseases in humans such as heart diseases, arthritis, insomnia, cancer, and weight gain. These endocrine-disrupting chemicals have been studied numerous times and each time they are linked with decreasing levels of vitamin D (Milanović et al. 2020). Moreover, studies have linked BPA exposure to obesity in children (Aktağ et al. 2021). It is also responsible for diabetes (Haq et al. 2020) and teeth deformation.

15.7.4 Occurrence in Water

The undeniable toxic effects of BPA are in part associated with water pollution. Several studies have detected and characterized BPA in surface water and associated it with adverse effects on ecosystems (Shen et al. 2021). The major source of BPA contamination is effluents generated from municipal and industrial units. Atmospheric pollution is another route of entering BPA into water bodies after precipitation. BPA and PAEs exposure from bottled water is also reported by various studies; however, commercial bottled water do not pose a serious threat to humans due to low detectable concentrations (da Silva et al. 2021). In addition to this, the fate of BPA in water bodies also depends upon its interaction with other substances such as colloidal particles, nanoparticles, and suspended particles (Shehab et al. 2020).

Pollution of water bodies due to BPA has been studied in all sorts of surface water including groundwater, aquifers, oceans, rivers, and lakes. Oceans have been sinks for BPA due to widespread plastic pollution. It has been reported that high levels of salt concentration in ocean water and sunlight can cause leaching of BPA into water from polycarbonates and resins. Although it would seem like BPA in ocean water would not be a serious problems in humans, the concentration of BPA has been increasing in the bodies of marine and freshwater organism including fish that are part of human diet. Consequently, BPA moves along various ecological ladders and eventually is consumed by humans. Even at very low concentrations, BPA has damaging effects on marine flora and fauna.

A study in Poland aimed at detecting and characterizing BPA in groundwater, surface, water, and springs concluded that the presence of BPA in groundwater is alarming and requires further investigation to assess it risks (Kmiecik et al. 2020). Another study concluded that contaminant of emerging concerns (CECs), including BPA, can be considered as environmental injustice and health disparities; the pilot study was conducted in tap water; Mt. Baldy Creek, Los Angeles; and Tijuana River (Gunasti 2020). BPA in water bodies has been detected overtime, and it can associate with other contaminants to produce secondary pollutants or disintegrate into its byproducts. A study conducted on Santa Catarina River, Mexico, depicted highest concentration of BPA in water samples with a value of 0.9 μg/L (Cruz-López et al. 2020). In tropical urban river, Malaysia, BPA was detected in colloidal and soluble phases with concentration of 1.13 and 5.52 ng/L, respectively (Nafi'Shehab et al. 2020). Similarly, in water, the concentration of BPA was detected with a range from <5.0 to 277.9 ng/dm^3 (Staniszewska et al. 2015).

15.7.5 Occurrence in Sediments

A study conducted in 2016 on the sediments of Taihu Lake represented BPA concentration of 32 ng/g dw, Liaohe River depicted 0.19–7.4 ng/g dw, and Pearl River showed 0.15–2.1 ng/g dw (Liu et al. 2021). Another study represented BPA concentration in the range of 3.94–33.2 (Wang et al. 2017). In addition, its concentration in dry weight (dw) of sediments came out with a range of 16.3–35.79 μg/kg with a mean value of 25.15 μg/kg (Tiwari et al. 2016). A study conducted in Songhua River, northeastern China, declared its concentration at 1.60–17.3 ng/g dw with a mean value of 4.90 ng/g dw (Zhang et al. 2014).

15.7.6 Occurrence in Soil

Due to high stability of BPA, its removal from soil is difficult and is easily detected, particularly in urban soil due to accumulation of BPA in soil near industrial effluents. BPA can accumulate in soil from sewage sludge that is part of the effluents from wastewater plants. In addition to this, leachate from dumping sites can also cause soil pollution. Although most studies have showed that BPA in soil dissipates quickly and might not cause a serious threat, its uptake by plants in agriculture soil can pose problems for both animals and humans (Zhang et al. 2016). Moreover, BPA converts into its metabolites within a few days (Fent et al. 2003).

15.8 Disintegration of Plasticizers in Riverine Ecosystems

There is an increasing concern about the activity and role of these phenolic estrogens in the riverine system (Li et al. 2019). In fact, it is considered as the most serious environmental issue due to their widespread distribution throughout the world and hence, proved themselves the great risk for biodiversity (Chae and An 2017; Guzzetti et al. 2018). Similarly, BPA and PAEs can cause ecological disintegration by interfering with hierarchical structure of ecosystems. Marine and freshwater ecosystems are especially prone to disintegration of these plasticizers. These compounds can pose threat to any trophic level and in that way whole ecological pyramid came under the influence of contamination. Although they have strong effect on various forms of life, including humans, their influence on aquatic ecosystem is particularly important. Studies have highlighted the disturbance of ecological integrity due to the presence of these compounds. For instance, species of fish, crustacean, and amphibian have been affected severely by butyl benzyl phthalate and BPA (Oehlmann et al. 2009).

The ecological risks and pollution levels of PAEs were studied in Poyang Lake that is considered to be largest freshwater lake in China. The study investigated six major PAEs and their seasonal variation. The evaluation of ecological risks in the lake was based on the predicted no-effect concentrations (PNECs). The PNEC values were measured as DEHP (0.0210 μg/L), DBP (2.31 μg/L), BBP (3.30 μg/L), and DEP (31.6 μg/L). It was concluded that 95% of the aquatic living organisms were affected by PAEs, especially DEHP risks which were unacceptable (Ai et al. 2021). Ecotoxicological studies in the U-Tapao canal in Southern Thailand investigated PAEs using GC-MS. The concentration of PAEs was in a range from 1.44 to 12.08 μg/L, and an average value of 4.76 μg/L. Here, the average value of PAEs concentration exceeds the recommended value set forth by the US Environmental Protection Agency (USEPA) that permits 3 μg/L of PAEs are nonhazardous for aquatic life, in which the ecological risk assessment is based on risk quotient (RQ) (Kingsley and Witthayawirasak 2020). Another study showed DEHP and DiBP have high ecological risk in the Hangzhou Bay, and it was determined that the distribution of PAEs was linked with their molecular weight (Shen et al. 2021).

Additionally, it has been reported that high concentration of PAEs can affect the growth and development of zebrafish larvae, especially exposure to DBP and BBP. This model study provides insights into how other organisms can be affected by PAEs (Pu et al. 2020). Another study for ecological risk assessment of PAEs based on hazard quotient (HQ) method that used sediments from Taihu Lake, China, found that DBP posed ecological risks moderately, whereas BIBP was linked with low risk, while DEHP, DMP, and DEP were not associated with ecological risks (Zhang et al. 2021). Furthermore, PAEs have been assessed to cause severe ecological risks to various animals in east, central, northeast, and northwest China in a study that was based on maximum cumulative ratio (MCR), RQ, and joint probability curve JPC (Kong Haoyue 2021). A study based on the investigation of PAEs for ecological risk assessment in the coastal water of Korea linked DEHP concentration with high ecological risks for benthic organisms. Similar results were obtained in Saharan rivers, where the RQ studies showed risk for algae, vertebrates, and fish populations (Ogunwole et al. 2021).

Several studies have been conducted that evaluate BPA for its toxicity in freshwater and subsequent impacts on freshwater ecosystems. The effects of BPA on microalgae were evaluated in a study based on detection and characterization in two species, *Chlamydomonas reinhardtii* and the clam *Corbicula fluminea.* It was found that the species were exposed to BPA by food and water and caused changes in biomarkers and were cytotoxic (Esperanza et al. 2020). Ecological risks caused by BPA based on hazard quotient (HQ) in various countries including China, Japan, and Korea have been reported, and high concentrations have been linked with various complications in animals (Liu et al. 2021).

Furthermore, BPA has been studied in plant ecosystems as well, and the phytotoxic effects of BPA are widespread. BPA is linked with moderate toxicity to plants mainly because of intake from soil and water. It hinders plant growth and development and affects photosynthesis and uptake of minerals (Xiao et al. 2020). Similarly, the toxic effects of BPA are also linked with exposure to microplastics (MPs), as most of them contain high levels of BPA and are linked with ecological disintegration in marine, aquatic, as well as freshwater ecosystems particularly rivers (Enyoh et al. 2020).

15.9 Worldwide Efforts to Combat the Issue

Different countries are working continuously to decrease the impacts of plastic pollution. For management strategies, European countries have incorporated these plasticizers in a pilot survey that was conducted in 17 countries, due to their emerging threat on the environment (Casteleyn et al. 2015). European Union has formulated a REACH (Registration, Evaluation, Authorisation and Restriction of Chemicals) regulation to protect the environment and human health from the harms of chemicals. This program is the most comprehensive regulation that ensures the safe use of plasticizers. This regulation has led EU to shift its demand from carcinogenic and mutagenic plasticizers (CM) to the non-CM substances, and North America has also showed the same pattern but the rest of the world is still in the process of achieving the desired results (ECHA 2013). Being a major contributor, China has introduced some legislation to decrease the sale and production of foam plastic and to restrict the use of plastic bags and has placed charges on their use. To further improve the scenario, China has enforced National Sword policy to ban different types of solid waste including plastic waste. In addition to all such regulations, it is necessary to opt proper methods of disposal and recycling to prevent improper dumping and ultimate transport of waste to water bodies (Wang et al. 2019a; b).

15.10 Conclusions

Plasticizers have become essential chemicals in most industries; their use is inevitable and so does exposure to them. Humans and animals have become highly vulnerable to the toxic effects of plasticizers. It can be ascertained that BPA and PAEs are among few of the most widely used industrial additives that have been proven to cause pollution. However, their use has not yet been completely banned because some of them have been used with restriction, notably the six phthalate esters, whereas BPA is considered as an emerging pollutant. Hence, BPA and PAEs pose serious threat to the well-being of humans. In addition, their phytotoxic and cytotoxic effects on various marine, freshwater, and plant ecosystems in terms of their investigations at various tropic levels have been widely reported. Furthermore, due to accumulation of BPA in various environmental matrices and biota, humans are vulnerable to exposure at any point by ingestion, inhalation, and dermal contact and are subject to subsequent harmful effects. Moreover, other living organisms are equally affected by exposure to BPA and PAEs. Hence, their use should be restricted and if inevitable proper disposal methods should be developed.

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Chapter 16
Consequences of Plastic Trash on Behavior and Ecology of Birds

Anum Tariq, Abdul Qadir, and Sajid Rashid Ahmad

Abstract Anthropogenic activities, particularly urbanization, have fragmented natural habitats. Generation of waste is perhaps one biggest consequence of human activities. Production, use, and disposal of plastic-based products have led to the generation of humongous quantities of waste which has its repercussions not only for the man himself but also for the other living organisms. Because of the incessant urban sprawl, various organisms are forced to adapt themselves to the man-made urban habitats, known as urban species. Birds are one such group of highly urbanized species, which by the virtue of their ability to fly, can quickly move from one place to another in search of food and nesting sites. Indiscriminate use of plastic and resulting production of plastic waste have replaced naturally available food and nesting material. This alteration in the natural environment has led to a significant impact on the ecology, and thus foraging and nesting behavior of birds. There have been several stances whereby birds have ingested plastic pieces mistaking it for food. Gut analyses of marine and terrestrial birds have revealed the prevalence of meso- as well as microplastics from direct and indirect ingestion. Likewise, in the absence of natural vegetation-based material, and because of the abundance of anthropogenic material, birds have been found to incorporate items like polyethylene bags, plastic sheets, plastic wires, yarn, etc. in their nests as a structural, defensive, or insulation component. Ingestion of plastic in birds has been linked with stomach obstruction and perforations. Likewise, plastic incorporation in nests has been associated with entanglement and even death of the nestlings by strangulation. These consequences of plastic trash on behavior and ecology of birds clearly highlights mankind's lacking diligence toward environment and the repercussions of his actions on the components of the ecosystem.

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16.1 Introduction

Birds have long been studied for their ecological behavior, diversity, and role in an ecosystem, as well as for the symbolism and spirituality that they reflect. Birds provide several ecosystem services, such as seed dispersal, pest control, and pollination (Whelan et al. 2008). Migratory birds help in nutrient flux (Murakami and Nakano 2000), and in some parts of the world, birds are hunted for sports and food (Bennet and Whitten 2003). In the evolutionary timeline, birds appeared about 165–150 million years ago, while modern humans originated way after, nearly 200,000 years ago (Galway-Witham and Stringer 2018). On the other hand, the first synthetic plastic was developed in 1907, and by the 1950s, mass production of plastic was taking place (Thompson et al. 2009). By the year 2017, annual plastic production had reached approximately 8300 MMT, 79% of which ended up in the environment, with 161 million tons being packaging waste alone (Geyer et al. 2017). Plastics are now believed to be one of the major indicators of Anthropocene as their release in the environment is associated with anthropogenic activities (Waters et al. 2016).

The very same properties that made plastic an ideal substitute for ivory also render it as an environmental nuisance because of its persistence and widespread dispersal in the environment (Ryan et al. 2009). According to one study, if no action is taken, by the year 2060, up to 265 million tons of mismanaged plastic waste will enter the environment, an amount three times more than that generated in the year 2015 (Lebreton and Andrady 2019). With an ever-increasing rise in the use and production, particularly that of the "single use" plastics which are also known as user plastics, the issue of waste disposal and associated environmental consequences are becoming a serious issue (Lusher 2015). Plastic pollution is now used to particularly refer to the plastic remains that do not serve any purpose, which can travel between land and aquatic ecosystems, and are widespread in the urban as well as rural environment (Hartmann et al. 2019).

Several sources of plastic emissions in the environment have been investigated (Chae and An 2018). For instance, domestic effluent comprising of microplastic beads from cosmetic products (Mason et al. 2016), textile fibers (Chae and An 2018), fertilizers (Nizzetto et al. 2016a) waste from dumping sites (Bläsing and Amelung 2018), microscopic plastic particles from tire wear off (Wagner et al. 2018), land application of sewage sludge from wastewater treatment plants (Nizzetto et al. 2016b) all contribute to the dispersal of plastic in terrestrial and aquatic environment. According to a study, land-based sources make up for 80% of the total plastics in the oceans, while the remaining 20% are from the marine sources themselves (Li et al. 2016), such as fishing gear and abandoned sea vessels (Macfadyen et al. 2009).

Most of the research on plastic pollution has been on marine environment, with lesser focus on freshwater (Hartmann et al. 2019), or terrestrial ecosystem (Horton et al. 2017), despite the fact that marine plastic pollution finds its source from land and freshwater pathways (Martin and Lambert 2018) in addition to the ever-growing problem of “white pollution” in the soils (Rillig 2012). Agricultural practices such as the use of plastic mulches and plastic tunnels forms another significant source of plastic in the soil, as these plastic films gradually degrade and persist in the environment for a long time (Liu et al. 2014; Steinmetz et al. 2016). Likewise, in European countries, sewage sludge is composted and applied to fertilize agricultural land, thus contributing microplastics in the soil (DEFRA 2012).

Plastics undergo slow environmental degradation by a combination of processes such as action of sunlight, oxidation, mechanical abrasion, and microbial breakdown (Ryan et al. 2009). Plastic enters in the environment in either of the two forms, viz., macroplastics and microplastics. Macroplastics are the plastics which are more than 5 mm in size, while microplastics are those which are less than 5 mm in size, and they enter the environment either directly in the form of primary microplastics, such as those used in personal care products, or as secondary microplastics which are formed from the degradation of macroplastics (Boucher and Billard 2019). Thicker plastic objects tend to persist in the environment for a longer time especially if they are protected from direct sunlight either by being hidden under water or sediments (Andrady 2003).

Plastic debris have been found to adversely affect ecological communities and thus the ecological systems within an ecosystem (Steer and Thompson 2020). The microplastics pose special threat to marine organisms by becoming a part of the food web through benthic invertebrates (Graham and Thompson 2009). The effects are aggravated by the fact that not only do these plastics have harmful additives like polychlorinated biphenyls (PCBs) present in them (Endo et al. 2005), but they also tend to act as the vector of toxic pollutants, by adsorbing them from the surrounding environment, thus increasing the concentration of toxic chemicals higher up in the food chain (Moore 2008).

The first scientific record of interaction between living organisms and plastics dates to the 1960s, when the gut analysis of prions (Harper and Fowler 1987) and Laysan albatrosses (Kenyon and Kridler 1969) revealed the presence of plastics in their stomachs. Numerous studies have reported the presence of plastic debris in a number of wildlife species (Li et al. 2016). Worldwide, more than 260 species of aquatic organisms including fish, birds (Roman et al. 2016), crustaceans (Whelan et al. 2008; Oehlmann et al. 2009; Cau et al. 2019), reptiles, and mammals (Montevecchi 1991; Jacobsen et al. 2010; Nelms et al. 2019) have been impacted by plastics, due to entanglement, ingestion, reduced fecundity, laceration, and even death (Gregory 2009). Plastic debris can affect wildlife species in different ways; for instance, the organism can get entangled in plastic objects, can ingest, or get wounded by them (Kühn et al. 2015). The interaction between the wildlife species and the plastic has been associated with the distance between the organism and the plastic material, organism’s foraging behavior, as well as its physiology (Steer and Thompson 2020).

16.2 Foraging Strategy and Plastic Ingestion

Up till now, most of the research on plastic ingestion in birds has been conducted for marine bird species (Day et al. 1985; Bester et al. 2011; Van Franeker and Law 2015; Roman et al. 2016; Caldwell et al. 2020), representing higher trophic level and sensitivity toward environmental contamination (Acampora et al. 2016). Records of plastic ingestion in birds date to as early as the 1960s when mass plastic production peaked. Leach's storm petrels were perhaps one of the first bird species in which plastic ingestion was reported, in Newfoundland, from 1962 till the 1970s (Rothstein 1973). Cases of plastic ingestion have been recorded for more than 700 marine species (Gall and Thompson 2015), among which seabirds are the most vulnerable and sensitive group with respect to plastic ingestion (Caldwell et al. 2020).

Furthermore, it is estimated that 95% of the seabirds would have ingested plastic by the year 2050 (Wilcox et al. 2015). On the other hand, very little information is available for plastic ingestion by terrestrial birds (Zhao et al. 2016). For instance, the northern fulmars have been studied as an indicator organism of the North Sea. 1300 specimens of northern fulmars were studied for the incidence of plastic particles over a period of 5 years, and each stomach was found to contain an average of 25 particles (Van Franeker 2011). These fulmars have been found to reduce up to 75% of the plastic on monthly basis by grinding it in their muscular stomachs before passing it out. This process results in accelerating the process of plastic breakdown, redistribution of 630 million microplastics in the marine environment, as well as its transportation to terrestrial habitat.

The frequency of plastic ingestion in birds has long been found to be a function of bird's foraging behavior, feeding techniques, food preferences (Cole et al. 2011), feeding habitat, exposure to plastic debris (Roman et al. 2019a; Madden and Eggermont 2020), body mass index, as well as the recency of data collection (Wilcox et al. 2015). The fact that seabirds rarely ever regurgitate indigestible items such as plastic makes them highly sensitive to plastic ingestion (Li et al. 2016). Terrestrial birds enjoy a much wider diversity of habitats (Tu et al. 2020) and diets (Sun et al. 2012). Similar to aquatic ecosystems, terrestrial environments are also plagued with the problem of plastic pollution (Ramos et al. 2015). Presence of microplastics has also been reported for freshwater bodies (Martin and Lambert 2018), thus putting terrestrial birds at an equal if not additional risk from diverse foraging habitats.

Industrial zones have particularly high levels of plastic pollution (Zbyszewski et al. 2014), hence increasing the chances of plastic ingestion among rural birds (Holland et al. 2016). For instance, plastic ingestion was investigated for two freshwater species: American black ducks and mallards of freshwater habitat in Canada (English et al. 2015). Incidence of plastic ingestion was found to be 46.1% and 6.9% for the mallards and black ducks, respectively. Plastic ingestion was found to occur during breeding season in far-off freshwater environments, and coastal winter habitats, either accidentally or by confusing the plastic particles with food during filter feeding. On the other hand, higher incidence of microplastics has been reported in the guts of terrestrial carnivorous birds, explained by secondary ingestion of plastics through prey, and the fragmentation of existing plastic particles in the gut (Zhao et al. 2016).

In terms of foraging strategies, those seabirds which obtained food through surface diving and plunging, or surface filtering and plunging, were found to be at the highest risk of plastic ingestion (Day et al. 1985; Kühn et al. 2015; Roman et al. 2019a). Those birds which feed below the water surface are less likely to ingest plastics. In terms of food preferences, those seabirds which had crustacean-based diet were much more susceptible to plastic ingestion, as compared to those having fish-based diet (Provencher et al. 2010). This is most probably because birds often mistake hard plastic items as small crustaceans (Roman et al. 2016). Certain birds feed on waste dumps as well in addition to feeding in marine habitats, and thus they are more prone to increase plastic ingestion (Kühn et al. 2015).

Likewise, foraging habitat in terms of factor of exposure density and the amount of debris to which birds are being exposed is another important driver in determining likelihood of plastic ingestion in birds (Van Franeker and Law 2015). Birds sampled from highly polluted Hawaiian Islands showed high frequency of plastic ingestion, as much as 100% in storm petrel (Youngren et al. 2018), and 11 out of 16 (68%) sampled species contained plastic in their GI tracts (Rapp et al. 2017). These studies surely do suggest that in the regions with high plastic pollution loads, birds also tend to ingest more plastic. The same can be understood by considering the other end of the spectrum: lesser plastic ingestion in relatively plastic-free environment. For instance, in case of Antarctica, it was found that the native species foraging in the area had no plastic in their guts, and cape petrels that arrived in Antarctica during breeding season also lost up to 90% of the plastic loads from their guts within a month, given their cleaner foraging habitat (Van Franeker and Law 2015).

In addition to habitat, foraging range also affects plastic uptake by the birds. This is evident from a study on flesh-footed shearwater that moves across a variety of migratory paths (Reid et al. 2013). Tracking of Eastern Pacific Australasian population revealed them to travel toward polluted Northwestern Pacific Ocean (Reid et al. 2013). On the other hand, Western Australasian population of flesh-footed shearwater migrates to comparatively less polluted Indian Ocean (Powell 2009). A comparison of debris ingestion among both populations shows higher incidence of ingestion (54.9%) as compared to the western counterpart (18.7%). This shows that even those bird species which are found to forage on debris show variability in terms of debris ingestion rate depending on their foraging habitat and range, from highly polluted to slightly polluted areas (Roman et al. 2019b).

Given the abovementioned factors, procellariids is one family which has been found to have highest rate of plastic ingestion (Colabuono et al. 2009). These birds have restricted regurgitation abilities because of their narrow gizzard (Van Franeker and Law 2015). Additionally, juvenile birds show greater amounts of plastic debris in contrast to adult birds, possibly because of "intergenerational transfer" of plastic when adult bird feeds the nestlings (Carey 2011). This intergenerational transfer of plastics is also the reason why ratio of plastics in nonbreeding birds is higher than the breeding birds, as they do not get to offload their plastic load to the offspring

(Ryan 1988). Similar findings have been reported by other studies as well (Carey 2011; Bester et al. 2011; Van Franeker et al. 2011; Rodríguez et al. 2012).

Plastic ingestion is also affected by trophic level transfer (Zhao et al. 2016; Nelms et al. 2018). A study on the quantification and identification of plastic debris in the guts of 16 species of birds of prey in Florida, USA (Carlin et al. 2020), revealed highest percentage of microplastics in the guts of ospreys and red-shouldered hawks because of direct intake as well as trophic level transfer. Red-shouldered hawk had comparatively higher incidence of microplastic ingestion, and their foraging terrestrial habitat also showed higher quantities of plastics (de Souza Machado et al. 2018). Smaller mammals, reptiles, and amphibians present in these terrestrial settings are likely to consume microplastics, which are then transferred across several trophic levels to the predator, which in this case is red-shouldered hawk. Researchers have mentioned that those top predators that consume whole prey instead of parts of it tend to have more accumulation of microplastics in their bodies (Nelms et al. 2018), which is explainable since biomagnification effect spans across trophic levels (Carlin et al. 2020).

Likewise, scavengers and surface seizing birds often mistake plastic particles for prey (Santos et al. 2016), as these lightweight plastic fragments are low weight material which tend to stay afloat (Petry and Benemann 2017). Color of these plastics is a critical factor governing birds' foraging strategy (Rapp et al. 2017), as many organisms are selective predators that prey based on visual characteristics like color or shape (Boerger et al. 2010; Caldwell et al. 2020). In most of the researches, the color of microplastics recovered from the guts of the birds reflect those present in their foraging environment (Carlin et al. 2020). Sometimes, presence of certain colored microplastics in birds' stomach is also associated with the inhalation of these airborne microplastics (Boucher and Friot 2017). Ingestion of different colors of microplastics has been reported by different studies. Clear and blue colored microplastics were reported in the GI tract of seabirds (Zhu et al. 2019; Carlin et al. 2020). On the other hand, Zhao et al. (2016) reported the presence of mid-toned particles such as blue and pinks to be more abundant (81.6%) in terrestrial birds.

Presence of similar colored microplastics has been reported for different aquatic bodies (Lefebvre et al. 2019), and birds of prey are found to ingest them either intentionally or by mistaking them for prey (Zhang et al. 2020). Presence of lighter or paler particles in the guts of the birds has been frequently reported (Santos et al. 2016), e.g., 84.2% of the total sampled freshwater bird species in Canada contained light colored debris (Holland et al. 2016), 86.5% of the storm petrels from Tern Island had ingested lighter shades of yellows and orange, also reflecting resemblance with their prey of choice (Rapp et al. 2017). Furthermore, in marine settings, the floating plastics serve as a substrate for the growth of dimethyl sulfide (DMS) emitting phytoplankton. This DMS acts as an olfactory stimulant and attracts certain procellariform species (Savoca et al. 2016).

Different studies have been conducted to analyze the health effects related to plastic ingestion in birds. Plastic ingestion has been found to be associated with endocrinal issues (Wright et al. 2013) such as delays in chick growth, reproductive cysts (Holland et al. 2016; Roman et al. 2019b), obstruction and perforation of the gastrointestinal tract (Provencher et al. 2019; Roman et al. 2019a), interference with feeding behavior, diminished metabolic rates (Derraik 2002), and changes in blood chemistry and body morphology (Lavers et al. 2019) either directly or because of plastic borne heavy metal transfer (Roman et al. 2020) to birds' feathers (Lavers et al. 2014; Lavers and Bond 2016).

16.3 Plastic Incorporation in Nests and Entanglement

Birds are the vertebrates known for the variety of nests that they build (Walsh et al. 2011). Nests are the structures that provide multitude of functions: they are linked with reproductive success of the bird – a vessel for holding eggs and subsequently bird offspring, a safe habitat against parasites, a signal to attract female bird for mating, etc. (Mainwaring et al. 2014). Structural modifications in nests, such as the incorporation of anthropogenic litter, can jeopardize the basic functions of a nest, with an ultimate effect on the survival of nestlings, eventually having adverse effects at population level (Thompson et al. 2020).

On a similar note, anthropogenic activities have produced such substantial changes in the natural environment that induce behavioral changes in animals inhabiting the same urban space (Miranda 2017). Urbanization has altered the nature and availability of nesting material as well (Seress and Liker 2015). Anthropogenic materials such as fabric, foil, plastic, metals, etc. have been observed in the nest of various bird species (Townsend and Barker 2014; O'Hanlon et al. 2017). Incorporation of anthropogenic debris in the nests of birds has become an indicator of an ever-increasing influence of man on the environment (Jagiello et al. 2019). Solid waste has become widespread in the environment (Hoornweg et al. 2013). This human generated waste is not only abundant, but it also tends to mimic natural nesting materials which is frequently used by birds (Votier et al. 2011). Despite similarities with natural material, these anthropogenic materials still require behavioral modifications and adaptations in birds, to enable them to successfully use it for nest building (Suárez-Rodríguez et al. 2013).

Different studies have explored possible reasons for the use of anthropogenic debris in nest building: unavailability of natural material because of urbanization (Lee et al. 2015), in order to increase the integrity of nest (Antczak et al. 2010), and ease of collecting lightweight and more durable debris such as plastic (Antczak et al. 2010), to decorate the nest in order to enhance female bird's devotion toward mating, to attract a mate (Polo and Veiga 2006) to ward off predators (Delhey et al. 2017), to show territorial control and dominance (Canal et al. 2016), to evade parasites, etc. (Suárez-Rodríguez et al. 2013).

Likewise, birds like black kites use white plastic in their nests (Fig. 16.1) as a phenotypic signaling strategy for several purposes (Sergio et al. 2011). Sergio et al. (2011) also found that very young and very old kites avoid using any nest decorations at all. Kites use white plastic (Fig. 12.2) to help camouflaging their eggs from predator, or to attract other kites which the nesting pair fights off to assert territorial

Fig. 16.1 Use of polyethylene bags as nesting material. (a) Nestling of black kite, (b) eggs of black kite

dominance. Incidence of presence of anthropogenic debris in marine and terrestrial birds varies considerably, but more information is available for the former and lesser for the latter group (Jagiello et al. 2018). Even for marine birds, use of plastic in nest building has not been extensively studied (O'Hanlon et al. 2017). A lot more information is available for the ingestion of plastics by the seabirds, such as gannets (Grant et al. 2018; O'Hanlon et al. 2019), albatrosses (Ryan 2015; Kühn et al. 2015), and gulls (Lenzi et al. 2016; Acampora 2017; Yorio et al. 2020), as compared to its use in nest construction. Spatial distribution of anthropogenic debris in marine and adjacent terrestrial ecosystem is largely affected by the distance to urban areas (Leite et al. 2014; Pedrotti et al. 2016). Another important factor in the accumulation of debris in marine environment is the distance to nearest rivers (Ivar do Sul and Costa 2013; Sadri and Thompson 2014), and the tide and wind direction (Walker et al. 2006).

Similarly, several colonies of northern gannets were studied in Scotland for determining the incorporation of anthropogenic litter in the nests. Data collected through observations and photographs revealed that 46% of the nests contained at least some form of debris. Fishing activities were found to be the significant contributor of plastic debris in the nests (Bond et al. 2012), with threadlike material forming 52% of the total recorded debris. Points having intense fishing activities also demonstrate higher proportion of marine debris (Unger and Harrison 2016). Also, gannets are known to use seaweed as nesting material, and use of threadlike plastics indicates that birds chose the anthropogenic material because of its similarity with the natural material (Votier et al. 2011).

Identification and quantification of plastics in birds' nests is search intensive and requires a good knowledge base of species population trends (Ryan et al. 2009), and can be used as an effective indicator of pollution, and areas vulnerable to plastic pollution (Grant et al. 2018). Plastic prevalence was investigated and compared to

a

b

Fig. 16.2 (**a**) Open waste dumping, (**b**) plastic bag in the nest of house crow present in the vicinity of waste dump

the nests of five species of seabirds, viz., herring gulls, great black-backed gulls, lesser black-backed gulls, great cormorants, and European shags, on Lady Isle (Scotland) (Thompson et al. 2020). Photographs of nests were taken and examined through "Coral Point Count with Excel extensions" (CPCe) software for identification and quantification of anthropogenic material (Dumas et al. 2009).

Overall, plastic prevalence in nests of all the examined species varied between 24% and 80%, with highest quantity of plastic debris being in the nests of shags. Major focus of the study was on herring gulls, in which case 35.6% nests had plastic, mostly comprising of sheet plastic (95%), and off-white in color (86.4%). A comparison of regurgitated plastic and the nesting material showed that the herring gulls collected plastic-based material from the local environment, because of its ease of availability in the nesting grounds, and not while foraging. This increasing intensity of plastic incorporation in nests has been associated with an increasing proximity to waste dumping sites (Witteveen et al. 2017). Nevertheless, probability of encountering debris in terrestrial nests has been found to be much higher than the marine nests (Jagiello et al. 2019). This is because in contrast to marine environment, solid waste is scattered and easily available throughout the terrestrial settings (Schuyler et al. 2016), and hence, the incidence of debris incorporation tends to increase with urbanization (Fig. 16.2).

Furthermore, within terrestrial habitats, effects of anthropogenic debris are much more pronounced in landscapes that have been modified by man, particularly urban and cultivated areas (Townsend and Barker 2014). Given the rapid expansion of urbanized areas, urban biota has gained some interest over the past few years (Chace and Walsh 2006). Nevertheless, the impacts of anthropogenic debris on urban organisms who adapt themselves to nonnatural anthropogenic resources (McKinney 2002) are mostly overlooked. Incidence and causes of nest incorporation of plastics along with other anthropogenic material have been investigated for different bird species. Incidence of plastic debris in birds' nests varies from species to species. In some cases, such as those of brown boobies, plastic was incorporated in the nests during breeding stage, at such sites where natural nesting material was less easily available (Lavers et al. 2013). In other stances, like in gulls, nest incorporation of plastics was found to be incidental, resulting from the material collected while foraging and then regurgitation in the nest (Witteveen et al. 2017). In this study, up to 67% of the observed nests contained plastic items, plastic packaging was observed in the regurgitated material, and ropes and strings were used in nest building.

A photographic study was carried out on brown noddy's nests at two islands, viz., Ducie Atoll (Southeast Pacific Ocean) and Inaccessible Island (South Atlantic Ocean), respectively. About 97% of the nests at Ducie Atoll contained mainly polypropylene plastic, followed by lids, fragments, mesh, etc. Of all these plastics, 68% items were blue-green in color. On the other hand, 41% of the nests from Inaccessible Island contained plastic, which were also predominantly blue-green ropes (56%), followed by gray-black ropes (14%). Greater incorporation of plastic in the nests at Ducie Atoll reflected higher abundance of anthropogenic and lower availability of natural vegetation-based material. Nonetheless, comparatively higher proportion of plastic was found in noddy's nests at Inaccessible Island where abundant vegetation is available. This observation showed that either the brown noddies are selective in the use of nesting material, or perhaps it was easier to collect material from nesting vicinities, brought ashore by the tidal action. Lastly, predominant use of rope and similar material suggests structural similarities between natural vegetation and anthropogenic material, as reported in case of other birds too (Witteveen et al. 2017; O'Hanlon et al. 2019).

Nesting habits of birds have been found to be characteristic of a specie's life traits which they themselves evolve over time (Martin et al. 2017). This explains for why different species show different behavior when selecting nesting site and material (Botero-Delgadillo et al. 2017). Adaption to a particular nesting material depends on the structural and nonstructural roles that it serves (Hilton et al. 2004; Schuetz 2005), and the presence of anthropogenic debris in nests has been suggested as a biological indicator of pollution (Tavares et al. 2016). For instance, 30 house sparrow (*Passer domesticus*) nests were examined for rural, suburban, and suburban settings in India (Radhamany et al. 2016). A total of 21.69% and 10.20% of anthropogenic material including plastic was found in urban and rural nests, respectively. The researchers also reported a decline in the usage of natural material along urbanization gradient. Behavioral plasticity of house sparrows (Martin & Fitzgerald 2005) helps them adapt to the use of anthropogenic material, such as plastic, in nest building.

On the other hand, in addition to environmental availability of anthropogenic nesting material, the incidence of debris incorporation in nests is sometimes found to be a function of the age of the bird in case of some species. A study on white storks (*Ciconia ciconia*) population in Western Poland showed that the likelihood of debris incorporation increased with the age of the bird (Jagiello et al. 2018). Furthermore, a strong correlation was found between the incidence of anthropogenic debris in the nest (38% plastic string) and its abundance in the surrounding environment (83%). Plastic strings are one of the most encountered anthropogenic materials in terrestrial birds' nests (Seacor et al. 2014).

In a similar way, presence of anthropogenic litter was studied in 106 nests of American crow (*Corvus brachyrhynchos*), in a rural-urban gradient setting of Sacramento Valley, USA (Townsend and Barker 2014). About 85% of the nests were found to contain anthropogenic material, predominantly synthetic ropes, twines, strings, and plastic strips. More material was found in the nests collected from agricultural settings as compared to those from urban areas, because of an abundance of plastic twines in the former environment. Presence of anthropogenic material tends to create "ecological traps" which attract bird communities to settle at a place; however, because of poor habitat quality, population viability is greatly reduced (Kokko and Sutherland 2001).

Likewise, a study was carried out on the prevalence and abundance of plastic twine in the nests of five species of neotropical birds in an orchard in Brazil (Batisteli et al. 2019). Plastic harvest bags are commonly used in orchards and acted as a source of plastic twines in this study too. Plastic was observed in 27% of the total nests, with highest prevalence in the nests of *Thamnophilus doliatus* and *Zonotrichia capensis* and highest abundance in *Thamnophilus doliatus'* nests. Other than the ease of availability in the surrounding environment, species-specific variation in the prevalence of specific anthropogenic material in birds' nest has also been associated with its structural use in nest construction (Bailey et al. 2016). Urban birds demonstrate behavioral plasticity that enables them to survive against broad environmental conditions (Abilhoa and Amorin 2017). Urbanization has been found to have profound negative effects on avian species (Clergeau et al. 2006); nevertheless, several birds are inhabiting and acclimatizing to urban environmental settings (Clergeau and Quenot 2007). Among these adaptations, one change is in the nesting behavior, such as a transition from natural to anthropogenic nesting materials (Wang et al. 2009). In fact, perhaps the most significant effect of urbanization is on the nature and availability of resources, including nesting material (Lim and Sodhi 2004). This shift in the nesting material is associated with the availability and abundance of anthropogenic material in urban environments (Luniak 2004).

This aforementioned effect of urbanization on the nesting composition was assessed for the Chinese bulbuls (*Pycnonotus sinensis*) in China (Wang et al. 2009). Variations in nesting material were recorded with respect to various land use types, viz., two urban sites and three rural sites. Streets and parks were the most urbanized centers, with highest availability of anthropogenic material (particularly plastics) and its incorporation in nests. The extent of anthropogenic material in nests was found to be correlated with the urbanization intensity. The researchers established that akin to other wildlife species (Luniak 2004), the Chinese bulbul is also evolving to adapt to urbanization, assisted with a shift in behavioral patterns (Yeh et al. 2007).

On a similar note, nesting use of anthropogenic litter has been found to be an avian response to anthropization (Jagiello et al. 2020). Impact of anthropogenic activities on nest composition of white storks was evaluated by Jagiello et al. (2020) in Madrid, Spain. Impact of anthropogenic activity was measured with the help of "Human Footprint Index (HFI)." Distance to nearest landfill site was calculated and both factors were correlated with the incidence of anthropogenic litter in white storks' nests. Anthropogenic debris was found in 57% of the total nests (28 out of 49). A decrease in the distance to nearest landfill site was accompanied with an increase in the HFI, and the corresponding incorporation of anthropogenic material in nests. Similar results have been reported by Henry et al. (2011) that as proximity to landfill sites decreased, white storks were more likely to mistake rubber bands for food and ingest them.

Six nests of gannets were observed in Grassholm, UK, for the presence of plastic (Votier et al. 2011). On an average, each nest contained 469.91 grams of plastics of different types. Plastic rope of synthetic fibers was predominant in all samples (an average of 83%), followed by netting and packaging material. Furthermore, over a duration of 8 years, 525 birds were found entangled in plastic material. Wings, legs, and feet of northern gannets got caught in plastic material, rendering the individual immobilized. Rate of entanglement was higher in breeding birds, which brought nesting material (plastic ropes, nets, etc.) from marine environment, mistaking them for marine algae and other natural material (Nelson 2002).

Likewise, a limited number of studies are available for plastics in the nests of freshwater birds (Jagiello et al. 2018). In a study in Santa Fe (Argentina), 20 nest samples of greater thornbirds were taken to determine prevalence of plastics (Blettler et al. 2020). A strong correlation was found between plastic present in the environment and the percentage of plastic in the nests. Nests present near open dumps had a higher percentage of plastic, as much as 95% of the total nest weight, as compared to those from relatively undisturbed areas. Softer plastics such as polyester insulation were being used as nest bedding (41%). Incorporation of plastic in nests has been frequently found to be a function of its availability, vicinity, and abundance in the local environment (Reynolds et al. 2016).

Avian species incorporate anthropogenic material in their nests wherever anthropization is involved. Most of the studies have shown higher ingestion and nest incorporation of plastics by the birds in urban settings (Townsend and Barker 2014; Radhamany et al. 2016; Jagiello et al. 2019). Nevertheless, rural areas also provide unnatural sources of nesting material. One such finding was observed in Yellowstone River, Montana, where plastic twines were reported in the nests of ospreys (Seacor et al. 2014). Prevalence of polypropylene twines was determined in 38 nests along low, medium, and high road density. Highest proportion (63.2%) of baling twine was observed in low road density, and lowest proportion (33.3%) was found in high road density area. This transition from rural to urban setting showed that more baling twine was available in the rural regions as compared to urban areas. Presence of plastic twines and strings in nests has been strongly correlated with its +availability in the surrounding areas (Bond et al. 2012).

Presence of plastic debris in the nests has been associated with a few positive and much more negative effects. For example, certain plastic materials like polyesters have high thermal insulation properties (Tilioua et al. 2016), because of which birds incorporate them in the nest for the protection of the chicks. Nevertheless, presence of such unnatural material in the nests can lead to drastic temperatures which can affect the developing embryos, thus affecting the survivals of birds. Smaller plastic particles can be ingested, or synthetic fibers can be inhaled by the chicks present in the nest (Blettler et al. 2020). In addition, entanglement is one of the most notable consequence of plastic in the nests, as its impacts are far more pronounced in terms of injuries, suffocation, immobility, etc. (Kühn et al. 2015). Observing the trends of plastic incorporation patterns in nest can serve as an important means for measuring plastic entanglement risks (Hartwig et al. 2007).

In marine settings, a small fraction of plastic material, e.g., fishing gear, is responsible for most of the bird entanglement cases (Ryan 2018). Fishing lines usually affect birds in aquatic settings, and land birds are affected by fishing line only when they forage in such an area where fishing line is already entangled in vegetation, or when it is used as nesting material. Birds caught in plastic are often rendered vulnerable to the predator as they have already spent a good amount of energy on trying to break loose from the entangling debris (Sazima and D'Angelo 2015). Moreover, entanglement by plastic incorporated in nests is usually more common in the nestlings that are weak and not able to untangle themselves (Votier et al. 2011). For instance, 3.3% of the 120 observed nestlings of ospreys were found entangled in the baling twine (Seacor et al. 2014). Likewise, out of 195 nestlings of *Corvus brachyrhynchos*, 5.6% were found entangled in the anthropogenic material, marking entanglement as an additional environmental stressor for avian community (Townsend and Barker 2014). In South Africa, chicks of bank cormorant were found dead in their nest, strangulated by fishing line (Robinson et al. 2012).

Composition of nest and its size has a critical role impact on the heat loss from nest. This microclimate of nest in turn influences the clutch size, nestling development, and growth (Akresh et al. 2017). Plastics are bad conductors and good insulators of heat. In other words, their presence in the nest prevents heat loss. Certain plastic materials tend to possess greater heat insulation properties (Suárez-Rodríguez et al. 2013). Nest insulation affects incubation costs (energy) as well as fecundity (Windsor et al. 2013). In some stances, thermal insulation capacity has been linked with chick survival (Tilioua et al. 2016). However, those plastics which have somewhat higher heat conductivity have been linked with adverse effects on embryo survival (Blettler et al. 2020). Likewise, in marine settings as well, presence of plastic is found to influence egg incubation temperature. Plastic fragments in beach sediments tend to change sediment properties such as heat transfer properties. Beaches that are contaminated with plastic fragments warm up comparatively slower, thus affecting those species which nest or lay eggs in sand (Carson et al. 2011), such as terns and snowy plovers.

16.4 Conclusion

In conclusion, birds are among high-risk individuals that are being affected by plastic pollution. Interaction between birds and plastic began right when mass production of plastic came into action. Birds occupy natural habitats, but with the encroachment of anthropogenic activities, even the most pristine environments are no longer undisturbed. Because of their generalized diets, and ability to fly, birds have adapted themselves to a broad set of urban settings, including plastic pollution. In the absence of natural material, birds make use of trash material to build their nests. Oftentimes, they ingest plastic material either intentionally or by confusing them for food items. Those living in the vicinity or waste dumping sites are particularly exposed to plastic ingestion and its use in nests. Trophic level transfer and offloading of microplastics from parent bird to nestlings and gastrointestinal, reproductive, and developmental disorders caused by plastic ingestion have also been demonstrated in various studies. Capping it all, plastic trash has affected nesting and feeding behavior and ecology of birds, requiring for more research and immediate action by the stakeholders.

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Chapter 17
Risk Assessment of Microplastic Pollution

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Abstract Microplastics are stubborn pollutants that are growing in attention in the twenty-first century. These pollutants are ubiquitous in the entire environment. The endurance of microplastics poses it is greatly resilient to decay and enables it to reach into the natural environment. Owing to its tiny nature, microplastics can be accessed readily and subsequently transported via the food web by many species

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from marine, freshwater, and terrestrial ecosystems. The ingestion of microplastics in the body tissue of marine and freshwater creatures, as well as terrestrial creatures and plants, causes severe biochemical consequences. Indirect intake of microplastics has also the prospects to produce genetic changes in human beings that might induce sterility, obesity, and chronic cancer. Due to the risk of microplastic pollution to the entire ecosystem and mitigating the environmental pollution risk, overuse of plastics and plastic-derived products must be controlled, and laws and strategies to restrict the origins of plastic debris must be implemented. In addition, in-depth research is required to fully quantify the adverse impact and environmental concerns of microplastics in marine, freshwater, and terrestrial ecosystems.

Keywords Microplastic · Terrestrial ecosystem · Environmental degradation · Biodegradable plastic · Microbial transportation

17.1 Introduction

Microplastic (MP), a pint-size particle of plastic with less than 5 mm in length (Andrady 2011), is found in the environment due to plastic pollution. MP can be found in a wide range of items, including cosmetics products, synthetic items of clothing, and plastic pouches and containers (Guerranti et al. 2019). But MPs are mainly categorized into two forms: primary and secondary (Zhang et al. 2020). Primary MPs consist of microbeads found in personal care products, plastic pellets, and plastic fibers, while secondary MPs are formed with weathering of larger plastics. There is another category originating from the human consumption of objects, which has recently been categorized. This category between primary and secondary MPs is directly derived from human use (viz., tires and synthetic fabrics for machine washing), so other experts contend that they should be categorized as primary MP. Hence, MPs are not biodegradable; once those primary and secondary MPs are accumulated in the environment, it persists forever and causes MP pollution.

MP pollution is wreaking havoc in a wide range of habitats, including marine (Cózar et al. 2014; Desforges et al. 2014; Sharma and Chatterjee 2017), freshwater (Di and Wang 2018; Xu et al. 2018), and terrestrial ecosystems (Wahl et al. 2021; Xu et al. 2020), as plastic production and its daily necessities are rapidly increasing since 1950 (Thompson et al. 2009). By 2015, around 6300 Mt. of plastic garbage was produced, and approximately 9%, 12%, and 79% were being reprocessed, devastated, and deposited (in the natural ecosystem), respectively (Geyer et al. 2017). Every year, around 300 million tons of microplastics move into the environment. Some estimation evaluated that 5.25 trillion marine plastic wastes (92% of marine wastes) belong to MP (Auta et al. 2017). However, only 1% of total MP is directly exposed to the marine ecosystem (ME), and the remainder is intermingled through terrestrial ecosystem (TE) and freshwater ecosystem (FE) (van Sebille et al. 2015). As MPs in the ME have varying volatility due to their different densities, they are omnipresent in nature, allowing marine creatures to grasp them at different depths.

Along with the marine environment, attention on the FE for MP pollution has also started to take place (Sarijan et al. 2021; Wagner et al. 2014). Recent studies determined the presence of MP in FEs, including estuary, surface waters, and sediments of rivers, lakes, and dams (Biginagwa et al. 2016; Castañeda et al. 2014; Zbyszewski and Corcoran 2011). Although the data of MP regarding the FE is not rich like the ME, scientists assumed that the pollution and risk level would not be negligible.

Moreover, there is also a close link between MP pollution and the TE (van Sebille et al. 2015) because land-based activities are the source of 98% of primary MPs (Boucher and Friot 2017). A small number of studies related to the abundance of MP in the TE have been carried out (Chen et al. 2020; Kumar et al. 2020; Wright et al. 2020). So, it is essential to reveal the knowledge deficit between MP pollution and risk assessment in the TE.

It's worth noting that understanding the presence and pollution risk of MP in various ecosystems (marine, freshwater, terrestrial) is critical for mitigating its harmful effects on the environment. Several investigations have evaluated that MP is present in diverse natural habitats as a result of various terrestrial sources, viz., sewage sludge, industries, wastewater treatment, washing, agroecological practices, road, and fishing (An et al. 2020; Dris et al. 2015). Therefore, this chapter intends to (1) summarize the sources and (2) the abundance and distribution and (3) overview the risk of MP pollution of MP in different ecosystems and (4) the future perspectives to mitigate the present research gap briefly.

17.2 Microplastic Pollution in the Marine Ecosystem

17.2.1 Sources of MP in Marine Ecosystem

Except for 20% of entire plastic waste in the ME that originates from terrestrial habitats, plastic waste enters the ME mostly through household, commercial, and maritime practices (Derraik 2002) (Fig. 17.1). Industry-based sources of MP release in the ME include the manufacture of plastic goods from the leftover of profitable industry (Lechner et al. 2014; Sadri and Thompson 2014), the dumping of microscopic plastic particles and inapplicable powdery resin during artificial air compressing (Claessens et al. 2011), marine recreation, commercial fishing, and aquaculture. According to Moore (2008), ME is notably polluted by the above practices, and derived MP from those sources get into the ME through sewage water and freshwater sources. With this concern, the waste produce by ships and insincere management of fishing tools are also not deniable (Claessens et al. 2013; Desforges et al. 2014). Other sources of MP in ME are beauty care products, toiletries products, and a range of cleansing goods, and they reach the aquatic environment via residential and industrial sewage lines as domestic pollutants (Carr et al. 2016; Duis and Coors 2016; Fendall and Sewell 2009). Furthermore, natural disasters like floods, strong winds, and cyclones play a role in accelerating the movement of

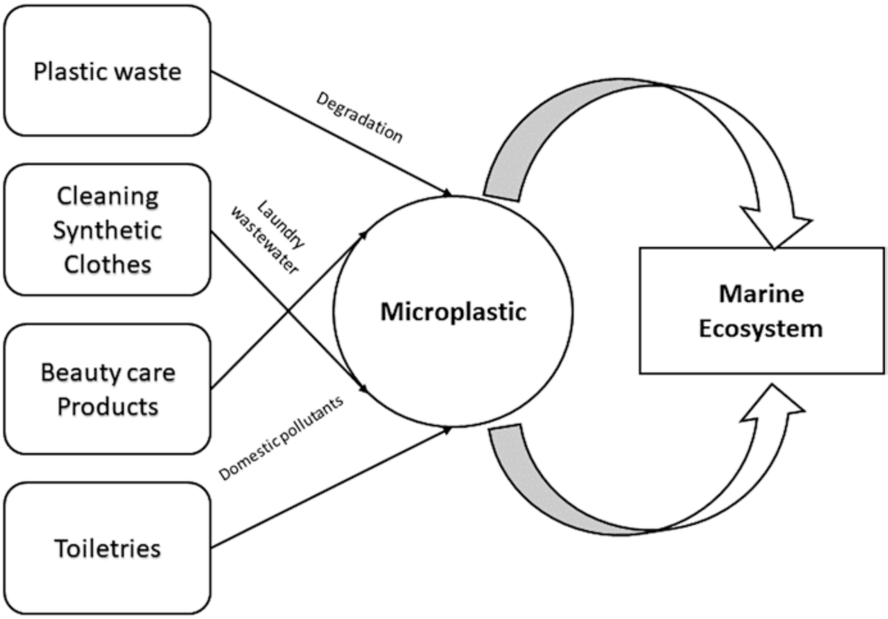

Fig. 17.1 Sources of microplastic in marine ecosystem

land-based MP from TE to ME (Barnes et al. 2009). With that Liebezeit and Liebezeit (2014) stated that MPs are released into the environment when polyethylene disintegrates in the cropland while mulching, clothing are dried, and MP-enriched wastewater is being applied as agricultural amendment. Consequently, the introduction of 3D printers for rapid manufacturing, usage of nanopolymers for providing medicine, and the thermoelectric plastic particles are responsible factors for the release of MPs in ME (Pohlmann et al. 2013; Stephens et al. 2013).

17.2.2 Occurrence of MP in Marine Ecosystem

Numerous comprehensive studies were conducted in recent years to assess the occurrence of MPs in MEs (Eriksen et al. 2013; Reisser et al. 2015) all over the world (Table 17.1). As per Li et al. (2016), buoyant MPs are a big problem in the North Atlantic tropical zone where plastic trash from both ME and TE was channeled into the particular semiarid zone of MPs hub. But Moore (2008) revealed that the maximum occurrence of MPs was in the Great Northern whorl in the earliest time. A projected amount of 26,898 particles/km^2 was detected, within the size of 0.35–4.75 mm in the southern ocean tropical whorl (Eriksen et al. 2013). About 22.5 tons of buoyant plastic waste, including plastic spacers, granules, polyethylene, polypyrrole, and flimsy plastic film, have been found in the north ocean tropical

Table 17.1 Occurrence of microplastic (MP) debris in marine ecosystem (adapted from Sharma and Chatterjee 2017)

Site	Area	MP size	Amount of MP (%)	References
Atlantic Ocean	North Sea	>20 mm mesh	48.3	Galgani et al. (2000)
	Channel East	>20 mm mesh	84.6	Galgani et al. (2000)
	Bay of Seine	>20 mm mesh	89	Galgani et al. (2000)
	Celtic Sea	>20 mm mesh	29.5	Galgani et al. (2000)
	Portuguese coast	>5 mm	43.8–91.7	Frias et al. (2014)
Baltic Sea	Baltic Sea	>20 mm mesh	35.7	Galgani et al. (2000)
Pacific Ocean	North Pacific Central Gyre	0.355 to >4.76 mm	98	Moore et al. (2001)
	Waters around Australia	0.4–82.6 mm	80	Reisser et al. (2015)
	The South Pacific subtropical gyre	0.355 to >4.75 mm	88.8	Eriksen et al. (2013)
	NE Pacific Ocean	64.8–5810 μm	74	Desforges et al. (2014)
Mediterranean Sea	Adriatic Sea	>20 mm mesh	69.5	Galgani et al. (2000)
	Gulf of Lion Marine	>10 mm mesh	70.5	Galgani et al. (2000)

whorl (Law et al. 2010). Five ocean whorls have been identified in the North and South Atlantic and Pacific Oceans and the South Indian Ocean, with about 270,000 tons of MPs accompanied in each of the tropical whorls during the past 5 years (Eriksen et al. 2014). A new waste whorl has been discovered in the Arctic Ocean zone (van Sebille et al. 2012). MPs ranging from 38 to 234 particles/m^3 were discovered in the Arctic Sea (Obbard et al. 2014). That occurrence amount is double than the amount reported earlier in the Pacific whorl (Goldstein et al. 2012). The large levels of buoyant plastics detected in the aquatic sources of Antarctica and the northern area (Lusher et al. 2015) and the central northern area are also contributing to enrich the world MP sink, which is concerning.

The occurrence of and pollution caused by MPs in ME are influenced by a variety of natural conditions like airflow, marine topography, and climatic activity (Barnes et al. 2009).

17.2.3 Impact of MP Pollution in Marine Ecosystem

MPs remain in the ME and are digestible to marine animals and living organisms because of their micro size (Browne et al. 2008). When oceanic species consume nondegradable MPs, they build up in the food web (Gregory 1996), eventually reaching greater trophic levels (Carpenter and Smith 1972). MPs are harmful to the living organisms of ME and can be the reason for harmful infections if the ME's organisms uptake these micropollutants (Fendall and Sewell 2009). Till now, all over the world, MP particles have been found in all the marine species of Aves, Reptile, Chondrichthyes, and Envertabrata class (Cole et al. 2011). MP harms these sea creatures by blocking the digestive system suppressing eating owing to tiredness, inhibiting gastrointestinal enzyme release, causing hormonal imbalances, delaying fertilization, and causing sterility (Eastman et al. 2020; Wright et al. 2013). Besides, the detrimental and persistent impacts of ingesting MP have an extensive influence causing the reduction of food consumption and deaths amid ME's creatures (Wright et al. 2013).

MP ingestion through the marine organism and reaching the food web is an old concern. But in this present time, MP's ability to uptake by the marine organism from the MP sources and transmit it to the next level is a serious threat for the ME (Browne et al. 2007; Mato et al. 2001). On the other hand, the ecological formation of coral reefs provides a defense to the ME, but the corals also absorb the MPs. As a result of their inability to decode MP particles, MP contamination has a severe impact on coral health. In a comprehensive investigation by Cole et al. (2011), MP consumption was investigated in 15 distinct phytoplankton species. In the Baltic Sea, zooplankton species also consumed MP, where *Marenzelleria* was identified with the maximum amount of MP ingestion (Setälä et al. 2014). Another notable MP impact was measured among the marine aves species. Around 30–35% of the MP particles discovered in marine bird's intestines were in the shape of industry-derived plastic pellets (Blight and Burger 1997; Ryan 1987). This MP uptake deteriorates the regular food consumption of the seabirds, later causing malnourishment and sickness (Tanaka et al. 2013). The harmful impacts of MP have also been stated for many large marine animals of ME (Derraik 2002). According to Nerland et al. (2014), almost 61% of Brazilian turtles were detected with MP particles in their intestine. Baleen whales, a sea animal, were particularly vulnerable to MP ingestion because they were engaged in screening organisms that screen oceanwater and allow the passage of MP into their systems (Fossi et al. 2012). Therefore, all the marine animals and organisms are directly affected, and some are indirectly admitted from the detrimental impact of MP pollution.

17.3 Microplastic Pollution in the Freshwater Ecosystem

17.3.1 Sources of MP in Freshwater Ecosystem

MPs can move in the ecosystem through a range of ways and from a range of sources. But there are three possible ways to infiltrate into FEs, viz., wastewater processing release, agronomic runoff from sludge-treated land, and outflow of wastewater due to excessive rainfall (Eriksen et al. 2013) (Fig. 17.2). It's noteworthy to mention that the existing framework of wastewater treatment plants (WWTP) is incapable of removing MPs. MPs removal rates in WWTPs are generally linked to the characteristics of treatment and technologies used in the process (Gatidou et al. 2019). Besides, natural calamities like hurricanes and other extreme weather events have also been interconnected to a rise in MPs in aquatic ecosystems (Anderson et al. 2016). Water conservation initiatives, viz., dams and reservoirs, can potentially impact MP abundance and its fluctuation in the aquatic environment.

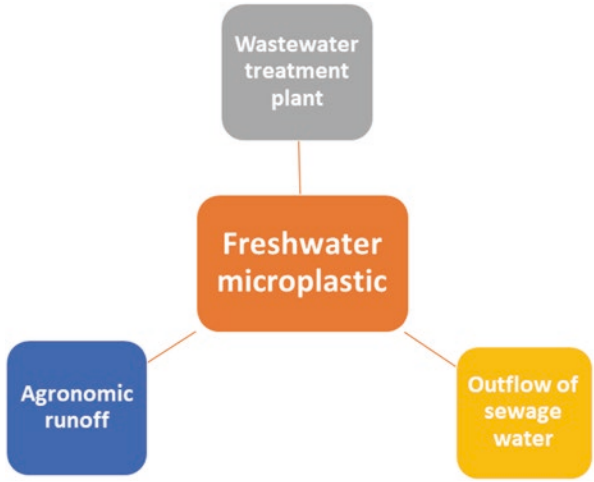

Fig. 17.2 Three potential sources of freshwater microplastic

According to Watkins et al. (2019), MP concentrations in reservoir sediments were more significant than downstream and upstream locations. Furthermore, these processes frequently result in MPs settling to the riverbed and being concealed by overlying silt. Sediments are much more resilient, and MPs are moved more slowly than those that move or buoy in the surrounding water (Su et al. 2016). So, depending on these considerations, the climatic components and permanence of microplastics should be explored extensively.

17.3.2 Occurrence of MP in Freshwater Ecosystem

Extraction of MPs in natural contexts could be difficult, especially when coupled with organic-rich components such as sediments and soils. As a result, high pigment concentration and particle and fiber degradation hinder spectroscopic observation and investigation of MPs, necessitating the use of specialized technology (Klein et al. 2018). According to findings from recent studies, MPs are abundant in freshwater habitats. In the Rhine River of Germany, the average MP concentration was 892,777 particles/km^2, where the highest abundance was reported to be 3.9 million particles/km^2 (Mani et al. 2015). Besides, although there were nearly zero wastewater transportation and infrastructure issues in the Three Gorges reservoir area of China, the top layer water's MP abundance was from 1597 to 12,611 n/m^3 (Di and Wang 2018). These investigations might overlook MP abundances as isolation and detection rely on eye viewing approaches, which might miss those within the range of micron size. The environmental prevalence of MPs in the FE in the different continents of the world is further examined by Dris et al. (2015), Wu et al. (2018), and Khan et al. (2018).

17.3.3 Impact of MP Pollution in Freshwater Ecosystem

The ruinous impact of MP pollution is happening over the freshwater species (especially fish). Sanchez et al. (2014) found MP in the digestive tract of 12% of the fish by investigating *Gobio gobio* in 11 French streams. This study was the primary study report for the intake of MP by fish species, but depending on the feeding technique of fishes. On the other hand, *Daphnia magna* rapidly intakes MP in controlled conditions, crossing the epithelial cells and accumulating the adrenal gland (Rosenkranz et al. 2009). More research has been done on MP intake-prone biota, but the toxicological consequences on freshwater species have yet to be more deeply studied.

Owing to their huge surface-volume proportion and chemical content, MP assembles waterborne pollutants such as metals (Ashton et al. 2010) and hazardous chemicals (Koelmans et al. 2013). An investigation about the relationship between MP and hazardous chemicals (viz., DDT) is reported by Engler (2012), and several studies also claimed the existence of polycyclic aromatic hydrocarbons in MP of the freshwater environment (Bakir et al. 2014; Fisner et al. 2013; Fries and Zarfl 2012). However, there is still a dearth of information on other key pollutants such as opioids and genotoxic substances. Those evidence of studies associated with chemical contamination from MP may act as a route transferring environmental contaminants from water to biota.

There is a consideration not only about the intricate combination of hazardous chemicals contained in and/or uptake MP by biota, but also biofilm development by microbes is also another concern. Mincer et al. (2019) reported a varied microbial community (plastisphere) adhering plastic marine detritus in the North Atlantic. Among the plastisphere members, some are hydrocarbon-degrading bacteria. Besides, they have also identified a predatory human pathogen such as the genus *Vibrio* dominating plastic particles. Therefore, MP has the potential to perform as a vector for waterborne human pathogens, affecting the hygienic quality of the water.

17.4 Microplastic Pollution in the Terrestrial Ecosystem

17.4.1 Sources of MP in Terrestrial Ecosystem

Generally, sludge exploitation, soil surface wrapping by plastic, irrigation linked with sewage, road runoff, atmospheric discharges, etc. are factors that contribute to the release of MP in the TE (Bläsing and Amelung 2018). But the core source of MP release in TE is agricultural utilization with sludge exploitation and mulching. In spite of removing the MP at a significant level in wastewater treatment systems, most microplastics persist in sludge. Several studies showed that the average amount of detected MP in wastewater is 1500 to 24,000 particles/kg (Mahon et al. 2017). The most concerning thing is that those sludges containing MP are applied as a fertilizer in agricultural fields, which enhance the MP level in farmland soils (Willén et al. 2016). China's projected sludge-derived MP reaching natural habitats was up to 1.56×10^{14} particles upon gross sludge output (Li et al. 2018), and the amount of MP input in Western farmland was up to 430,000 tons (Nizzetto et al. 2016). Besides, mulching is a common practice for increasing crop yield and improving quality crops, and that's another key source of releasing MP in TE. This mulching practice is going on covering a huge area of farmland, and the usage of this practice is increasing 5–10% every year (Steinmetz et al. 2016). After that, significant quantities of plastic will end on the earth's crust, where it will gradually degrade as MP or even nanoplastics.

Secondly, MP also generated from washing machine discharge, cosmetics, and skincare items is abundant in untreated sewage water. Using untreated sewage water for watering the farmlands provides a source of MP release in soils and contaminates TE (Bläsing and Amelung 2018). According to Hartline et al. (2016), a considerable quantity of plastic content with mean sizes 164–327 mm was found in irrigation wastewater. Moreover, MPs were found in home wastewater in amounts as high as 627,000 particles per cubic meter (Majewsky et al. 2016). These plastic particles possibly find their way into farm soils due to irrigation with processed wastewater or as a result of natural flooding.

Discharge from highways or metropolitan cities, atmospheric transfer, and other factors can contribute to the accumulation of MPs in TE. Furthermore, MP pollution in soils can also be influenced by unlawful garbage disposal near roadways and tire corrosion. However, there has been no research that determines the number of MPs caused by unlawful garbage disposal into soils. But for the tire corrosion, a study by Bläsing and Amelung (2018) estimated that the amount of the tire grit in Swedish and German TE is about 10,000 and 110,000 tons, respectively. Atmospheric transportation has the ability to take MPs across distant locations and is reported to lead a percentage of MP in soils. Zhou et al. (2017) investigated the varied forms, precipitation flows, and climatic variation in MPs in greater depth in the coastal ecosystem. It recognized atmospheric MP transportation as the crucial source of MP release in China's coastal region. In addition, atmospheric dispersion analysis of MPs in the urban environment of Paris identified 29–280 particles/m^2/day, where about 90% were fiber plastic, and 50% among the detected particle's size was >1000 mm (Dris et al. 2015). So, conceivably, MPs bearing other air contaminants may spread between various locations of TE via atmospheric transportations.

17.4.2 Occurrence of MP in Terrestrial Ecosystem

MP in TE has met very minimal interest even though multiple inquiries have quantified their presence in ME. In TE, soils interact with the earth surface, hydrological cycle, weather, and living organisms. When MP enters into the TE, it can prevail, obtain, and attain significant concentration in the soil, posing a threat to ecosystems and living biota (Chae and An 2018; de Machado et al. 2018). Besides, MP also functions as a medium for the transmission of hazardous contaminants from soil

Fig. 17.3 Occurrence of microplastic in different soil types of terrestrial ecosystems

dynamics to soil organisms, thus posing a concerning risk for the TE. For instance, to quantify the risk assessment of MP, Zhang et al. (2018) identified potential hazardous MP particles in the coastal environment of China as the TE is the major source of MP release to the aquatic ecosystem.

According to Horton et al. (2017), the overall amount of MP contamination on soil might be 4–23 times higher than the ME. Scientists have been focusing so much on MP pollution in the TE in recent times. Figure 17.3 summarizes recent MP research phenomena in several soil conditions.

The MP range in industry-based soils in Australia was from 300 to 67,500 mg/kg (Fuller and Gautam 2016). In the coastline of Hebei, China, MPs were isolated, and their physical properties were examined by Zhou et al. (2016). They measured the MP abundance following the dry weight method where the occurrence of MP's range was 75% and 20% for granular and fragment size, respectively, of total MP abundance. Another study revealed the occurrence of MP up to 593 particles/kg in several Swish floodplain locations (Scheurer and Bigalke 2018). On the other hand, Li et al. (2018) investigated the wastewater sample where the MP range was relatively greater than floodplain or coastal soils.

MP release in agricultural land is mostly caused by the use of plastic in farming practices. Logically, the main source of MPs might be using of plastic covering on soil and the application of sludge (Nizzetto et al. 2016). Besides, Zhang and Liu (2018) found plastic particle abundances ranging from 7100 to 42,960 particles/kg among all the 50 samples taken from agricultural fields, where except 5% plastic particles were all recognized as MP. Considering the concerns with soil characteristics, such as soil pH, living organism matter content, and inorganic matter contents, an increasing number of experts are focusing on MP contamination in TE.

17.4.3 Impact of MP Pollution in Terrestrial Ecosystem

Some publications highlighted the prospective consequences of ubiquitous MP pollution, emphasizing the detrimental effect on TE. Therefore, the impacts of MPs on TE are mostly unknown at this time. TE encompasses a wide range of soil types, as well as soil animals, soil microbes, soil-plant growth, and land-based food production. In Fig. 17.2, we have demonstrated the occurrence of MP in different types of soil. In this section, we'll look at the negative effects of MP pollution in various sections of TE. As per Chae and An (2018), earthworms, collembolans, isopods, and mites are the most commonly employed soil creatures.

Earthworms are typically originated in the soil, where they consume organic compounds and live on it. Earthworms are treated as one of the most valuable soil biota because they are convenient in performing experiments and they can swallow MP and then produce secondary MP, as well as transmit MP into the soils through their activities. As a result, earthworms are often frequently chosen to investigate numerous pollution and their consequences on TE (Ng et al. 2018). In the study of Gaylor et al. (2013), they focused on the impact of MP-enriched biosolid on *Eisenia fetida*, which indicated the transportation of MP in earthworms. Another study revealed that *L. terrestris* carried MP from the topsoil into their crevices through particle size choosing manner, implying that this biotic MP transportation within soils could negatively impact groundwater and the TE's food web. According to recent research, MPs can also be uptaken by *Caenorhabditis elegans* (Lei et al. 2018a, 2018b). The detrimental impacts comprised gastrointestinal injury and oxygen imbalance between production and accumulation in the body, as seen by the lower level of Ca and active influencing oxidate damage gene as well as decreasing life expectancy, body size, and spawning on nematoid. One study by Zhu et al. (2018) found one springtail species named *Folsomia candida* in soil that was ingested PVC, which had a detrimental impact on its intestine, height, and reproduction capabilities.

According to the latest report by Bläsing and Amelung (2018), the fungus *Zalerion maritimum* can use plastic in a particular nutrient broth to lower MP's bulk and shape. It suggests that fungi could play a role in the decomposition of MPs. Despite this, most of the microbiomes in TEs hold a limited lifespan and are also modest in size. The relationship of MPs and microorganisms in TE remains a major concern. Besides, MPs have been found to accumulate in yeast and spoilage microorganisms in several investigations, implying that MPs may accumulate or enhance across the terrestrial food chain (Chae and An 2018). Several investigations (de Souza Machado et al. 2018; Rillig 2018) reported that MPs are responsible for disrupting essential soil-water connections, which impact soil formation and microbiological activities. Additionally, that MPs could be treated as long-term anthropogenic disturbances and factors of shifting the worldwide TE.

Analyzing MPs collecting the sample from the soil, soil nematodes, and poultry excreta belong to the small home farm, Huerta Lwanga et al. (2017) claim that MPs can transmit across the terrestrial food web where they found chicken excrement had a significant proportion of MPs. The findings revealed that MPs might enter human's food web through a systemic mechanism. To quantify the transportation of MP, *Hypoaspis aculeifer* and *Folsomia candida* were the key species in the investigation by Zhu et al. (2018). However, MP transmission through the food web in TEs poses a risk to public health to a greater extent.

17.5 Future Perspectives

Stopping things at their source is the simplest and most considerate strategy to reduce or eliminate the risk of MP pollution at all the ME, FE, and TE. We should reevaluate plastic as a recyclable, reusable material rather than stuffing massive landfills with it.

- Plastic manufacturing companies should think about improving technology and generating plastic as a more biodegradable substance.
- Alternative sources and technology of plastic goods (viz., Sonali bag, a 100% biodegradable cellulose-based bioplastic and alternative to polythene bags, developed in Bangladesh, Pavel et al. (2019)) can mitigate the risk of MP pollution.
- On an individual basis, we can all contribute to curtailing the risk proportion of MP pollution we utilize on a daily basis. Being more cognizant in using plastic-derived products without affecting our lifestyles can influence mitigating MP pollution.
- Researchers should consider the potential risk of MP pollution on human health not just through food chains but also by respiratory systems.
- Until today, the behavior of MP in aquatic ecosystems is not known to us to a certain extent. Based on the available data, precise modeling methods are necessary to identify MP loads in FEs.
- It will be vital to identify which plastic features facilitate absorption and the consequence of MP in biota. In this regard, laboratory and field tests must be examined to assess the real vulnerability of MP.
- The biological impacts of MP ingestion by marine, freshwater, and terrestrial species must be studied in greater depth to reduce severe metabolism and toxicological consequences.

- Establish a novel paradigm for assessing the risk of MP, which can be both explicit and implicit stressors for the aquatic (ME and FE) environment.
- Due to scarcity of information on MP level, amounts, kinds, and components in FE and TE, existing datasets are inadequate to assess the potential risk status of aquatic and soil MPs on a global scale. The upcoming study should focus on assessing MPs in numerous sources of FE and TE, as well as their usage trends, on getting more statistical data.
- MPs can be absorbed by ME, FE, and TE's microbial communities as developing chronic pollutants. So, the MP's possible impact on microbiomes of all the ecosystems needs to be examined to a greater extent. Furthermore, MP transmission across the food web can cause a significant possible health risk. Transportation of MP through the generation of living organisms' impacts also needs to be considered in the long run. Research should be encompassed into ecological and industrial habitats, particularly plants, as well as how natural microorganism communities react to MP pollution in the environment.

17.6 Conclusions

The prevalence of MP pollution in the different ecosystems of the environment is evaluated in this chapter. From the overall risk evaluation of MP pollution, higher-level MP pollution is identified in the ME, followed by the FE and TE. The airborne nature of MPs and their movement by wind could be a significant source of marine MP pollution. Although MPs are abundant in various environments, their morphological similarities and chemical characteristics show that marine MPs are ultimately derived from TEs via atmospheric deposition. It is indeed undeniable that understanding of MPs in FE and TE is also improving at a phenomenal rate. Nevertheless, a significant lack of information about MP pollution and its risk level is still on hand. Numerous issues about scientific methods, existing proportion in the environment, origins, distribution, and implications of MPs in ME, FE, and TE remain unanswered which could be assessed by future research perspectives on MP pollution research.

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Chapter 18
The Ecotoxicological Effects of Microplastics on Trophic Levels of Aquatic Ecosystems

Vildan Zülal Sönmez, Ceyhun Akarsu, and Nüket Sivri

Abstract Spatial and temporal variations of microplastics (MPs) studies in both fresh water and seawater ecosystems have produced many results that support the adsorption of toxic pollutants to the microplastic surface. In addition, small-sized polymer fragments have increased their participation in the food web since phytoplanktonic organisms. This situation causes consequences that can severely limit the growth and/or development of many aquatic species. In this part of the book, the toxicity studies results examined in the last 10 years show that the properties of microplastics (polymer type, shape, size, colour, etc.), the exposed dose, the forms of exposure and the way in which functional disorders occur afterwards are addressed; methodically and conceptually. In the methodology studies of toxicity studies, it was determined that the most preferred microorganism was *Daphnia magna*. Many factors taken into account due to the ease of operation of the organism, the clarity of the test procedures, its comparability and the purpose of the studies carried out are effective in these choices. In addition, *Danio rerio, Mytilus galloprovincialis, Mytilus edulis* and *Scrobicularia plana* were found to be among the other organisms of frequent choice.

Toxicology studies focus more on the effect of exposure to a single concentration or independent chemicals. Therefore, researchers have struggled to find answers to the type of interaction. The movement and dynamic of microplastics in water, the similarity of MP colour to nutrients for the organism or pollutant absorption due to surface load affect the accumulation of pollutants in the organism. In addition, it has been observed that polymer type is an important factor in determining microplastic

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toxicity, while polypropylene (PP) is the most common type of microplastic in detection and analysis studies, toxicology and MP studies have shown that studies on polyethylene (PE) and polystyrene (PS) are high. The pressure of these polymers on each step in the food web, when additives used in the plastic manufacturing process are added, leading to toxicology results reach to a toxic or very toxic level.

Keywords Ecotoxicological · Microplastic · Trophic level · Aquatic ecosystem

18.1 Introduction

Aquatic and terrestrial ecosystems are linked, thus, a change that occurred in one system leads to have an effect on the other system. Many factors such as anthropogenic activities have an adverse influence on the coastal and marine ecosystems for decades. Owing to unsustainable development and construction activities make the accumulation of debris or garbage is one of the serious human-posed threats to marine and coastal systems (Richmond 1993; Thushari and Senevirathna 2020). Plastics are considered more resistant in ocean basins because of their unique characteristics compared to other types of debris (Rosevelt et al. 2013; Thushari and Senevirathna 2020). Due to the fact that aquatic ecosystems are open systems leading to be contaminated with all kinds of macro and micropollutants from many sources (Lu et al. 2021). As discussed in the previous chapter, environmental issues related to plastic pollution have become more serious since microplastics (MPs) in the aquatic environment have shown to increase. The studies on MPs in aquatic ecosystems have focused on the relations between MPs and biota as well as their effects on biodiversity (Seymour et al. 2017; van Hoof et al. 2019). MPs, predominant plastic litters with less than 5 mm in size are found to be on the ocean surface and it is estimated that there are more than 4.8 billion microplastics in the marine environment. (Thompson 2006; Eriksen et al. 2014; Auta et al. 2017). It is reported that freshwater environments, water column, sediments and biota of coastal areas, and open ocean, are contaminated with MPs worldwide (EC 2000; Thompson 2015; Coyle et al. 2020).

Monitoring and measuring the changes in the aquatic area is the most effective way of protecting aquatic ecosystems (Canning and Death 2018). Toxic substances at even very low concentrations can affect organisms directly at the lower trophic level and affect organisms indirectly at the higher trophic level. Therefore, ecotoxicological evaluations play important role in protecting these environments from pollution and monitoring water quality (Bekturganov et al. 2016). Ecotoxicity tests are important for monitoring water quality and discharge sites, protecting living organisms in the food web, and determining the stimulating effects of toxic substances on organisms (Berber 2021; Leblond et al. 2001; Whitford et al. 1999). Monitoring pollutants and evaluating their pollution status can be performed by bioindicators that are used in the analysis of toxic effects (Fossi et al. 2018; Berber 2021). This is also used in determining the ecotoxicological properties of plastic waste and its degradation products that are recently discovered pollution types in aquatic ecosystems.

Many disciplines, especially environmental sciences, have shown great interest in microplastics and toxicity studies. The differences in the method of obtaining MPs, whose toxicity will be studied, determine the scope of studies. The studies with MPs are isolated from aquatic ecosystems, in which their surface and/or chemical structure of the polymers changed by unknown factors are different than the studies of pristine polymers. Removing toxic factors from these MPs to obtain a non-toxic plastic surface and working with these structures are very challenging. Likewise, studies on whether toxicity is caused by polymer properties or changes in MP surface or structure, or additives used during MP formation are multidisciplinary and require high cost. Therefore, in the studies of determining the toxic effects of polymers, commercial or pristine MPs are used instead of MP particles that are found and isolated from ecosystems (Paul-Pont et al. 2018). However, in the studies focused on MPs that are degraded in nature, they can be successful with preliminary analyses after undergoing the necessary preliminary processes including rinsing with seawater, modification of surface load and biofilm formation.

This section focuses on ecotoxicological studies that are performed to determine the possible biological effects and importance of microplastics (MPs) on aquatic organisms that are detected in aquatic ecosystems. For this purpose, all ecotoxicological studies of MPs, including MPs obtained from aquatic ecosystems and MPs consisting of pristine polymers are considered. The studies performed in the last 10 years with using the keywords of "MPs" and "ecotoxicology" are represented from the most commonly used organisms to size ranges and from the most studied polymer structure to exposure duration. Therefore, the Scopus database was used in data search and collection. The search strategy was conducted by searching the titles, abstracts, and keywords of the articles and performed from 2010 to 2020. Furthermore, "Environmental Science" was selected as the subject field.

18.2 Environmental Fate of Microplastics in Aquatic Ecosystems

The breakdown of plastic waste creates a large number of secondary microplastics that are the main source of MPs in aquaculture (Jiang 2018). In addition, plastics can originally be produced as primary microplastics in micro sizes, and they can often be used as cleaners in some personal care products (Fig. 18.1) (Cole et al. 2011). Various micro and nanoparticles with dimensions greater than 100 nm are also detected in marine ecosystems (Gangadoo et al. 2020). Due to the problems that they caused in the environment in the last 10 years, researchers have mainly directed their studies to the sources, determination of situation and ecological effects of microplastic pollution and ecotoxicological studies rather than macroplastic pollution (Hall et al. 2015; Hartmann et al. 2019; Verla et al. 2019).

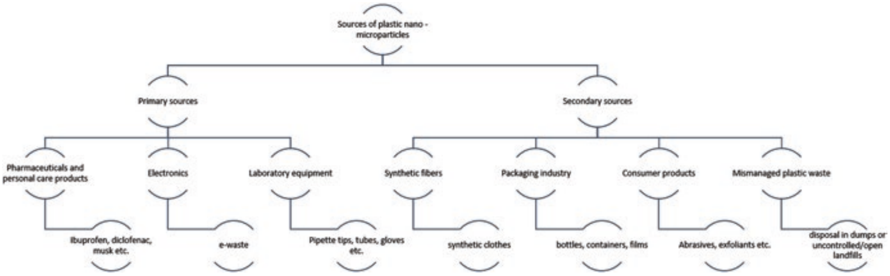


Fig. 18.1 Primary and secondary sources of microplastic and nanoplastic in the environment (Adapted from Kik et al. (2020))

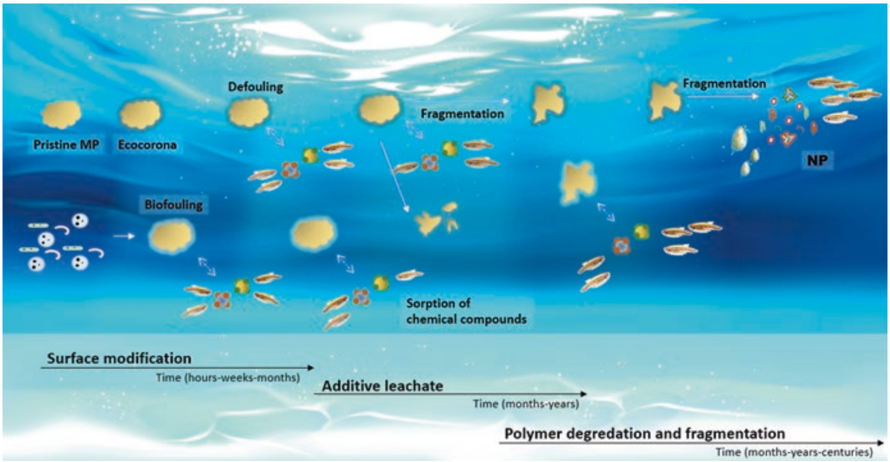


Fig. 18.2 Hypothetical evolution of the physicochemical and biological modifications of microplastics and nanoplastics released in aquatic environments (Adapted from Paul-Pont et al. (2018))

The plastics entering into aquatic ecosystems are broken down by environmental, physiochemical and biotic factors such as ultraviolet radiation, mechanical wear, and biological degradation by microorganisms (Fig. 18.2) (Barnes et al. 2009; Cole et al. 2011; Paul-Pont et al. 2018). All plastic wastes are exposed to changes that are called "aging" after entering into the environment. These changes may affect polymer composition independently and/or simultaneously and may change the physical integrity and surface properties of the particle (White 2006). Coating of surfaces with organic and inorganic material (ecocorona (Galloway et al. 2017)), hydrolysis, photo-oxidation, mechanical wear, additive release and pollutant adsorption, microorganism colonization and possibly biodegradation can be added to the weathering processes in the marine environment. During this process, there may be an increase in the plastic surface/volume ratio caused by ecocorona and the plastic surface property, chemical identity and ways of interaction with organisms may change (Lin et al. 2014; Mattsson et al. 2015; Canesi and Corsi 2016). The formation and colonization of ecocorona by microorganisms is a fast process that can be completed in hours and days, while other decomposition mechanisms such as additive leakage and photo-oxidation take months or years to complete. During their occurrence in the aquatic environment, their geometries, surface properties and chemical compositions are permanently altered and ultimately their toxic effects could greatly change (Andrady 1990; Rajakumar et al. 2009; Paul-Pont et al. 2018).

In prior studies, MPs were considered inert particles with no toxic effect other than their mechanical effects, however, they are now considered potentially harmful structures for organisms due to their chemical structure (no solids, unpainted and/or physical change) or exposure and sensitivity (Anbumani and Kakkar 2018; Galloway 2015; Prata et al. 2020). Plastic additives (PAs) were uniformly dispersed to improve or modify mechanical properties (fillers and reinforcements), modify colour (pigments and dyestuffs), provide resistance to heat and aging (antioxidants and stabilizers), provide resistance to light degradation (UV stabilizers), improve flame resistance (flame retardants) and processing characteristics (recycling additive) and improve the performance (anti-static/conductive additives, plasticisers, blowing agents, lubricants, mold release agents, surfactants, preservatives) of the polymer. MPs can have toxic and endocrine effects because PAs exist in their structure as well as their mechanical effects (Fries et al. 2013). MPs, which have non-toxic properties due to their polymer structure, can be transformed into a toxic plastic due to the presence of PA or can show synergistic effect/additive effect with microplastic PAs with toxic polymer structure.

Furthermore, MPs detected in aquatic areas are exposed to other pollutants (e.g., heavy metals, CECs) and can act as vectors for different pollutants. MPs show strong adsorption capacity due to their particle size, surface area and high hydrophobicity (Prata 2018; Fang et al. 2019). They adsorbed environmental contaminants (e.g., pesticides, herbicides, persistent organic pollutants (POPs), hydrocarbons) and heavy metals (e.g., Al, Cd, Co, Cr, Cu, Fe, Hg, Mn, Pb and Zn) on their surface (Guzzetti et al. 2018). MPs' surface features may alter enabling them, to adsorb hazardous contaminants, organic substances, nutrients and living organisms (e.g., microorganisms, plants, algae and marine animals) from marine environment. MPs also affect these contaminants' bioavailability and toxicity (Galloway et al. 2017; Guzzetti et al. 2018). The concentration of some environmental chemical contaminants adsorbed in MPs can be 10^5 to 10^6 times higher than the surrounding seawater (Mato et al. 2001; Rios et al. 2010; Sharifinia et al. 2020; Ziccardi et al. 2016).

18.3 Ecotoxicity Tests and Microplastics

Microplastic pollution is found to have an adverse effect on biodiversity in aquatic ecosystems and be one of the biggest threats to biota. Other organisms are exposed to these pollutants through food webs and these toxic chemicals can be accumulated in higher trophic levels such as fish. However, the studies reported that MPs are found in the digestive system of animals (beached marine mammals) where does not clarify whether the ingestion of MPs takes place owing to starvation or whether starvation is induced by MPs. Furthermore, exposure reveals that in-vivo assays are needed depending on concentrations, testing conditions and related endpoints. Particle's concentrations, characteristics and exposure conditions that are used in these assays are the factors that need to be taken into account since most of the studies focused on direct microplastics toxicity (Prata et al. 2021). The most important finding in the detection of these threats is the ecotoxicology studies, which should be carried out frequently at various trophic levels.

Ecotoxicology is a multidisciplinary research area that focuses on investigating the level of exposure, accumulation and effects of exposure of environmental stressors at the levels ranging from ecosystem and biosphere (Zhou et al. 2019). The development of new chemicals with contemporary technology, the increase in the use of micro-nano-sized particles make ecotoxicology studies use new techniques (Campana and Wlodkowic 2018). In fact, the use of molecular biological techniques is of great importance in unveiling unknown threats of micro/nano polymer structures to organisms.

In recent times, many dynamic changes in aquatic ecosystems such as physics-chemical or current systems and freshwater flow models such as water temperature-salinity increase as well as climate change have been detected globally (Delorenzo 2015; Mckinney et al. 2015). In fact, extensive studies on the effects of such environmental interactions on organisms were performed. However current data on toxicity studies in aquatic ecosystems and the effects of global change; unfortunately, is considered insufficient for simulation and modelling studies (Hooper et al. 2013). It was found that there were more than 10,000 research articles on the subject of "ecotoxicology" and in the field of "Environmental Science" from 2010 to 2020. There are 4985 and 658 publications on MPs and NPs in the Web of Science, respectively (21 March 2021). These studies encourage investigating the issues of MP abundance and distribution and the possible toxic effects of MPs in different environmental matrices. However, these studies are very limited when investigating the toxicological effects of plastics according to polymer structures and content, size and exposure time. In 2008, the first study of the potential ecotoxicological effect of MPs on aquatic organisms was conducted by Browne et al. (2008) (de Sá et al. 2018). Since then, field and laboratory studies on the effects of MPs on aquatic organisms have increased significantly. The 165 research articles were obtained in the field of "Environmental Science", which studied the ecotoxicological effects of MPs on aquatic organisms from 2010 to 2020. "Ecotoxicology of microplastics" studies account for only 1.65% of all "ecotoxicology" studies. It is found that there has been an increase in the ecotoxicology of MPs over the years and these studies reached the maximum in 2018.

Ecotoxicological tests are conducted to determine the harmful effects of chemical substances on biological systems, the dose-response relationship, the conditions in which toxicity occurs and the nature and quantity of toxic effect. The structure of the chemical substance, the organism exposed to the chemical and the form of exposure play an important role in determining the toxic effect. The structure, chemical activity, physical properties (pH, resolution, density, etc.) and the presence of other chemicals are among the factors that change toxicity. Therefore, the toxicity of pollutants varies based on pure chemical or unprocessed/less treated chemicals (Patel et al. 2019).

In toxicological tests, selecting not only the correct test type, but also the correct test organism is important to achieve ecologically up-to-date and meaningful results (Peyravi et al. 2020). The trophic levels of the species, their reproductive potential and strains of these species are considered the most important detail (Walkinshaw et al. 2020). This is because many factors such as species and strains of selected organisms, age, sex, temperature, body mass index, maturity and other experimental conditions are known to affect the numerical value of EC_{50}/LC_{50} (Sönmez 2016).

Another cause in the evaluation of toxicity is exposure pathway (oral, respiratory, skin and injection), duration and frequency of exposure which refer to how quickly and how often concentration is taken, respectively. The dose-response relationship, on the other hand, is the key to ecotoxicology and is essential for all exposures regardless of being acute or chronic. This describes the relationship between the level of exposure to cause and the biological response to that cause. There are different results obtained in biological experiments that are performed with a large number of organisms. However, with the standardization of test organisms and experimental conditions, uncertainties in the detection of endpoints (LC_{50}, EC_{50}, etc.) can be minimized (Sönmez 2016; Merola et al. 2020).

18.4 Factors that Affect Toxicity of Microplastics

Test organisms and their physical and ecological characteristics (species, sex, life stage, trophic state, starvation, age), duration, shape and concentration of exposure to MPs; the physical (type, size, density, colour) and polymer type of MPs affect ecotoxicological studies of MPs (Miloloža et al. 2021) (Fig. 18.3). These factors are discussed in sub-headings.

18.4.1 Organisms in Toxicity of Microplastics

The species to be used in ecotoxicological tests should be eurytopic species, and abundant and useful organisms when making international comparisons. Their physical and ecological characteristics should be well known, such as their places at the trophic level, actual nutrition (such as not being an omnivore), and no need for long generation time in obtaining a new generation (Ma et al. 2019; Sönmez 2016).

With the response of bioindicators to MPs, it should have the ability to reflect the response of taxon or even other organisms in the ecosystem. In terms of its availability in toxicity tests, it provides advantages such as having easy taxonomy, being

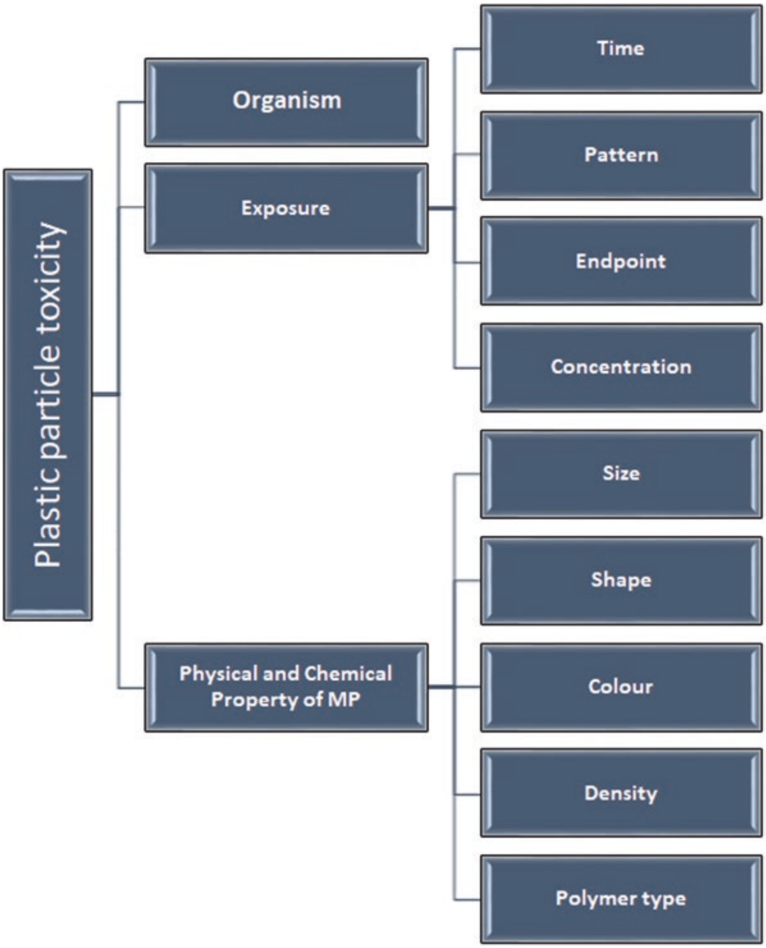


Fig. 18.3 Main and the other dependent factors in ecotoxicology of MPs

able to be identified even by non-specialists and doing culture with low cost and easily in laboratory conditions (Gerhardt 2002; Sönmez 2016).

In ecotoxicological studies, organism species show different sensitivity based on their trophic levels (producers, primary and secondary consumers, decomposers). Figure 18.4 gives an idea of the sensitivity of the tested organisms to changes in microplastic concentrations affecting different trophic levels. These organisms fulfil many ecological functions (Lu et al. 2021).

The preferred organism distribution in ecotoxicology studies of MPs that performed from 2010 to 2020 over the year is given in Fig. 18.5. In this figure, the organisms are presented depending on the most preferred rates of organisms. The most preferred type of organism is *Daphnia magna*; *Danio rerio, Mytilus*

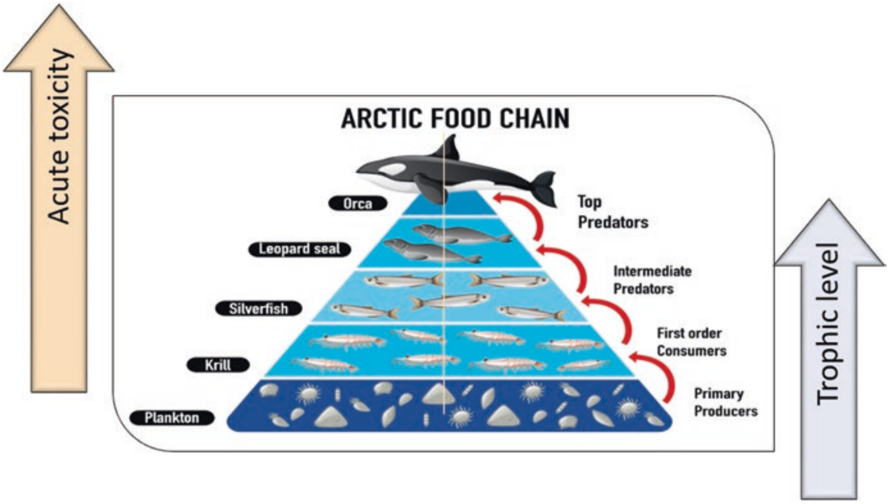


Fig. 18.4 Summary of effects of MP on aquatic organisms observed in exposure studies (Modified from Rebelein et al. (2021))

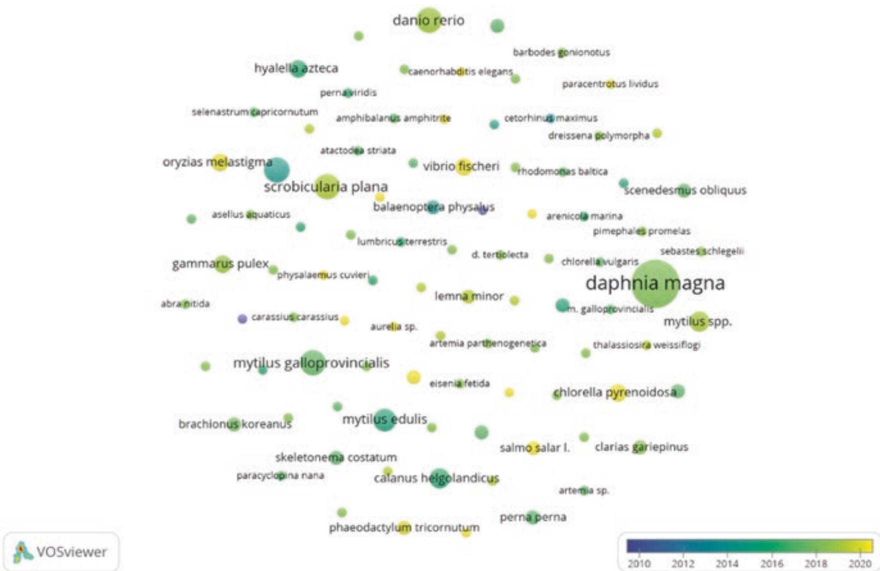


Fig. 18.5 Toxicity tests used in ecotoxicology of MPs studies over the years

galloprovincialis, Mytilus edulis and *Scrobicularia plana* are the other organisms that are frequently preferred.

Daphnia magna (D. magna) is a planktonic crustacean having an adult length less than 1.0–5.0 mm. They are the most widely used fish food in aquaculture and aquaria and have been used as one of the biological research subjects. Chronic and acute tests conducted with *Daphnia magn*a are among the most common studies in the field of aquatic toxicology (Martins et al. 2007; Sönmez 2016). *D. magna* has been widely used for ecotoxicology because this organism is reproduced easily in the laboratory and its place in food chain where it is found (Imhof et al. 2017). These organisms can also be used to determine the toxicity of liquid plastics since filtering water is used to feed them (Sönmez et al. 2020).

In the studies that were performed to determine the possible effects of MPs, the effect of *D. magna,* which is exposed to many types of polymers (Polyethylene (PE), Polyvinyl chloride (PVC), Polypropylene (PP), etc.), on the vital activities of different experimental scenarios was determined. Particle size, polymer structure, plastic additives, concentration and ambient temperature were the most commonly applied optimization parameters (Serra et al. 2020; Zimmermann et al. 2020). Studies have determined that *D. magna* can tolerate a single stressor, but with the increase in the number of stressors, it is found that they cannot cope with ambient conditions (Serra et al. 2020). Also, depending on the shape of MPs, the preference of nutrition was found to differ (Eerkes-Medrano et al. 2015; Jemec et al. 2016). All these parameters have been revealed to affect toxicity on *D. magna* and restrict their vital activity.

Danio rerio (Zebrafish) is a small tropical freshwater fish used as a vertebrate model in ecotoxicological studies (Spitsbergen and Kent 2003; Segner 2009; Lei et al. 2018). Since zebrafish is a species that can develop rapidly and whose early development stages can be followed, its genetic, morphological, biochemical and physiological parameters in the field of toxicology are followed easily. Like all other pollutants, zebrafish was used to model the toxicological effects of MPs on vertebrates. There are studies in which MPs are exposed with other stressors and their effect is determined by several biochemical and genetic biomarkers (Santos et al. 2021). However, zebrafish left in the same environment as MPs to determine the transition with the food chain and they were presented as a food source to vertebrates in the higher food chain and their possible effects were investigated (da Costa Araújo et al. 2020). Studies have shown immune deterioration (Xu et al. 2021), deterioration of the reproductive system (Qiang and Cheng 2021), affecting larval fitness (Santos et al. 2021) and having mutagenic and cytotoxic potential on them (da Costa Araújo et al. 2020) in zebrafish exposed to MPs. In fact, nanoplastics (NPs) were found to accumulate in different tissues (head, gastrointestinal tract, gallbladder, liver, heart membrane, pancreas, etc.) in *Danio rerio*, and altered larval behaviours and genotoxic effects were observed. In addition, it has been reported that the upregulation of gene expression in the nervous system is increased (Chen et al. 2017; Pitt et al. 2018a; Pitt et al. 2018b).

Molluscs (*Mytilus galloprovincialis, Mytilus edulis,* etc.) are organisms that feed by a number of ecologically and commercially important filtrations. Due to their habitat and nutritional behaviour, molluscs are more likely to be affected by MPs among other groups of benthic organisms. Molluscs absorb a large number of pollutants that can make bioaccumulation. Given that many of these organisms are widely used for food (for example, *M. edulis*), they become a potential source of MPs for humans (Wegner et al. 2012; Van Cauwenberghe and Janssen 2014; de Sá et al. 2018).

According to literature, the most commonly studied species after fish were found to be molluscs with 15% of the overall distribution (de Sá et al. 2018). Molluscs contain higher concentrations of MPs compared to marine predators (Naji et al. 2018), that depending on different nutrition strategies (Abidli et al. 2019). For example, since the bivalves feed by filtering the seawater directly, they accidentally take the MPs in the water column as nutrients leading to accumulation of MPs in their bodies (Li et al. 2015). Carnivorous molluscs (*Hexaplex trunculus, Bolinus brandaris,* etc.) are exposed to MPs at lower concentrations depending on the number of MPs coming directly from their prey.

The most preferred organisms such as *Daphnia magna* had lost popularity among researchers by the end of 2020 (Fig. 18.6). Among the different trophic levels in the aquatic ecosystem, where the ecotoxic effects of MPs are investigated, crustaceans are the most studied taxonomic group followed by fish and molluscs. According to a recent study conducted by de Sá et al. (2018), the most studied groups are given as follows; fish (44%), crustacea (21%) for large and small crustacea combined, molluscs (14%) and annelid worms (6%). Studies prove the change in the preferred toxicity test organisms within a few years.

The analysis of toxicity tests could be performed in the laboratory or the natural environment (microcosm) of the organism. The study by de Sá et al. (2018) found that the number of studies in the field (48%) and laboratory (52%) was nearly equivalent but there were differences in the examined organism groups. Fish were the most common used organism group in the field, although small crustaceans are the most common group in the lab scale studies (de Sá et al. 2018; Miloloža et al. 2021).

Recently, *Oryzias melastigma* and *Salmo salar* from the high trophic levels have received attention. Marine medaka (*Oryzias melastigma*) is considered a common ecotoxicological model when performing in-vivo molecular responses to various

Daphnia magna *Mytilus galloprovincialis* *Mytilus edulis* *Danio rerio*

Fig. 18.6 The commonly used organisms (Berserkon 2021; CleanPNG 2021; Nicepng 2021; Pngio 2021) toxicants or stresses in marine and estuarine environments. This is because of its adaptation to different salinity (Kong et al. 2008; Zhang et al. 2021). *Salmo salar* is among the fish species that is important economically and has been preferred as an ecotoxicology test organism (Abihssira-García et al. 2020).

In organisms at low trophic levels, *Chlorella pyrenoidosa* and *Vibrio fischeri* species were preferred (Fig. 18.5). Microalgae and Cyanobacteria, the major primary producers of aquatic ecosystems, are essential for primary production, eco-balance, energy flow, diazotrophy, and their sensitivity to pollutants directly leads to general deterioration in many ecosystems (Lu et al. 2021; Ramakrıshnan et al. 2010). Microalgae *Chlorella pyrenoidosa* are preferred because they possess exceptional characteristics of a fast growth cycle, ease of observing and sensitive to toxic pollutants (Liu and Xiong 2009; Yang et al. 2020).

Vibrio fischeri is a decomposer group that plays an important role in transferring nutrients and energy flows in aquatic ecosystems to higher trophic levels. Any alteration of the nutrient/food/prey balance at different levels of the food web can cause an indirect 'bottom-up' effect on species with higher trophic levels (Gambardella et al. 2019; Han et al. 2016; Trenfield et al. 2015; van Dam et al. 2008). Therefore, bacteria from the group of decomposers were also added to the MP studies and gained attention because they gave relatively simple and faster results compared to other tests (Abbas et al. 2018; Booth et al. 2016; Gagné 2017; Gambardella et al. 2019; Piccardo et al. 2020; Sönmez 2016).

18.4.2 Exposure

18.4.2.1 Duration of Exposure

Exposure assessment requires the determination of the expected or predicted concentration in organisms. The exposure route (oral, respiratory, skin pathway and injection) is the main starting point, after which studies are carried out on the duration of exposure and frequency of exposure (de Ruijter et al. 2020). Understanding the exposure route of MPs can help prevent the severe effects they can have on biota (Enyoh et al. 2020). It is known that MPs can have both acute and chronic effects on aquatic organisms (Au 2017). Acute exposure is usually in the form of a single dose intake in a short time. More broadly, the chemical is in contact with organisms in a single or complex state in less than 24 h by any means. Chronic exposure is that organisms are exposed to low concentrations of chemicals (at doses less than acute exposure) over a period of at least 10% of an organism's lifetime, continuously or at certain periods (Newman 2020).

In the studies from 2010 to 2020, chronic exposure studies (54%) are found to be more than acute exposure studies (46%). Exposure time for organisms that are exposed to MPs may not be equivalent to exposure times to environmental pollutants. In many studies, the duration of acute exposure of 24 and/or 48 h is extended to 72 h or even 96 h. Exposure periods, which may vary depending on the type of studied organism (associated with the life expectancy of the organism), were generally studied for 2, 3 and 4 days in acute toxicity tests. Depending on the conditions of the study, chronic exposure is found to have an effect as well as acute exposure. The exposure is increased to 21 days or more in cases where no response is received from the organism within 96 h. However, the exposure periods of chronic toxicity tests were selected 21, 28, 30 and 60 days.

18.4.2.2 Endpoints of Exposure

When expressing the results of ecotoxicological studies, the concentration of compounds that cause certain effects on the analysed population is used. The most common effect considered in these studies is the percentage of population deaths. Therefore, the LC_{50} represents the median lethal concentration. If the final point of the test is a negative response other than death, an effective concentration (EC) or effective dose (ED), toxicity parameter is used. Concentration causing 50% side effects is widely used in the tested population (EC_{50}). However, other levels (i.e. EC_{10} or EC_{20}) can be applied if necessary (Gagné 2017). In studies where *Vibrio fischeri* was used as a test organism (Casado et al. 2013; Gagné 2017), 5–15 min of acute exposure were studied and their endpoints were given as EC_{20} (Gagné 2017) and EC_{50} (Casado et al. 2013).

An hermetic dose response characterized by a stimulatory and inhibitory impact at low concentrations and high concentrations, respectively, which was detected in the swimming speed alteration of rotifers after both exposure times (Calabrese and Baldwin 2001). Many biological systems with low levels of toxicity show an overcompensation response to homeostasis disruption, giving a response curve with an apparent low-dose simulation (Calabrese and Baldwin 2001). The studies by Garaventa et al. (2010) and Costa et al. (2016) reported this response curve in the same model organism which was being exposed to pesticides and toxic environmental samples (Gambardella et al. 2018).

Changes in fertility, mortality and behaviour are reported as direct and indirect evidence of the negative effects of MPs (Ma et al. 2020). In many studies, the result of the ecotoxicity test of microplastics could not be expressed in terms of LC_{50} or EC_{50} (Khan et al. 2019; Selonen et al. 2020). In such cases, the researchers explained the presence of toxic effect with different biological effects and/or organism deformations, but no numerical data was reported.

Various reactions can be observed in selected species after exposure to MPs through non-fatal endpoints. After being exposed, many effects such as inhibition of algal growth, adsorption of particles on algae, formation of hetero-aggregates, decreases in the photosynthetic activity., oxidative stress, morphological changes, decrease of chlorophyll-*a* content and photosynthetic activity can be detected in microalgae. For example, MPs cause morphological changes, reduced growth, and reduced photosynthetic activity (>10 mg/L, 0.1 and 1 μm Polystyrene (PS)) in microalgae *Chlorella pyrenoidosa* (Mao et al. 2018). In planktonic forms, many studies focusing on the effects on growth and reproduction are found. For example, MPs led to immobilization (>12.5 mg/L, 1 μm PE) (Rehse et al. 2016), mortality (>0.01 mg/L, 2 μm PS) (Aljaibachi and Callaghan 2018), including transgenerational effects of reduced growth and reproduction (>0.1 mg/L 1–5 μm proprietary polymer) in the planktonic crustacean *Daphnia magna* (Martins and Guilhermino 2018). The change in trophic levels shows significant differences in toxic effects. For example, in mussels *Mytilus* spp., microplastic exposure affected the homeostasis with the production of stress and immune-related proteins and consequently increased energy expenditure (>4.6×10^5 MPs/L, 1–50 μm high-density PE (HDPE)) (Détrée and Gallardo-Escárate 2018), impacted key metabolism enzymes, and cause the upregulation of biomarkers for antioxidant response (>1.5×10^7 MPs/L, 1–50 μm HDPE) (Détrée and Gallardo-Escárate 2017).

The environmental origin of the clone is apparently linked with the differences in tolerance of *D. magna.* After several generations of toxicant-free cultivation, ponds in agriculture originate clones and high tolerance is sustained. To demonstrate critical parameters including the EC_{50}, the use of different clones of *D. magna* was used. Toxicity results can vary even with the same test species (Zocchi and Sommaruga 2019).

The effects of *Daphnia magna* were determined as immobilization rate, oxidative stress, mortality, accumulation in the gut (Jaikumar et al. 2018; Jemec et al. 2016; Rehse et al. 2016; Zhang et al. 2019). Different deformations resulting from acute and chronic exposures of the Polycarbonate (PC) (<50 μm), any PA-free pristine polymer of *Daphnia magna*, and Polyvinyl alcohol (PVA), a water-soluble plastic, are given in Figs. 18.7 and 18.8.

The response of organisms at secondary trophic level to MPs effects on development, morphological deformations, suppression of locomotor activity, damage of the intestine, mortality, decrease in size, deterioration of nervous and visual systems, accumulation in the gills, liver, and gut of the fish, deterioration of liver metabolism, oxidative stress (Jemec Kokalj et al. 2019; Lei et al. 2018; Lu et al. 2016). Studies with *Cyprinus carpio*, *Caranx hippos, Sphyrna tiburo Trichiurus lepturus,* etc. investigated the effects of accumulation in the stomach and intestine (Jabeen et al. 2017; de Souza e Silva et al. 2018).

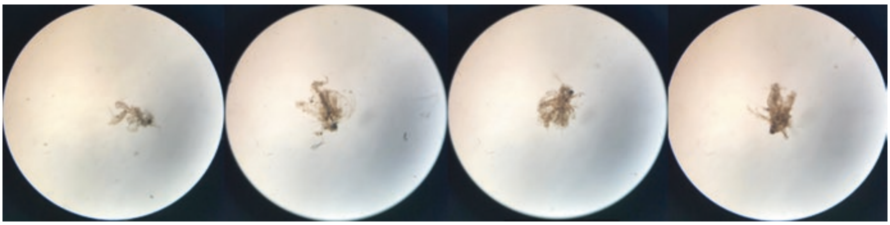


Fig. 18.7 Deformations in its body after *Daphnia magna*'s chronic exposure to PC (21 days). The working concentrations are 5 mg/L, 10 mg/L; 20 mg/L and 50 mg/L (Magnification: 4 × 10)

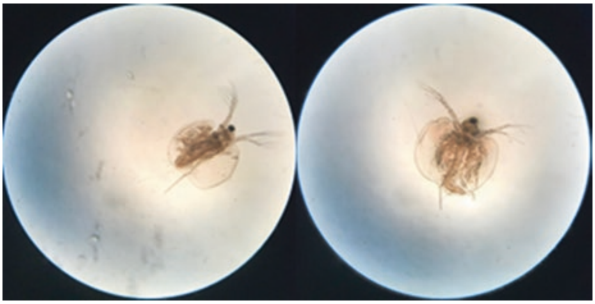


Fig. 18.8 Deformations caused by density difference after *Daphnia magna*'s acute exposure to PVA (48 h). The working concentrations are 50 mg/L and 20 mg/L (Magnification: 4 × 10)

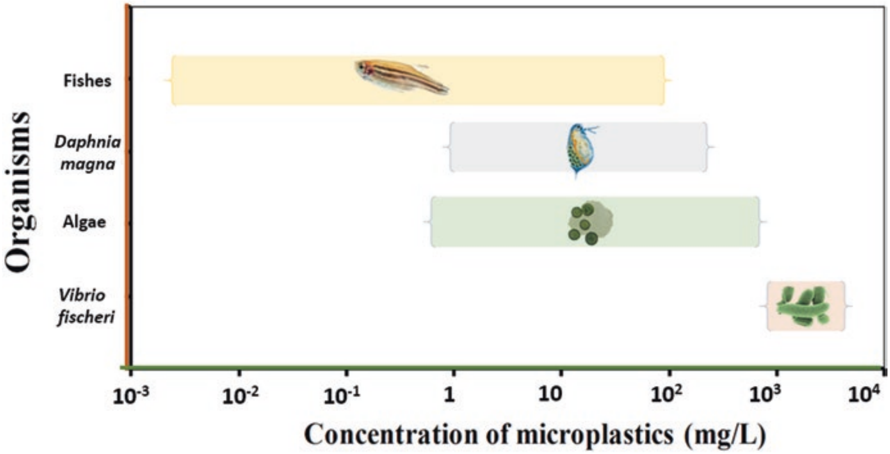


Fig. 18.9 The toxicity of MPs at different trophic levels (Adapted from Miloloža et al. (2021))

18.4.2.3 Exposure Concentration

The initial concentration differs due to two main factor that are accepted by the researchers. Over time, the abundance of MPs will exhibit increased concentration depending on both the amount of consumption and the chemical structures of plastics. For this reason, some researchers have exposed test organisms to MPs at a greater concentration than those found in nature whereas some of researchers have preferred to work at existing concentrations. As a result of the studies, there was no positive/negative relationship between the organism at the trophic level and the concentration of MPs. In order to give an insight into the sensitivity of the tested organisms to changes in MP concentrations impacting different trophic levels, the ecotoxicological concentrations range for the organisms at various levels is given in Fig. 18.9. *Vibrio fischeri* is exposed to a dose varies between 3–1000 mg/L (Casado et al. 2013; Gagné 2017), this value is between 0.1 and 800 mg/L for algal organisms (Bhattacharya et al. 2010; Casado et al. 2013; Mao et al. 2018; Wu et al. 2019). The concentrations value ranges between 0.01 and 600 mg/L in *Daphnia magna* studies (Casado et al. 2013; Rehse et al. 2016; Jemec et al. 2016; Jaikumar et al. 2018). *Danio rerio* has been exposed to different concentrations between 0.001 and 100 mg/L (Lu et al. 2016; Lei et al. 2018; Jemec Kokalj et al. 2019). The findings for toxicity effect and dose-effect response could not be presented (Ma et al. 2020).

18.4.3 Physical and Chemical Properties of Microplastics

18.4.3.1 Size

To study the trophic transfer and effects of MP intake, some approaches such as exposure to a certain size, various sizes and a mixture of different sizes have been determined (Paul-Pont et al. 2018). Since sizes of MPs studied by the researchers differed, the studies from 2010 to 2020 are compiled considering the size range. As a potential alternative, the SI scale can be used. Therefore, this would attribute to nanoplastics (1–1000 nm), MPs (1–1000 μm), *milliplastics* (1–10 mm), *centiplastics* (1–10 cm), and *deciplastics* (1–10 dm). Unfortunately, there is inconsistency between the terminologies. For example, NPs are typically considered as 100 nm in the nanomaterials field. As a result, compromise and clear subcategorization are required (Hartmann et al. 2019; Welden and Lusher 2020). 100–50 μm, 50–1 μm, 1–0.01 μm are included in the size ranges of MPs, but polymer structures with sizes between 1 and 100 nanometers (nm) are expressed as "Nano". Therefore, in current studies, plastic particles with a size range of 1–100 nm have been introduced for nanoscale plastics (Paul et al. 2020). Although NPs have been proven to have an adverse effect on growth, reproduction, oxidative stress and immune function, information about the toxicity mechanisms of NPs in ecologically related organisms remains limited. Therefore, the studies that examined in this section categorize the size ranges of 5000–1000, 1000–500, 500–100, 100–50, 50–1, 1–0.01 μm (Table 18.1). The most studied MP size range varies between 50–1 μm (Fig. 18.10).

Table 18.1 The size ranges[a] of the MPs studied and the ratio within the studies

Size range (μm)	Article percent (%)
5000–1000	1
1000–500	3
500–100	15
100–50	13
50–1	56
1–0.01	12

[a]Size range: The range that includes the MP sizes studied in the studies between 2010 and 2020. For example, in the study by Chen et al. (2020), MPs with dimensions of 5 μm were studied. It is included in the 50–1 μm range in the specified size ranges. Article rate (%): The ratio of the number of articles in the size range studied to the total number of articles regardless of size range (%) has been found in the proportion of total articles number

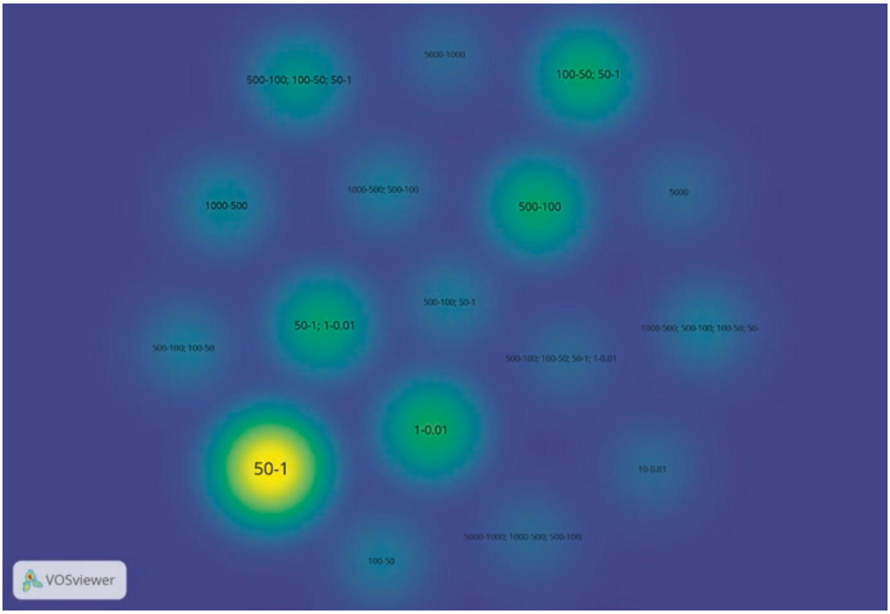


Fig. 18.10 Particle size of MPs their ecotoxicity studied from 2010 to 2020

Compared to other size ranges, this size range was studied almost 5 times more. Following this size range, 500–100 μm, 100–50 μm and 1–0.01 μm are available. Recently, researchers have conducted studies on NPs rather than MPs.

The researchers prefer to work not only with MPs of a certain size, but also with MPs in multiple size ranges (e.g. any size between 100–50 μm and 50–1 μm ranges) (Sjollema et al. 2016; Rehse et al. 2016; Magara et al. 2019; da Costa Araújo et al. 2020). In multiple working size ranges, it is seen that the "100–50 μm and 50–1 μm" and "50–1 μm and 1–0.01 μm" size ranges are mostly studied. The most important factor of why studying in this range is found to be NPs. Because NPs, which are smaller than the average cell diameter (10–30 μm) of plants and animals, potentially can easily pass-through contact surfaces such as the gastrointestinal tract, cellular wall and translocate to internal organs. Therefore, it can interact directly with the organism at the cellular level. For this reason, NPs can create toxicity in organisms or act as vectors for other pollutants like MPs (Piccardo et al. 2020).

MP sizes used in experimental exposure studies are considered a critical factor that manages any toxic effects observed in organisms (Kashiwada 2006; Lee et al. 2013; Van Cauwenberghe and Janssen 2014; Tang et al. 2020; Murano et al. 2020; Sharifinia et al. 2020). Depending on the body mass index of the test organism, the dimensions of the MPs exposure vary. According to the purpose of the adjustment, size of MPs is for food intake behaviour should be preferred in accordance with the preferred test organism (Ma et al. 2020). However, it can be difficult to ensure and replicate the homogeneous distribution of particles among experimental conditions. In most studies, traditionally, MPs size distribution was selected in the same range as the dimensions of prey of test organisms (Paul-Pont et al. 2018). The most studied types of organism (*Daphnia magna*) and the most studied MP size range (50–1 μm) were evaluated. Accordingly, it is concluded that the range of particles studied the most is within the size range that the relevant organism can digest. The study by Ma et al. (2020) reported the most common size of MPs ingested by daphnids (below 100 μm) (Kokalj et al. 2018).

In the size ranges of 0.06–0.11 μm were studied in determining the toxicity of NPs on microorganisms (*Vibrio fischeri*) in the decomposition trophic level (Casado et al. 2013). In the literature, it was determined that a maximum NP size of 1–3 μm was studied (Gagné 2017). The primary producers are test organisms of the trophic level (*Pseudokirchneriella subcapitata, Chlorella* sp., *Scenedesmus* sp., *Scenedesmus* sp., *Chlorella pyrenoidosa, Microcystis flos-aquae, Chlamydomas reinhardtii*) and NPs (0.02–1.0 μm) and MPs (1.0–1000 μm) studies were carried out (Bhattacharya et al. 2010; Casado et al. 2013; Mao et al. 2018; Wu et al. 2019).

Test organisms studied at differential and primary producer at the trophic levels, the primary consumer is in the trophic class of NPs (0.06–1.0 μm) (Casado et al. 2013) and MPs (1.0–20 μm) (Rehse et al. 2016; Jaikumar et al. 2018) in different particle sizes. Studies with test organisms in the secondary consumer trophic level for the studies carried out with *Danio rerio*. It is seen that this test organism is exposed to a wide range of size ranges from MPs (1.0–45 μm) to NPs (0.07–1.0 μm) (Jemec Kokalj et al. 2019; Lei et al. 2018; Lu et al. 2016).

18.4.3.2 Colour

Due to their visual properties, MPs can influence the hunting strategies of organisms (Ding et al. 2019). In this context, MP studies have shown that some species prefer to consume MPs of a certain colour in the environment and their causes have been investigated. The study by Pham et al. (2017) found that the fish used as a test organism prefers to consume MPs of colour and size, similar to its typical prey. In another study by Qu et al. (2018), it was determined that coloured MPs are more likely to be ingested by aquatic organisms compared to colourless MPs. Although there is no awareness of the discrimination that works correctly depending on the species' preference to look for food, it is thought that they are more likely to eat MPs similar in shape and colour to their food (Sleight et al. 2017; Wang et al. 2019a; Xiong et al. 2019) but in order to reach a more definitive judgment, the visual properties of coloured MPs and their effects on aquatic life need to be further investigated in the future (Çullu et al. 2021; Wu et al. 2019; Yin et al. 2020).

Another topic is the adsorption capacity of MPs towards pollutants. Biological responses are not induced by MPs not only because of their physical activity. The POPs can be adsorbed or monomers and additives from plastic, including endocrine-disrupting plasticisers, can be released (Law and Thompson 2014) due to the surface properties of small plastic particles. Alimi et al. (2018) reported that PE fragments showed greater adsorption capacity towards environmental toxic substances than that of other polymer types. Furthermore, colourless MPs show high adsorption capacity towards PCBs than that of coloured plastics (Prokić et al. 2019). However, another study suggests that discoloured MPs adsorb higher polycyclic aromatic hydrocarbons (PAH) than newer MPs (Verla et al. 2019). Therefore, colour is considered as a parameter that determines both which pollutant to MPs will be adsorbed and by which living creature that pollutant will prefer to be consumed.

18.4.3.3 Density

Polymers are divided into three groups which are positively buoyant polymers (HDPE, LDPE and PP), neutrally buoyant (PS), and negatively buoyant (PVC, polyethylene terephthalate (PET), and polyamide) based on their buoyancy in freshwater or seawater (Karami 2017) (Table 18.2). PS density is estimated to be equivalent to or slightly higher than seawater, and PS particles greater than 100 mm will sink in seawater, while 10 mm particles will hang (Enders et al. 2015; Karami 2017). In addition, additives, pollutants they adsorb, biofilm layers can give rise to a change in density.

The distribution of MPs in various ways in different parts of the aquatic environment (water surface, water column and sediment) affects its accessibility to organisms of different trophic levels (Betts 2008; Thompson et al. 2009; Cole et al. 2011). For example, pelagic organisms such as phytoplankton (Long et al. 2015) and small crustaceans (e.g., zooplankton) (Desforges et al. 2014), benthic organisms molluscs are more likely to encounter MPs. Because both benthic and pelagic organisms (fish, etc.) can consume MPs directly or indirectly (prey-predator relationship) (de Sá et al. 2015, 2018; Rummel et al. 2016).

In toxicity tests, another factor affecting the density of microplastic is the feeding strategy. Most of the polymers are detected on the surface or the bottom of the

Table 18.2 The buoyancy of common polymers (Adapted from Gago et al. (2019))

Polymer	Abbreviation	Density (g cm^{-3})
Polystyrene	PS	0.01 – 1.06
Polypropylene	PP	0.85 – 0.92
Low-density Polyethylene	LDPE	0.89 – 0.93
Ethylene vinyl acetate	EVA	0.93 – 0.95
High-density polyethylene	HPDE	0.93 – 0.98
Polyamide	PA	1.12 – 1.15
Nylon	PA 6,6	1.13 – 1.15
Polymethyl methacrylate	PMMA	1.16 – 1.20
Polycarbonate	PC	1.20 – 1.22
Polyurethane	PU	1.20 – 1.26
Polyethylene terephthalate	PET	1.38 – 1.41
Polyvinyl chloride	PVC	1.38 – 1.41
Polytetrafluoroethylene	PTFE	2.10 – 2.30

water column. Because of that, utilizing more various types of food enables the organism to prefer their natural feeding way and make them not to have the food from the water levels where MPs are detected at the high level of concentrations. A combination of floating, slow or fast sinking feeds is considered a potential feeding strategy. This feeding strategy could lead to a reduction in the passive ingestion of MPs by organisms and a more accurate situation whilst experiments that performed in the laboratory (Karami 2017). These changes in the density of MPs greatly affect their transport and fate in aquatic environments. Due to its density, exposure to MPs accumulating on the surface of the water and digestive status may be lower than the specified concentration. Therefore, in toxicity tests, it should be preferable to use MPs with a density close to the standard range of the polymer (Karami 2017).

18.4.3.4 Microplastics Shape

MPs can be categorized as spheres, fibers, fragments, pellets and films. The shape causes a change in the hydrodynamic characteristics of MPs and there is a relationship found between a series of biological and toxicological effects by disturbing the distribution and bioavailability. In the aquatic environment the behaviour of discharged plastic debris with the same composition was found different (Ma et al. 2020). The shape of MPs has been grouped as regular and irregular shapes, giving a critical morphological feature. The initial shape, the aging and weathering conditions have an effect on that. In general, the toxicity of fibers was found to be higher than microbeads. For example, Canniff and Hoang, where *D. magna* can digest smooth-shaped MPs more easily, was found by 2018. Another study by Eerkes-Medrano et al. (2015) found that *D. magna* feeds at a higher rate of microbeads than fibers. In another study, fibers were reported to bend in *D. magna's* intestine, thereby preventing it from feeding (Jemec et al. 2016).

18.4.3.5 Surface Charge of Microplastics

Surface charge of nano and micro materials in studies where marine organisms are used as test organisms; stability is very important for predicting the behaviour of MPs/NPs in the aquatic environment, as it affects collection and accumulation. The presence of salts can increase the particle surface charge, resulting in the absence of measurable ecotoxicological effects. In a study, the toxic effects of functional groups in polymer structure on *D. magna* were examined and it was reported that the positive surface charge showed high affinity to algae cells (Zhang et al. 2020). The studies that performed on *Artemia franciscana* larvae (Bergami et al. 2016), *Paracentrotus lividus* (Della Torre et al. 2014) and *Brachionus plicatilis* (Manfra et al. 2017) have shown that surface charge plays an important role in influencing the bioavailability and toxicity of MPs on living organisms.

18.4.3.6 Polymer Types of Microplastics

Polymer type plays an important role in investigating the toxicity of MPs. The polymer types that are taken by an organism are dependent on the ecological niche of the organism. Because, these types of particles sink and they usually are detected in water (Abihssira-García et al. 2020; Bråte et al. 2017). Plastics are detected in elasmobranchs (sharks and rays); 34.88% and 25.6% of juvenile and blue sharks, respectively, were detected to ingest marine litter in the Mediterranean Sea. Of the 109 pieces of litter, 107 were plastics of which 25.71% of plastics were MP (Bernardini et al. 2018).

The most common of the Big Six is high-density polyethylene HDPE which is followed by PET, PVC, low-density polyethylene LDPE, PP, and polystyrene (PS). Which is estimated that Europe supplies 80% of that plastic demand (Plastics Europe and EPRO 2016) and are the most frequently reported plastics that are still in marine environments (Browne et al. 2010; Karapanagioti et al. 2011; Vianello et al. 2013). PE and PP are the polymers largely detected in all environmental compartments (Enders et al. 2015; Frère et al. 2017), in line with their global production and their use worldwide (GESAMP 2016). The chemical nature of these polymers can modify their effect and first it is found related to test the big six, individually and in complex mixtures (Paul-Pont et al. 2018).

The most studied polymer types in different aquatic ecosystems were preferred in ecotoxicological studies. The majority of the studies conducted between 2010 and 2020 are PS (46%) and PE (36%) studies with polymer chemical structure (Table 18.3). Studies with other polymer structures (PP, PA, PVC, PET etc.) have an 18% share (Scott et al. 2019; Zimmermann et al. 2020; Chen et al. 2020; Piccardo et al. 2020). Just like in size range, there are studies carried out with multiple polymer types (two and three different types of polymers) in this field.

18.5 Recommendations and Conclusion

Due to the size of microplastics, the density of microplastics studies, especially in marine environments and the damage they cause to other aquatic organisms by joining the food chain, are increasing day by day. The accumulation of microplastics after they are included in the organism structure and/or the toxic effects, especially oxidative stress, suggest that more studies are needed to investigate the detailed mechanisms of microplastic toxicity. In addition, the question of whether a pollutant adsorbed to the microplastic surface will be desorbed within the organism is still being sought. It is known that microplastics increase the intake of other organic pollutants in the environment by organisms and can cause damage to their living function. In determining the toxicological effects of both microplastics and nanoplastics in the future; determining factors such as physical and chemical properties, type of organisms, dietary patterns and way of exposure of organisms to pollutants should be included. In addition, not enough studies have been done on the possible

Table 18.3 Studies on the toxicity of different polymer types

Polymer type	Shape	Particle Size (μm)	Organisms	Year	References
PE	Bead	50–1	*Pomatoschistus* sp.	2015	Luís et al. (2015)
PE		500–100; 100–50; 50–1	*Lumbricus terrestris*	2016	Huerta Lwanga et al. (2016)
PE		500–100; 100–50; 50–1	*Daphnia magna*	2017	Chae and An (2017)
PE		50–1	*Platorchestia smithi*	2017	Tosetto et al. (2017)
PE		1000–500; 500–100	*Eisenia andrei*	2017	Rodriguez-Seijo et al. (2017)
PE	Bead		*Pimephales promelas*	2018	Malinich et al. (2018)
PE		50–1	*Tetraselmis chuii*	2018	Prata et al. (2018)
PE			*Euphausia superba*	2018	Dawson et al. (2018)
PE	Bead		*Brachionus koreanus*	2018	Jeong et al. (2018)
PE	Bead	500–100	*Montastraea cavernosa; Orbicella faveolata*	2018	Hankins et al. (2018)
PE	Spheres	50–1	*Gammarus duebeni; Lemna minor*	2019	Mateos-Cárdenas et al. (2019)
PE		500–100; 100–50; 50–1	*Vibrio fischeri; Phaeodactylum tricornutum*	2019	Gambardella et al. (2019)
PE	Bead	500–100; 100–50; 50–1	*Daphnia magna; Lemna minor*	2020	Kalčíková et al. (2020)
PE	Bead	50–1	*Daphnia magna*	2020	Felten et al. (2020)
PE		50–1	*Aurelia sp.; Cnidarian jellyfish*	2020	Costa et al. (2020)
PE		100–50; 50–1	*Danio rerio*	2020	Malafaia et al. (2020)
PE	Spheres	500–100	*Mytilus galloprovincialis*	2020	Chae and An (2020)
PE		500–100; 50–1	*Chlorella pyrenoidosa*	2020	Yang et al. (2020)
PE	Fragment	50–1	*Oryzias melastigma*	2020	Le Bihanic et al. (2020)
PE		50–1	*Daphnia magna*	2020	Castro et al. (2020)
PE; PS		50–1	*Salmo salar*	2020	Abihssira-García et al. (2020)
PE		50–1	*Tigriopus japonicus*	2020	Yu et al. (2020)

(continued)

Table 18.3 (continued)

Polymer type	Shape	Particle Size (μm)	Organisms	Year	References
PE			*Vibrio fischeri*	2021	Martín et al. (2021)
PE; PP	Fiber		*Hyalella azteca*	2015	Au et al. (2015)
PE; PS		50–1	*Mytilus galloprovincialis*	2020	Zhang et al. (2020b)
PP; PE; PS		1000–500	*Cetorhinus maximus; Balaenoptera physalus*	2014	Fossi et al. (2014)
PP; PE; PS; cellulose; polyester; nylon	Fiber; fragment; bead	100–50	*Mytilus edulis*	2019	Scott et al. (2019)
PS	Bead	50–1; 1–0.01	*Tigriopus japonicus*	2013	Lee et al. (2013)
PS		50–1	*Crassostrea gigas*	2015	Cole and Galloway (2015)
PS		50–1	*Danio rerio*	2016	Lu et al. (2016)
PS		50–1	*Amphibalanus amphitrite; Artemia sp.*	2017	Gambardella et al. (2018)
PS		50–1	*Scrobicularia plana*	2017	Ribeiro et al. (2017)
PS		50–1	*Daphnia magna*	2018	Horton et al. (2018)
PS		1000–500	*Echinodermata sp.; Holothuroidea sp.*	2018	Renzi et al. (2018)
PS		500–100	*Oncorhynchus mykiss*	2018	Ašmonaitė et al. (2018)
PS		50–1	*Xenopus laevis*	2018	De Felice et al. (2018)
PS		1–0.01	*Chlorella sp.; Scenedesmus sp.*	2018	Troost et al. (2018)
PS		500–100; 100–50; 50–1	*Gammarus pulex; Hyalella azteca; Asellus aquaticus; Sphaerium corneum; Tubifex spp.; Lumbriculus variegatus*	2018	Redondo-Hasselerharm et al. (2018)
PS	Bead	1–0.01	*Vibrio anguillarum; Dunaliella tertiolecta*	2018	Gambardella et al. (2018)
PS	Fragment	50–1	*Danio rerio*	2019	Qiao et al. (2019)

(continued)

Table 18.3 (continued)

Polymer type	Shape	Particle Size (μm)	Organisms	Year	References
PS		50–1	*Artemia parthenogenetica; Daphnia magna*	2019	Wang et al. (2019a, b)
PS	Bead	50–1	*Daphnia magna*	2019	De Felice et al. (2019)
PS	Spheres	50–1	*Oryzias melastigma*	2019	Cong et al. (2019)
PS	Bead	1–0.01	*Meretrix meretrix*	2019	Luan et al. (2019)
PS	Spheres	50–1; 1–0.01	*Saccostrea glomerata*	2019	Scanes et al. (2019)
PS		50–1	*Chlorella sp.*	2019	Thiagarajan et al. (2019)
PS	Bead	50–1	*Daphnia magna*	2019	Eltemsah and Bøhn (2019)
PS	Bead	100–50; 50–1	*Epinephelus moara*	2020	Wang et al. (2020)
PS	Bead	50–1; 1–0.01	*Tigriopus japonicus*	2020	Choi et al. (2020)
PS		50–1	*Oryzias melastigma*	2020	Li et al. (2020)
PS	Bead	50–1	*Daphnia magna*	2020	Aljaibachi and Callaghan (2018)
PS		100–50; 50–1; 1–0.01	*Oreochromis niloticus*	2020	Ding et al. (2020)
PS, PA	Spherical bead	50–1	*Mytilus spp.*	2020	Cole et al. (2020)
PS; PVC		50–1	*Scenedesmus obliquus; Daphnia magna*	2020	Chen et al. (2020)
PS		1–0.01	*Tetrahymena thermophila*	2021	Wu et al. (2021)
PS			*Oncorhynchus mykiss*	2021	Karbalaei et al. (2021)
Environmental MPs (EMPs)		50–1; 1–0.01	*Dicentrarchus labrax*	2021	Zitouni et al. (2021)
LDPE		50–1	*Scrobicularia plana*	2020	O'Donovan et al. (2020)
LDPE		50–1	*Scrobicularia plana*	2021	Islam et al. (2021)
PVC; PE		500–100	*Arenicola marina*	2016	Bakir et al. (2016)
PVC		1–0.01	*Perna perna*	2018	Santana et al. (2018)
PVC; PUR; PLA		50–1	*Daphnia magna*	2020	Zimmermann et al. (2020)

(continued)

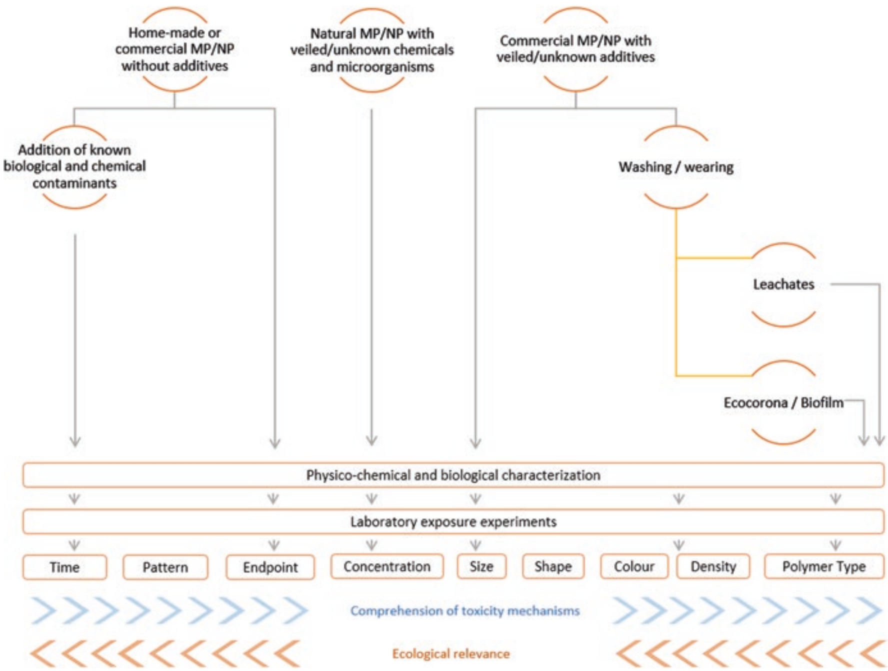
Table 18.3 (continued)

Polymer type	Shape	Particle Size (μm)	Organisms	Year	References
PVC		100–50; 50–1	*Clarias gariepinus*	2020	Iheanacho and Odo (2020)
PVC; PET		500–100	*Lepidostoma basale*	2020	Ehlers et al. (2020)
PET		500–100; 50–1	Gammarus pulex	2018	Weber et al. (2018)
PET		5000–1000; 1000–500; 500–100; 100–50; 50–1	*Vibrio fischeri; Phaeodactylum tricornutum; Paracentrotus lividus*	2020	Piccardo et al. (2020)
PA		500–100; 100–50; 50–1	*Chironomus riparius*	2020	Khosrovyan and Kahru (2020)
PA		500–100; 50–1	*Chlorella pyrenoidosa*	2020	Yang et al. (2021)
PER; PEN; PE		500–100	*Scenedemus subspicatus; Thalassiosira weissiflogii; Cordicula fluminea*	2019	Baudrimont et al. (2020)
Virgin nylon	Fiber	50–1	*Calanus helgolandicus*	2019	Procter et al. (2019)
PBT			*Balaenoptera physalus*	2016	Fossi et al. (2016)
Acrylic wool	Fiber	500–100	*Gammarus pulex*	2020	Yardy and Callaghan (2020)
Unknown composition	Spheres	50–1; 1–0.01	*Tetraselmis chuii*	2019	Davarpanah and Guilhermino (2019)

toxic effects of chemically specific additives such as azo dyes, organic and inorganic pigments used in the process of plasticising polymers on both MP and organisms. In the conditions where these pollutants are mixed both in their current form and with other microcontaminants, studies on changes in toxicity level should increase.

It is a well-known fact that there is no longer an aquatic area that is not contaminated with plastics and no more unaffected biota. The plastic pollution created by these structures, which have become impossible to remove, has recently become cross-border. Because more than half of the plastics, which need to be produced and managed with great care, are considered to be disposable. When these disposable structures reach the aquatic areas directly, they appear as undevelopable pollution. As we are affected by viruses and cannot breathe in the pandemic, it may be possible to say that soon, the aquatic areas of countries that do not manage sustainable water effectively and successfully will not be breathing due to plastic-borne waste.

Graphical Abstract



*Adapted from (Paul-Pont et al. 2018).

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Chapter 19
Ecological and Public Health Effects of Microplastics Pollution

Maria Arias-Andres and Keilor Rojas-Jimenez

Abstract Humans and ecosystems are constantly exposed to microplastics (MP). The magnitude of contamination, their ubiquity, and high persistence over time raise serious concerns about their effects on ecosystems, wildlife, and human health. MP represent a diverse class of contaminants occurring on a continuum of sizes and in various shapes and presenting a complex composition that includes several types of polymers and several associated pollutants. In short, MP are perhaps one of the most challenging contaminants created by humankind. The effects of exposure to these pollutants are of growing concern even though the type and level of exposure and the specific risks for humans and ecosystem health have not yet been entirely determined. In this chapter, we identify critical qualitative and quantitative aspects of MP sources and exposure routes and toxicity profiles and confront them with research on MP effects and estimations of risks to human and environmental health. Finally, we highlight that some novel sources of MP contamination pose a serious risk of exposure to humans and ecosystems, such as nanoplastics and the recycled plastics incorporated into road pavements and construction.

Keywords Microplastics · Nanoplastics · Ingestion pathways · Exposure · Health risks · Ecological risks

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19.1 Sources of MP Pollutants

Plastic is a material that has provided enormous benefits to modern societies. However, a substantial proportion of the plastic produced each year is improperly disposed into the environment, constituting an unprecedented pollution problem for humanity. The bulk estimate of plastic materials produced annually worldwide today is in the order of millions of tons, and it is projected to increase to the order of billions of tons by 2050 (Mbachu et al. 2020). The availability of different plastic polymer types results in a wide variety of applications, and therefore sources of MP to the environment (Jakubowska et al. 2020). The immense amount of plastic polymers produced since the 1950s, coupled with their long persistence in environmental compartments, has elicited the concept that we live in the "Plasticene" era (Haram et al. 2020).

In the environment, plastic objects such as car tires, textiles, bottles, and bags are degraded by abrasion and by the photo-oxidation of ultraviolet radiation, generating smaller fragments. If the size is less than 5 mm, they are called microplastics (Cozar et al. 2014). If it is less than 100 nm, they are called nanoplastics (Revel et al. 2018). Understanding the different MP sources is essential in prioritizing their characteristics during exposure and effects assessments and estimating toxicity risks from the data.

One of the first and more straightforward categorizations for MP regarding their source is that of "primary" or "secondary." The first category refers mainly to those particles purposely manufactured in sizes of a few mm, and most commonly composed of polyethylene (PE), polypropylene (PP), and polystyrene (PS) (Padervand et al. 2020). The "secondary" particles often refer to those produced after fragmentation or degradation of larger plastic during their life cycle or afterward, and where other materials such as polyester, acrylic, and polyamide become relevant (Padervand et al. 2020; Adam et al. 2021).

The public has mainly associated sources of primary MP with personal care products such as toothpaste (Ustabasi and Baysal 2019), but they also form part of decorative glitters, abrasive cleaners, or industrial pellets (Yurtsever 2019). In some literature, it is noteworthy that they also include fabric fibers or tire particles released by abrasion of the materials in washing machines and roads, respectively (Waldman and Rillig 2020). Broad estimates for primary MP in the ocean are close to 30% (Adam et al. 2021). Intentional production of MP still occurs in many places, even when regions such as the European Union, the United States, Canada, New Zealand, and the United Kingdom, among others, have passed or proposed legislation banning the manufacture and sale of specific products containing microbeads (Mitrano and Wohlleben 2020). Likewise, nanoplastics are increasingly common in industrial products such as paints, adhesives, drugs, electronics, and new 3D printing technologies (Stephens et al. 2013; Koelmans et al. 2015; Lambert and Wagner 2016).

Point sources of MP to aquatic and terrestrial environments include WWTP effluents and sludges, domestic (e.g., fibers from the laundry), urban (e.g., rainwater drainages carrying litter), and industrial drainages, while others may be described as more diffuse sources such as agricultural runoff or landfills (Karbalaei et al. 2018;

Iyare et al. 2020; Sang et al. 2021). Among these diffuse sources, tires and roadways are recognized as a constant source of MP from land to freshwaters and eventually to oceans, the ultimate destiny (Siegfried et al. 2017). Plastic pollution in the oceans resulting from activities carried out at sea, such as the remains of fishing nets, represents a significant problem; however, it is much less than the volume of contamination produced by activities carried out on land (Xu et al. 2020; Wang et al. 2021).

Other novel sources of MP pollution that could be relevant for human and ecological exposure, which are until recently investigated, include the use of recycled plastic in pavement and construction (Wu and Montalvo 2021; Loria-Salazar and Gomez-Sandoval 2021; Conlon 2021).

19.2 Environmental Fate and Exposure to Humans and the Environment

Microplastics and nanoplastics (MNPs) represent a very diverse class of contaminants. They occur on a continuum of sizes and in a variety of shapes. Fibers are the most frequent form, followed by particles such as beads, foam, and irregular fragments (Browne et al. 2011). MNPs have a complex composition, including polymeric materials and mixtures of associated chemicals such as plasticizers and additives, increasing their polluting effect. The surfaces of MNPs are also colonized by microbial communities that form biofilms where the transfer of genes for resistance to antibiotics can occur (Arias-Andres et al. 2018, 2019).

Many methods are used to characterize MP (e.g., electron microscopy, X-rays, magnetic resonance, and chromatography coupled to mass spectrometry). However, RAMAN spectroscopy and FTIR (Fourier transform infrared spectroscopy) are among the better suited to identify polymers in environmental compartments (aquatic, terrestrial, air), biota, and food (Veerasingam et al. 2020). In general, polyethylene (PE) and polypropylene (PP) are among the most frequently found polymers in environmental samples (aquatic and terrestrial) identified by these techniques (Zhang et al. 2020c; Veerasingam et al. 2020). However, specific polymers can be relevant in certain exposure scenarios, such as polyethylene terephthalate (PET) on household dust and indoor environments (Zhang et al. 2020c).

Freshwater ecosystems differ widely in their MP concentrations according to distance and frequency of source input. However, estimates range from dozens to thousands of particles per L or m^3 of water, with higher concentrations found by using smaller mesh and pumps for collection (Rios Mendoza and Balcer 2019). Studies on MP exposure in groundwater are scarce, even though this environmental compartment is considered an important source of water for human consumption. However, their presence has been documented and associated with what happens on the surface (Selvam et al. 2021; Luo et al. 2021). Concentrations of MP on soil range in the mg/Kg concentrations, or from dozens to thousands of items/Kg of soil (Wang et al. 2020). Air is a much less studied compartment, but concentrations range from a few to thousands of items per m^3 (Zhang et al. 2020a), and fibers seem the most relevant shape (Huang et al. 2020).

In the ocean, the final destination of a large proportion of MP, with concentrations of thousands of particles per m^2, has been determined in high retention areas (Li et al. 2020). Studies on marine sediments are disproportionately from coastal areas of Europe and Asia, showing concentrations of MP in the range of dozens to thousands of particles per L or m^2 (Phuong et al. 2021). Still, scarce information is available from unique coastal ecosystems such as mangroves, which provide food for human populations and are crucial habitats for marine biodiversity protection and recovery (Deng et al. 2021). The fate of MP in biota (some consumed by humans) has been seen to vary according to size and shape, and in some cases, polymer type affects distribution among different biological groups (Pan et al. 2021a).

19.2.1 Ingestion Pathways in Humans

MNPs can enter the human body by inhalation, ingestion, or dermal route, directly through the skin (Fig. 19.1). Once in the body, they can accumulate and exert localized particle toxicity by inducing or increasing the immune response (Galloway 2015; Wright and Kelly 2017; Teles et al. 2020).

Microplastics are abundant in the atmosphere, especially in urban environments, and therefore can be inhaled by people. For example, atmospheric measurements determined microplastics deposition rates in Central London between 575 and 1008 particles/m^2/day (Wright et al. 2020). On the other hand, studies of exposure to environmental particles among workers in the plastics and textile industry have reported lung injuries, including inflammation, fibrosis, and allergies (Wright and Kelly 2017).

The possibility of movement of the MNPs by the lung lining fluid is reduced in the upper airways, where the lining is thick. In the lungs, mucociliary clearance is likely for particles >1 µm. However, for particles <1 µm, uptake through the epithelium and its deposition in deeper parts of the lungs can be possible (Geiser et al. 2003; Ruge et al. 2013).

Oral ingestion is perhaps the primary source of entry for MNPs into the body. Evidence suggests widespread exposure to MNPs in various foods and drinking water (Catarino et al. 2018; Cox et al. 2019; Wright et al. 2020). For example, in tap and bottled water, concentrations of up to 104 particles per liter can be found (Koelmans et al. 2019). The release of particles in plastic containers containing liquids represents another source of ingestion of MNPs. These particles have even been documented in polypropylene bottles for babies (Lim 2021). Likewise, microplastics have been reported in processed foods and beverages such as sugar (Liebezeit and Liebezeit 2013), seafood (Van Cauwenberghe and Janssen 2014; Li et al. 2015), beer (Liebezeit and Liebezeit 2014), and table salt (Yang et al. 2015), among many others.

Once in the intestine, smaller particles can cross epithelial barriers. In this sense, the ingestion of MNPs of sizes between 0.1 and 10 µm may occur in the gastrointestinal tract through endocytosis by the M cells of Peyer's patches. There, M cells

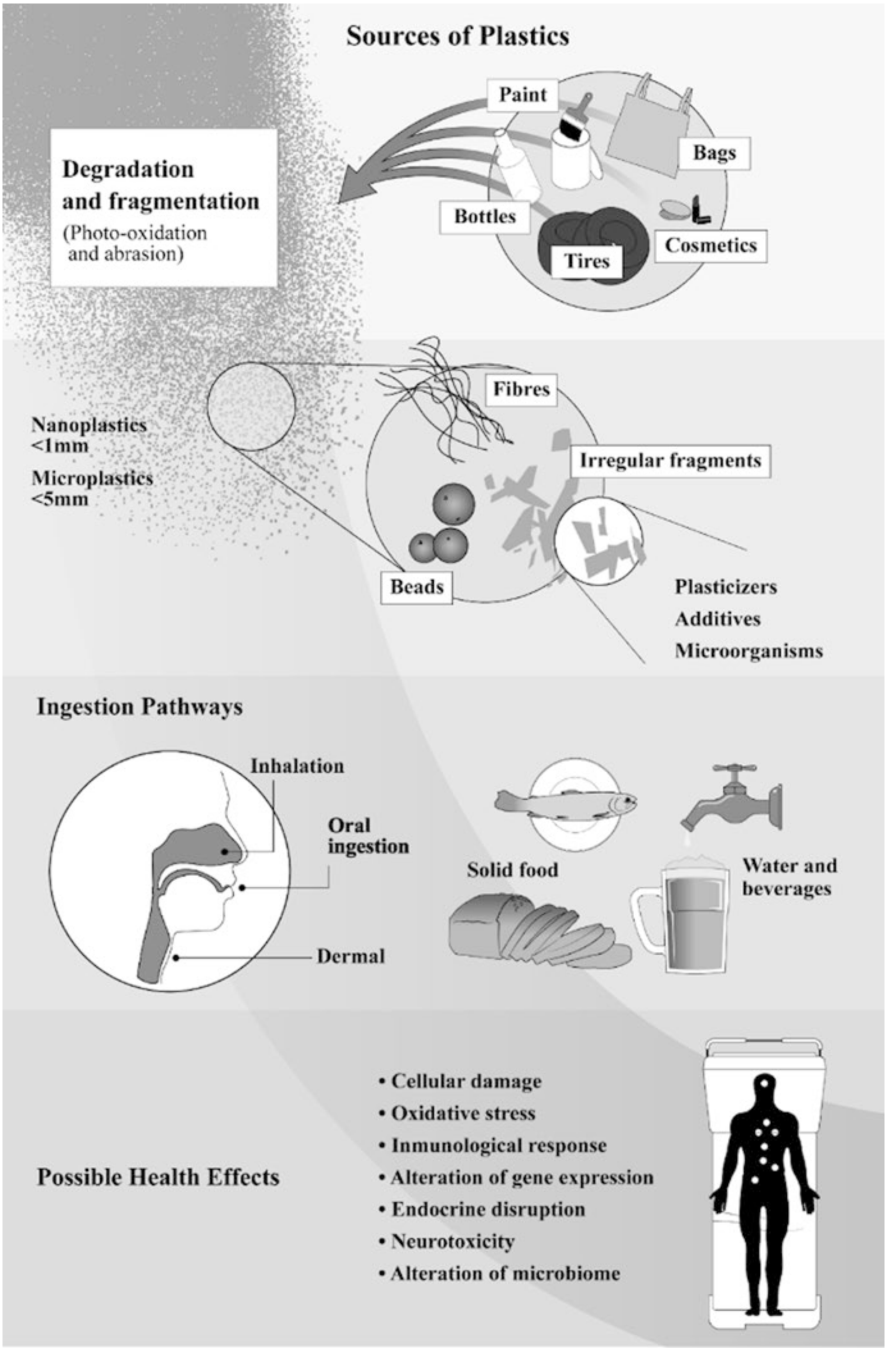

Fig. 19.1 Possible health effects of microplastics and nanoplastics (MNPs). Microplastics (<5 mm) and nanoplastics (<100 m) are generated as a result of abrasion and photo-oxidation of plastic objects such as car tires, textiles, bottles, bags, and cosmetics, which are improperly disposed of in the environment. MNPs can enter the body by inhalation, oral ingestion, and dermal routes, generating chemical, physical, and biological toxic activity. Once inside human tissues, MNPs can cause oxidative stress, immune responses, alteration of gene expression, genotoxicity, endocrine disruption, neurotoxicity, reproductive abnormalities, transgenerational effects, and behavioral abnormalities and alteration of the microbiome

can transport the particles from the gastrointestinal lumen to the lymphoid tissues of the mucosa (Mowat 2003; Wright and Kelly 2017). Likewise, it would be possible for MNPs to cross the loose junctions in the single-cell epithelial layer at the villus tips of the gastrointestinal tract, where dendritic cells can phagocytose them and transport them to the underlying lymphatic vessels and veins. MNPs can later be distributed to secondary tissues, including the liver, muscles, and (Mowat 2003; Wright and Kelly 2017).

It has been estimated that humans, on average, ingest 0.1 to 5 g of microplastics weekly through various ingestion pathways (Senathirajah et al. 2021). Many MNPs, particularly the larger ones, are excreted through the feces or after their deposition in the respiratory tract or lungs through mucociliary clearance (Wright and Kelly 2017). However, it is unknown how much of what is ingested is secreted and how much it accumulates. In this regard, although possibly a tiny part of what is ingested accumulates, the effect over time could represent an incremental risk to human health. Furthermore, it must be emphasized that human exposure to MNPs has not yet been sufficiently studied, which implies considerable uncertainty in evaluating human risk, despite its potential toxicity (Yong et al. 2020; Vethaak and Legler 2021).

19.2.2 Toxicity Profile

MNPs can exhibit a different and broader toxicity profile than other environmental particles due to their persistence, wide range of size and shapes, chemical composition, and surface charge (Wright and Kelly 2017; Ribeiro et al. 2019). Much of the research on the health effects of MNPs is based on the knowledge and lessons learned from the study of other particles, such as those associated with air pollution.

Once in contact with tissues or internalized, MNPs can cause physical, chemical, and microbiological toxicity. The toxicity effects could also be synergistic. Chemical toxicity occurs due to the leaching of endogenous additives and environmental pollutants adsorbed to the external environment. MNPs act as vectors that transfer dangerous chemicals, proteins, and toxins present in or on the particles to the human body (Hirai et al. 2011; Ribeiro et al. 2019; Koelmans et al. 2019).

When considering the entire set of substances that make up plastics, including monomers, additives, and processing aids, more than 10,000 substances have been identified. Of these, about 2400 are of potential concern due to their persistence, bioaccumulation, and toxicity (Wiesinger et al. 2021). In addition, plastics can adsorb and concentrate hydrophobic organic pollutants such as polycyclic aromatic hydrocarbons, organochlorine pesticides, and polychlorinated biphenyls (Mato et al. 2001; Ogata et al. 2009). They also accumulate heavy metals such as cadmium, zinc, nickel, and lead (Holmes et al. 2012; Rochman et al. 2014). Furthermore, the effect of chemical toxicity could be even more significant on nanoparticles, which are more effective at traversing biological membranes and have a greater surface area of chemical reactivity (Revel et al. 2018). This effect might also be enhanced by bioaccumulation.

Regarding microbiological toxicity, it has been established that MNPs can act as vectors for potential bacterial pathogens and fungi and viruses (Lu et al. 2019). Microplastic-associated microorganism communities can alter the structure of endemic communities where they are deposited, for example, the gastric mucosa of the intestine. Likewise, biofilms formed on microplastics favor horizontal gene transfer in bacterial communities, including antibiotic resistance genes (Arias-Andres et al. 2018). Therefore, MNPs can directly impact the gut microbiome, which, according to recent research, has been related to human health in multiple aspects (Fackelmann and Sommer 2019; Fournier et al. 2021; Lear et al. 2021).

In addition, nanomaterials have been attributed antimicrobial activities due to their unique physicochemical properties such as ultrasmall size, large surface area to mass ratio, high reactivity, and functionalizable structure (Shimanovich and Gedanken 2016). Therefore, it will be imperative to determine in the future if nanoplastics could have a direct toxic effect on commensal bacteria in the digestive tract and therefore cause alterations in the human microbiome.

Physical toxicity refers to the effect of the presence of the particles, as foreign agents, in the tissues. For microplastics <10 μm, the translocation from the intestinal tract to the lymphatic and circulatory systems is possible, causing systemic exposure and accumulation in tissues including the liver, kidney, and brain. The translocation and accumulation could lead to oxidative stress, cytokine secretion, cell damage, inflammatory and immune reactions, and DNA damage (Yong et al. 2020).

19.2.3 Possible Effects to Human Health

The potential impact of MNPs on human health became a matter of public concern until recently, despite the evidence from ecotoxicological studies both in vivo and in vitro showing adverse effects on other organisms (Noventa et al. 2021).

As noted, MNPs enter the human body by different ingestion pathways. A large part can be excreted, but a small part remains, temporarily or permanently. As a result of the toxic activity of MNPs, either physical, chemical, or microbiological, the host cells will suffer various effects, among which oxidative stress has been noted through the generation of free radicals. Other effects of MNPs are immune responses, alteration of gene expression, genotoxicity, endocrine disruption, neurotoxicity, reproductive abnormalities, transgenerational effects, and behavioral abnormalities (Alimba and Faggio 2019; Hwang et al. 2020).

However, many aspects related to the fate and effects of MNPs on the human body are still unknown; little is known about dose-dependent effects, adsorption mechanisms through membranes, the translocation pathways to secondary tissues and organs, the impact of the cumulative effect of chronic exposure, as well as natural elimination processes (Academies 2019; Noventa et al. 2021).

Table 19.1 Examples of methods applied to characterize ecological risks from MP in recent literature

Environmental compartment/biological group or function in effects assessed	Hazard/effects assessment	Risk characterization methods	Results	References
Coastal, continental shelf, and deep-sea areas Aquatic invertebrates, primary producers, fish, and corals	Effects on mortality, growth, development, reproduction, and population growth	Predicted no-effect concentration (PNEC) by analysis of species sensitivity distributions (SSD) of chronic toxicity endpoints (LOEC, NOEC, EC10), to derive a hazardous concentration for 5% of the species (HC5) and using an assessment factor, for 20–300 μm sized MP vs. exposure field data	Detected MP concentrations did not exceed the derived PNEC of 12 particles/L	Jung et al. (2021)
Estuarine ecosystem	Hazard classes and categories in the EU classification and labeling (CLP) regulation based on the UN Globally Harmonized System	Hazard score (H) assigned to MP polymer and calculation of polymer composition and pollution load (PL) from exposure field data (0.3–2.5 mm MP)[b] Hazard level and ecological risk index calculation for single and combination of polymers	Hazard Level II for MP pollution in the estuarine ecosystem (acute toxicity, skin corrosion/irritation, serious eye damage/eye irritation, specific target organ toxicity — single exposure, hazardous to the aquatic environment, explosives[a,b]) "Minor" ecological risk category of studied site[b]	Lithner et al. (2011) Pan et al. (2021b)
Global ocean	Effects on population size (e.g., growth, reproduction, and survival)	SSD of toxicity thresholds (TT, mg/L) calculated for MP size ranges Comparing the thresholds of biological effects with the probability of exposure to those concentrations	Large MP poses a negligible global risk MP bioavailability, translocation, and toxicity increase as size decreases, and particles <10 μm are not identified by current monitoring methods	Beiras and Schönemann (2020)

Environmental compartment/biological group or function in effects assessed	Hazard/effects assessment	Risk characterization methods	Results	References
Freshwater, river Algae, invertebrates, and vertebrates	Growth, mortality, reproduction, feeding, morphology, neurological effects, histopathology, biochemical effects, microbiota diversity, innate immune response, predatory performance, digestion, and energy production	PNEC by analysis of SSD of chronic toxicity endpoints (NOEC), using the HC5 as criteria and using an assessment factor, vs. exposure field data in a risk quotient (exposure/PNEC)	Most of the monitored sites posed negligible risks to freshwater biota, except for two sites in the urban center	Zhang et al. (2020b)
Freshwater Pelagic and benthic freshwater organisms (invertebrates, algae, plants, and fish)	Effects on survival, growth, reproduction, and changes in photosynthesis	PNEC by analysis of different percentiles of SSD using chronic toxicity endpoints (NOEC) (>30% of ecotoxicological studies with MP <0.45 μm Exposure probability distribution based on literature data (Europe, North America, and Asia regions) Risk characterization ratios (RCR) by comparison of probability distributions of PNEC and exposure data for each region	No influence of particle shape or type of polymer on the NOEC The probability distribution of the global exposure concentration overlapped to a small extent with the PNEC probability distribution. A 0.12% probability distribution calculated for the global RCR was >1 but related only to exposure data from Asia	Adam et al. (2019)

NOEC: no observed effect concentration, *LOEC:* lowest observed effect concentration, *EC10:* effect concentration to 10% of the exposed population

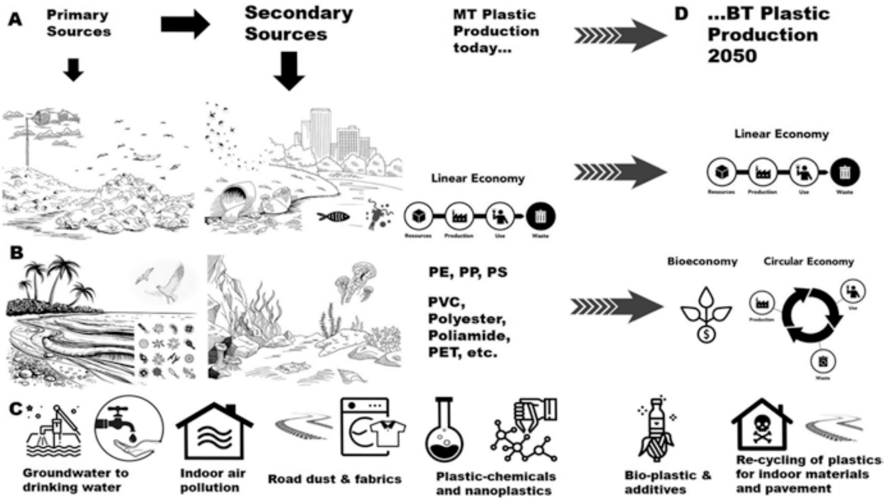

Fig. 19.2 Exposure and ecological scenarios for assessing effects of MP today and in the future. (**a**) Today, millions of tons of plastics are produced per year, and a large proportion of MP comes from waste after using plastic objects. (**b**) Most effects are known for freshwater and marine species, but terrestrial compartments should be considered further. (**c**) Some types of environmental exposure require more attention, such as groundwater pollution and its impact on drinking water, long-term MP inhalation on indoor spaces, and effects emphasizing high and frequent exposures such as those from MP of road dust and fabrics washed at home. Further, the effect and risk assessment should focus on the smaller particles and the interaction with the large number of chemicals added to plastics. (**d**) In the future, billions of tons of plastic will be produced annually. If the economic model of waste production continues, an exponential increase in exposure and effects will be expected. In the scenario of a transition to a circular-like (bio)economy, new bio-based materials and additives are expected to be generated and produced. In this latter case, it will be essential to assess the toxicological profile for the new generation of materials, including toxicology regarding recycled plastic (e.g., the plastic used in pavement and the exposure to plastic-toxic substances in construction materials)

19.3 Ecological Risks Estimated from MP Exposure Data

The magnitude of contamination by MNPs, their ubiquity on the planet, and their high persistence over time raise serious concerns about their effects on ecosystems, wildlife, and human health. However, the type and level of exposure and the specific risks for humans are far from being elucidated (Vethaak and Legler 2021).

Information regarding toxic effects of microplastics on organisms from different ecosystems, i.e., ecotoxicological effects, increased exponentially over this decade (Anbumani and Kakkar 2018; Zhu et al. 2019). The assessments for effects include data on the traditional ecotoxicological endpoints of mortality, growth, development, reproduction, and population growth. Most of the published data is originated from aquatic species such as invertebrates, algae, plants, and fish. Accordingly, most of the ecological risk assessments (ERA), analyzing the risks of effects based on current exposure levels, were developed for aquatic environments, and many of them with exposure data from Asia. Some examples of standard methods applied in ERA for MP are provided in Table 19.1.

The first ERA show that MP can pose toxicity risks in ecosystems but is relatively low in terms of the toxicity endpoints. However, results also indicate that underestimating MP concentrations (exposure assessment) due to size during sampling can affect the results of risk assessment characterization (Covernton et al. 2019). It is also important to consider that the assessments of effects often do not look into toxicity mechanisms, such as the production of reactive oxygen species (ROS) and immune reactions, changes in animal microbiota, or genotoxicity (Jeong and Choi 2019; Palmer and Herat 2021). In addition, MP is a source for exposure to other chemical substances, many of which are emerging contaminants with most likely long-term effects, but this is still not analyzed during ecological risk assessments (Fig. 19.2).

Chemical substances in MP include those added to plastics, those originating from plastic polymers, and others taken up by plastic in the environment (Campanale et al. 2020). For example, plastics are known to contain or absorb flame retardants, plasticizers, lubricants, and dyes that are made of organic and inorganic forms of heavy metals, different organic pigments and chromophores, salts of stearic acid, halogens such as chlorine and bromine, bisphenol A, phthalates, and pesticides, among many others (Campanale et al. 2020; Bhagat et al. 2021). Many of these substances are known or presumed to be toxic, carcinogenic, and endocrine disruptors (Mohamed Nor et al. 2021). Many additives in plastics are non-covalently added; therefore they can leach to the surrounding environment (Gewert et al. 2015).

The interaction of MP and these substances forms complex contaminants that require more sublethal and chronic ecotoxicological methods that address more complex and long-term responses (e.g., immune responses, microbiomes, and their activities). The study of these interactions is particularly important for ecological and human risk assessments intended for upcycling and recycling plastics and circular economies.

19.4 Conclusions

In this chapter, we analyze the sources and processes that generate MP pollution and their environmental fate. We also indicate the type and level of exposure and the specific risks for humans and the ecosystem, for example, the routes of ingestion, the toxicity profile, and possible health effects. Further research is needed to understand the toxicity, mechanisms, and long-term effects of human exposure to MP. The ecological risks associated with MP exposure also need to (1) emphasize the smaller fraction of MP, (2) provide more data regarding MP-emerging contaminant interactions, and (3) utilize more long-term, chronic, and sublethal endpoints to assess these interactions. Due to the complexity of the problems associated with MP pollution, it will be critical that research challenges will need to be addressed by multidisciplinary teams, including biologists, chemists, physicians, environmentalists, economists, politicians, sociologists, and philosophers. Furthermore, we propose that the solution to a problem as complex as contamination by MP will be political, economic, and social rather than technical.

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Part IV
Treatment Technologies

Chapter 20
Microplastics as an Emerged Contaminant and Its Potential Treatment Technologies

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Abstract Microplastics are widely distributed across the different environments by means of which they affect the various living organisms. This distribution majorly arises from dumping of plastic wastes into landfills and sewage treatment systems. This chapter focuses on the occurrence of microplastics in the environment and its treatment technologies. Landfills serve as the main source of plastic pollution in the environment. It slowly disintegrates into microplastics by weathering processes. Sewage treatment plants are effective in removing up to 99.2% microplastics from the sewage. However, further investigation is needed to develop treatment technologies for the removal of microplastics from landfills and for the complete removal (100%) of microplastics from sewage.

Keywords Microplastics · Environment · Organisms · Landfills · Sewage treatment plants

20.1 Introduction

The ever-increasing accumulation of plastic debris has demanded human attention (Avio et al. 2017a). It was projected that about 4.9 billion tons of discarded plastic has reached the environment (Geyer et al. 2017). These discarded plastics undergo

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abrasion and abiotic degradation and eventually break down into smaller particles called as microplastics (Klein et al. 2018). Though microplastics are generally defined as particles of <5 mm, Hartman has suggested that the upper size limit needs reconsideration (<1 mm). Frias and Nash (2019) have defined the microplastics size range as 1 μm to 5 mm.

According to their sources, microplastics are classified into two types as primary and secondary microplastics (Li et al. 2018c; Pinon-Colin et al. 2018). Primary microplastics are manufactured in the desired size (<5 mm) (Cole et al. 2011) and are commonly found in synthetic textiles and cosmetic products, while secondary microplastics are formed up by the destroying bigger plastics (Ling et al. 2017).

The frequently used plastic polymers include polystyrene (PS), acrylic, polyethylene (PE), polyamide (PA) (nylon), polypropylene (PP), polyvinyl chloride (PVC), polyvinyl alcohol (PVA), polyethylene terephthalate (PET), and polyester (Mathalon and Hill 2014; Avio et al. 2017b) which have diverse applications while the mono polymers are used in different materials. The shapes such as fibers and pieces are the most frequently observed microplastics ((De Sá et al. 2018).

Globally, microplastics are considered as a promising threat (PlasticsEurope 2018; Rochman et al. 2019; Zeng 2018). They are abundantly present in diverse environments such as air (Abbasi et al. 2019), soil (Guo et al. 2020), marine (Wang et al. 2020), fresh water (Han et al. 2020), and arctic lakes (Gonźalez-Pleiter et al. 2020) while they can also absorb an array of pollutants (Sørensen et al. 2020; Singla et al. 2020). Exposure to these pollutants-adsorbed microplastics leads to chronic toxicity within organisms (Li et al. 2018b).

Plastic is widely distributed in a variety of environments, affecting a wide range of organisms. This distribution mainly arises from the dumping of plastic waste into landfills and sewage treatment plants. The present chapter deals with microplastics pollution and its cause on the landscape, aquatic, atmospheric, and treatment modalities.

20.2 Microplastics Occurrence in Various Environmental Media

20.2.1 Terrestrial Media

Plastics are readily manufactured and easily disposed of in the terrestrial regions where they undergo various treatments such as UV and increased temperature (Ng et al. 2018) and enter the soils via pathways such as water infiltration (O'Connor et al. 2019) weather conditions or tilling (Rillig et al. 2017a) and soil biota (Rillig et al. 2017b).

The common microplastics sources from terrestrial environments are tire abrasion, sewage effluents, landfills, and agricultural, domestic, and industrial discharges (Chae and An 2018). Laundry washing is the chief contributor of fibers from terrestrial environments since it contributes around 35% of the total microplastics load in the marine environment as observed by Boucher and Friot (2017). Brand et al. (2018) opined that several landfills are located in low-lying areas which tend to get flooded and subsequently disperse the microplastics in the oceans. Kole et al. (2017) pointed that about 5–10% of microplastics disposed in the marine environment are contributed by tire abrasion.

Panno et al. (2019) observed microplastics from groundwater and proposed that these may have arisen from drainage/septic tanks or wastewater treatment plants as these effluents are introduced into aquifers to improve groundwater quantity. They further opined that wastewater from oil and gas production plants are also let out into aquifers from groundwater replenishment. Hence, microplastics occur in the groundwater due to accidental or purposeful discharge of effluents into the aquifers.

Yukioka et al. (2020) studied the presence of microplastic in surface dust on roads. Their observation revealed that microplastic abundance on the road ranged from 2.0 ± 1.6 pieces/m^2 to 19.7 ± 13.7 pieces/m^2 in their study areas. Their microplastic composition is comprised of packaging and rubber materials. The packaging materials were dominant in the regions near dumping sites, while rubber materials arose from the abrasion of tires. The microplastics composition is dependent on the waste management practices employed in the particular area.

20.2.2 Aquatic Media

Sekudewicz et al. (2020) studied the microplastics abundance in the Vistula River (Poland). Microplastic abundance in the water was 1.6–2.55 items/L, and in the sediments, it was 190–580 items/kg. The study proposed that microplastic abundance will be high in the metropolitan regions than that of the less urban regions. In another study done by Zhang et al. (2020), they observed microplastics in the Lijiang River (China). Approximately 67.5 ± 65.6 items/m^3 items, 0.67 ± 0.41 items/m^3, and 0.15 ± 0.15 items/m^3 microplastics were retrieved from plankton nets of 75 μm and 300 μm mesh sizes. Likewise, several other studies have projected the presence of microplastics in different aquatic systems (Zhu et al. 2018; Zhang et al. 2019).

20.2.3 Atmospheric Media

In terrestrial regions, microplastics have been observed in atmosphere, soil, and dust. Several studies have confirmed the occurrence of microplastic polymers in air and dust as reviewed by Lucattini et al. (2018). This review primarily deals with the presence of atmospheric microplastics within households. However, atmospheric microplastics have also been discovered in workplaces (Suzuki et al. 2009). PET and polyester fibers are found in the indoor and outdoor environments that children are more prone to ingest (Liu et al. 2019b).

The atmospheric transfer of microplastics has also been reported by several authors. Dris et al. (2016) calculated that synthetic fibers of about 3–10 metric tons per annum may be deposited atmospherically. Allen et al. (2019) suggested that rain and snowfall may aid in microplastic deposition, while they also noted that microplastic fibers may be atmospherically transported to several thousand kilometers.

Wright et al. (2020) conducted an analysis to monitor microplastics transport through the air. In their study, they collected samples from high-rise buildings and observed 575–1008 microplastics/m^2/day. They found 15 different petroleum-based polymers that were present due to airborne transfer. Their study suggests that the atmospheric exchange of microplastics is also an important pathway.

20.3 Microplastics in Biota

20.3.1 Fishes

Numerous investigations have confirmed that aquatic fishes have consumed microplastics. Karuppasamy et al. (2020) observed microplastics contamination in 17 specimens of economically important edible fishes out of 190 collected specimens offshore regions of Chennai and Nagapattinam. They retrieved a total of 20 microplastic particles from the specimens. These coasts indulge in multiday fishing; hence, the fishes are degutted immediately after its capture. This is the primary reason for its low contamination rate. In a study conducted by Naidoo et al. (2020), 52% of the 172 sampled specimens were contaminated with microplastics, which were dominated by fibers. Similarly, several researchers Al-Salem et al. (2020) and Koongolla et al. (2020) have also confirmed the presence of microplastics in fishes.

20.3.2 Mollusks

Hariharan et al. (2021) examined the toxicity effect of weathered polyethylene on *Perna viridis*. After 30 days of exposure, the organisms should reduce feeding rates; however, there was no mortality. The study concluded that the gills, adductor muscles, and foot tissue were sensitive to microplastics. Dowarah et al. (2020) observed the presence of microplastics in the body tissues of *Perna viridis* and *Meretrix meretrix* collected from three estuaries of Pondicherry coast, India. They predicted that an average person from these areas may uptake about 3917.79 ± 144.71 Nos of microplastic particles/year by consuming these mussels. Revel et al. (2020) observed the occurrence of microplastics in the tissues of Pacific oyster *Crassostrea gigas;* however, they did not notice any biological effects in the organisms. Many authors have also studied the occurrence of microplastics on mollusks. Some of them are Scott et al. (2019), Li et al. (2018a) (*Mytilus edulis*), Abidli et al. (2019) (commercial mollusks), Arossa et al. (2019) (*Tridacna maxima*), and Scanes et al. (2019) (*Saccostrea glomerata*).

20.3.3 Crustaceans

Wang et al. (2021) observed that water filtration and food consumption equally contribute to microplastic accumulation within the marine crab (*Charybdis japonica*). Microplastics have been observed in various organs such as hepatopancreas, gut, gills, and muscles that may have been translocated from the intestine. Hara et al. (2020) detected the microplastics in the gastrointestinal tract of *Nephrops norvegicus* and estimated that about 15–4471 particles of microplastics enter the human body via consumption of this crustacean. A number of former studies have confirmed the occurrence of microplastics in edible crustaceans (Devriese et al. 2015; Welden and Cowie 2016; Abbasi et al. 2018).

Hara et al. (2020) detected the presence of microplastics in the gut of *Nephrops norvegicus* and estimated that approximately 15–4471 particles of microplastics transferred into the human body by ingesting this meal. Several previous studies have reported the presence of microplastics in edible crustaceans (Devriese et al. 2015; Welden and Cowie 2016).

20.3.4 Other Biotic Forms

Ajith et al. (2020) reviewed that among the aquatic organisms, fish and mollusks were easy to examine; hence, most of the studies were conducted on them. However, other organisms such as phytoplankton, zooplankton, echinoderms, and birds are also known to consume microplastics.

In addition to aquatic organisms, terrestrial organisms such as chickens are also contaminated by microplastics.

Lwanga et al. (2017) found microplastics from earthworms and poultry gizzards and crops of chicken. The review by Walkinshaw et al. (2020) stated that low tropic organisms are more susceptible to microplastic contamination and are at risk of tropic transmission that eventually reaches humans.

20.3.5 Other Consumables Contaminated by Microplastics

Microplastics were recovered from processed and unprocessed salts collected from salt pans (Nithin et al. 2021). The study suggested that microplastics contamination may occur while packaging processed salts, and its presence in unprocessed salts is owed to the usage of salt brines from the estuaries. Li et al. (2020) observed microplastics uptake by the xylem vessels of the wheat plants. Liebezeit and Liebezeit (2014) observed microplastic contaminants in honey which may have occurred during its packaging. Likewise, Kosuth et al. (2018) observed microplastics from beer and drinking water and proposed that microplastic contamination may have occurred during processing or packaging.

20.4 Health Impacts

Microplastics are known to accumulate several harmful toxicants such as heavy metals, persistent organic pollutants (POPs), and polychlorinated biphenyls (PCBs). These toxicants may be leached into the tissues of animals ingesting them, which may further be transferred across the trophic levels (Auta et al. 2017; Andrady 2011). When humans consume microplastic contaminated seafood, these toxicants may cause several health impacts (Van Cauwenberghe and Janssen 2014); however, much information is not available regarding these impacts (Farady 2019).

Some reports have revealed the toxic effects of microplastics, including antioxidant pressure (Barboza et al. 2018; Brandts et al. 2018a; Pitt et al. 2018; Qu et al. 2018; Tang et al. 2018; Liu et al. 2019a), immunological responses (Brandts et al. 2018b; Revel et al. 2018; Tang et al. 2018), altered genetic expression (Sleight et al. 2017; Brandts et al. 2018a, b; Liu et al. 2019a, b, c), genotoxicity (Brandts et al. 2018a), disrupted endocrine activity (Rochman et al. 2014), neurotoxicity (Barboza et al. 2018), and reproductive abnormalities and trans-generational effects (Gardon et al. 2018; Tallec et al. 2018; Pitt et al. 2018; Martins and Guilhermino 2018; Liu et al. 2019a).

It was estimated that humans consume about 39,000–52,000 particles/year only from foodstuffs (Cox et al. 2019). Likewise, Prata (2018) reported that humans may inhale about 26–130 microplastics/day. However, these estimates may vary based on the evaluation methods. Microplastics enter a human's gastrointestinal tract through contaminated food or by inhalation and enter into the gastrointestinal tract causing a possible inflammatory response (Salim et al. 2013). Small size and less dense particles are known to enter the lungs, while the activity of macrophages causes particle translocation to the circulatory system (Prata et al. 2019a). Though certain possible health impacts are reported, they remain largely unproven.

20.5 Treatment Technologies

Microplastics primarily originate due to poor terrestrial waste management (Jambeck et al. 2015). This improper management leads to magnifications of microplastic contaminants in the environment, which causes a hazard to the ecosystem (Prata et al. 2019b). Studies on microplastics pollution in the land sources are still at the elementary levels particularly regarding the effect of long-term human activities, i.e., landfills and dumping (Xu et al. 2020). Geyer et al. (2017) observed that from the total plastic wastes, 79% reaches the landfills, 12% is incinerated, and 9% is recycled. Landfills (Hahladakis et al. 2018) and sewage treatment plants (Hou et al. 2021) receive their plastic wastes from domestic and industrial discharges. Comparatively, landfills receive magnanimous amounts of plastic wastes from the same sources (Duis and Coors 2016).

20.5.1 Landfills

Most of the plastic wastes are dumped in the land, within the landfills, and macroplastics disintegrated and degrade into microplastics in the absence of sunlight and oxygen due to the presence of variable temperature (60–90 °C), pH (4.5–9), and microbial activity (Mahon et al. 2016; Sundt et al. 2020). Some landfill plants treat the leachates to remove the pollutants (Sui et al. 2017). This procedure can be practiced in all facilities to improve the treatment and eradicate microplastics.

Silva et al. (2021) suggested that studies of microplastic contamination from landscapes and leaks may be too limited due to the difficulty of the samples and the limitations in the analysis techniques.

Silva et al. (2021) opined that studies on microplastic contamination from landfills and leachates are very limited probably owing to the difficulty of samples preparation and limitation in the analyzing methods.

The landfill treatment includes various stages such as aerobic decomposition, anaerobic decomposition, hydrolysis, methanogenesis, and stabilization (USEPA 2007). In each phase, the rate of plastic degradation increases. Further in conditions of heat, moisture, enzymes, and soil, polymers become shorter with weak chains which facilitate plastic fragmentation (Horton et al. 2017). At the initial stage, an increase in humidity and temperature (up to 70 °C) contributes to the polymer breakdown (Hanson et al. 2009).

The review of da Costa et al. (2020) elaborated the existing landfill treatment practices practiced by many of the European countries. These practices include ban of single-use plastics, recycling policies, trash-sorting policies, and limiting the use of plastics. Another set of waste-filling practices involves the excavation of waste and the conversion of plastic waste into energy for the use of plastics as secondary raw materials (Avolio et al. 2019).

Table 20.1 Existing treatment technologies employed in landfills

Sl. no.	Location	Type of waste	Type of landfill leachate	References
1.	Helsinki, Finland	MSW IW	None	Kilponen (2016)
2.	Turku, Finland	MSW	None	Praagh et al. (2018)
3.	Fifholt, Iceland	MSW IW	Sand bed Filtration	
4.	Skedsmokorset, Norway	MSW IW	SBR	
5.	Southwest Finland	IW	Filtration and active carbon	
6.	Lahti, Finland	MSW IW	Artificial soil	
7.	Shanghai, China	MSW	No information	He et al. (2019)
8.	Wuxi, China	MSW	No information	
9.	Suzhou, China	MSW	No information	
10.	Changzhou, China	MSW	No information	

Abbreviations: *MSW:* municipal solid waste, *IW:* industrial waste, *SBR:* sequencing batch reactors

Recovery of plastics from landfills could produce 19–28 MJ of energy that can be used as an alternative treatment method to contaminated plastic waste (Quaghebeur et al. 2013). Similarly, Hou et al. (2021) suggested that microplastics can be converted into oil under optimum conditions by employing supercritical water which is an energy-saving process and produces lesser greenhouse gases than incineration. This conversion method can be employed globally while the converted oil may be used as a source of fuel (Table 20.1).

20.5.2 Sewage Treatment Plants

The level of microplastics in wastewater treatment plants is dependent on numerous factors, i.e., the source of water, population, economy, and lifestyle. Some other factors such as the extraction and detection processes, volume treated, and errors in assessing the density of microplastics contribute to the difficulties in estimating the efficacy of wastewater treatment plants (Liu et al. 2021).

Domestic and industrial discharges, stormwater runoffs, and sewages are the chief sources of sewage treatment plants (Hale et al. 2020). The preliminary step of sewage treatment plants is initial screening where the larger particles such as sand and grit are filtered. The next step involves surface skimming and solids settling, viz., the primary treatment. The secondary treatment includes aerobic digestion and further solids settling. Particle sedimentation is enhanced by the addition of inorganic flocculent. Within these primary and secondary treatments, 90–95% of microplastics are removed. However, this is depending on the effectiveness of the treatment plants (Carr et al. 2016; Raju et al. 2018).

Liu et al. (2021) reviewed and submitted a meta-analysis of the existing treatment technologies comprising of primary, secondary, and tertiary treatments. The primary treatment processes include primary settling treatment, grit, and grease treatment, while the secondary treatment processes comprise A2O, biofilters, and other bioreactors, whereas the tertiary treatment processes consist of UV, O3, chlorination, biologically active filters (BAFs), disc filters (DFs), and rapid sand filters (RSFs). It was observed from their meta-analysis that after the primary treatment, there was a decrease of 4.06–98.96% of microplastic densities. It has been suggested that biological treatment as division of the secondary treatment was the vital step and A2O most broadly used method. They further noted that following secondary step, there was a further decrease of 20.45–95.45% of microplastics. Some treatment plants included advanced oxidation and membrane filtration processes as tertiary treatments which were able to reduce 85.71% of microplastics after treatment.

Accordingly, filter-based technologies such as biofilter, ultrafiltration, and rapid sand filtration (RSF) have been suggested as the best methods to remove microplastics from wastewater. In total, after primary and secondary treatments, 99.1–99.2% of microplastics will be removed as claimed by Hidayaturrahman and Lee (2019) in their study. However, Ajith et al. (2020) recommended that the sewage treatment system must include advanced tertiary treatment devices to achieve 100% removal of microplastics.

Flocculation is a vital step in the primary treatment process. During this process, microplastics interact with flocs by means of hydrogen bonds and electrostatic pressure (Duan and Gregory 2003; Lapointe et al. 2020). The flocs and microplastics aggregate due to the Brownian motion and mechanical agitation (Larue et al. 2003). The commonly used flocculants are iron-based and aluminum-based salts (Ma et al. 2019). These flocculants adhere to the microplastics surface and cause them to settle to the bottom. These settled particles are removed as sludge (Murphy et al. 2016), while the non-settling microplastics are precipitated to the flocs and eliminated as scum (Lee et al. 2012). This process is effective in the elimination of microplastics from the top layer as a sum and also at the bottom as sludge. However, its efficiency in removing microplastics from the water column has not been identified. The possibility of microplastics to release harmful chemicals on their interaction with iron-based and aluminum-based salts is also not studied. Therefore, these factors should be considered to improve the effectiveness of flocculation.

Membrane bioreactor is an excellent technology that is capable of removing 99.9% of microplastics from the sewage (Talvitie et al. 2017). This technology involves the adsorption mechanisms which enhance the separation of microplastics. These membranes have a pore size of 0.1 μm which facilitates maximum removal (Li et al. 2020). Microplastics are retained in the biofilm where adsorption takes place. This is an efficient system that yields maximum recovery of microplastics. However, these membranes do not guarantee long-term usage since the constant water transfer resulted in the formation of a polarization layer (Enfrin et al. 2020).

Table 20.2 Existing treatment technologies employed in wastewater treatment plants

Sl. no.	Location	Capacity (m^3/day)	Treatment process	Source	References
1.	Australia	65,000	Sec	Municipal	Ziajahromi et al. (2021)
2.	Scotland, UK	166,422	Pri, Sec, and Ter (nitrification)	Municipal	Blair et al. (2019)
3.	Cartagena, Spain	35,000	Pri and Sec	Municipal and Industrial	Bayo and Olmos (2020)
4.	Madrid, Spain	28,400	Sec (A^2O)	Municipal	Edo et al. (2020)
5.	Hong Kong, China	2,400,000	Pri and Ter (chlorination)	After primary treatment	Ruan et al. (2019)
6.	Daegu, Korea	26,545	Pri, Sec, and Ter (coagulation, O3)	Municipal and industrial	Hidayaturrahman and Lee (2019)
		469,249	Pri, Sec, and Ter (coagulation, DF)	Municipal and industrial	
		20,840	Pri, Sec, and Ter (coagulation, RSF)	Municipal and industrial	
7.	Wuhan, China	20,000	Pri, Sec (A^2O), and Ter (chlorination)	Industrial, agricultural, and municipal	Liu et al. (2019c)
8.	Wuxi, China	50,000	Pri, Sec (OD), and Ter (UV)	Municipal	Lv et al. (2019)
9.	Wuxi, China	70,000	Pri and Sec (A^2O + MBR)	Municipal	
10.	Helsinki, Finland	–	Ter (DF)	Municipal	Talvitie et al. (2017)
11.	Turku, Finland	–	Ter (RSF)	Municipal	
12.	Hameenlinna, Finland	–	Ter (DAF)	Municipal	
13.	Mikkeli, Finland	–	Ter (MBR)	Municipal	
14.	Paris, France	240,000	Pri and Sec (biofilter)	Municipal and industrial	Dris et al. (2015)
15.	Los Angeles, USA	–	Ter (Centrata thickening)	–	Carr et al. (2016)
16.	Oldenburg, Germany	35,616	Ter (PF)	Municipal and industrial	Mintenig et al. (2017)

(1) Pri, Sec, and Ter refer to primary treatment, secondary treatment, and tertiary treatment. (2) *A2O*: anaerobic-anoxic-oxic; *A/O*: anoxic oxic; *OD*: oxidation ditch; *DF*: disc filter; *RSF*: rapid (gravity) sand filter; *DAF*: dissolved air flotation; *BAF*: biologically active filter; *GF*: gravity filter; *PF*: post-filtration; *SAF*: sand filter; *UF*: ultrafiltration

This reduced the filtration efficacy by accumulating microplastics and other solutes on the membrane surface (Baker 2012). More research is required to understand the formation of the polarization layer and limit its impact on the performance of membrane filters (Table 20.2).

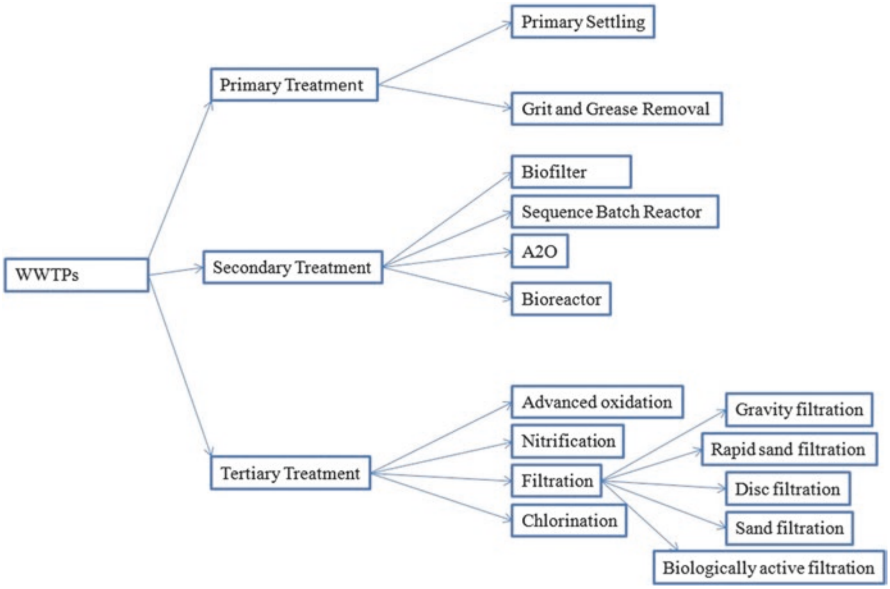

Fig. 20.1 Some of the existing treatment technologies employed in wastewater treatment plants

To facilitate treatment solution, it is necessary to understand the morphological features and the physical and chemical properties of microplastics. To retrieve microplastics from the surface waters, numerous methods have been used. Hale et al. (2020) employed a net of 300 μm mesh size. However, Chae et al. (2015) used a 20 μm mesh size hand net to collect microplastics. Murphy et al. (2016) opined that microplastics of less than 100 μm will not be removed by sewage treatment plants. Hence, microplastic sizes are important criteria to be considered.

Likewise, shapes are also essential criteria to be noted in the treatment process. Fibers are easily eliminated during the primary treatment (Sun et al. 2019). Fragments are easily removed during the secondary step of treatment process (Jeong et al. 2016). But pellets were removed only during the tertiary treatment process where filter-based and advanced oxidation treatments aid in its removal (Liu et al. 2021).

Microplastics vary in their morphology and chemical composition which are detrimental to their environmental distribution. After its discard, microplastics behave differently compared to its usage (Hale et al. 2020). Hence, all these factors are to be taken into consideration during its treatment. Talvitie et al. (2017) suggested some methods such as rapid sand filtration, membrane bioreactor, disc filter, and dissolved air flotation to improve the efficiency of sewage treatment plants. Some treatment plants use bio-beads as biological filters to remove microplastics from wastewater. However, these beads form a biofilm that digests the organic matter present in the sewage (Hale et al. 2020). Moreover, these biofilms facilitate colonization of bacteria which may be harmful or beneficial (McCormick et al. 2014). The protozoa and metazoan present in the biofilms are known to consume microplastics (Scherer et al. 2018); hence, these also influence the treatment process.

Polyesters and polyamide are the most dominant polymers that are retrieved from sewage treatment plants (Murphy et al. 2016; Talvitie et al. 2017). Other polymers such as polypropylene (PP), alkyd, acrylic, and polystyrene are also found; however, the composition of polymers is dependent on the input sources. Since a majority of the polyesters arise from synthetic fibers of clothes, Hou et al. (2021) suggested the use of additional filters in washing machines to remove these microplastics at the sources. The treatment plants should consider the implementation of polymer-based removal systems targeting the commonly available polymers (Fig. 20.1).

20.6 Conclusions

Microplastics have been quoted as emerging pollutants because of their ability to cause chronic toxicity. Microplastic research has gained an impetus over the past decade as it is widely distributed in a variety of environments and is known to affect numerous organisms, including humans. Though the information on the effect of microplastics on human beings is not completely known, there is enough evidence that they are harmful contaminants. Hence, touting these toxic materials as emerging contaminants will be an understatement. Therefore, microplastics must be considered as emerging contaminants. Landfills are a source of microplastics since a large amount of plastic wastes is dumped. Macroplastics can be visually sorted and removed through the recycling process. However, microplastics can be treated only by implementing landfill leachate treatment plants. Landfill mining is effective in reducing macroplastics; however, advanced treatment technology is required to treat microplastics. Data on the treatment of landfill leachates is limited. However, it is understood that there are no effective methods to treat microplastics from landfill leachates. Sewage treatment plants have several advanced mechanisms to remove microplastics. Among these several methods are eco-friendly and cost-effective. However, none of the existing methods yield 100% removal of microplastics. Therefore, the authors suggest that future research needs to focus on analyzing the flaws associated with each treatment mechanism and improving its efficiency in the complete removal of microplastics. As suggested, the eliminated microplastics can be converted into oils and used as fuels.

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Chapter 21
Green Treatment Technologies for Microplastic Pollution

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Abstract In the current situation, to tackle global climate change and minimize plastic pollution, many countries are enacting policies and strict rules on the usage and alternatives of plastics. As a result, understanding the mechanisms of microplastic release into environmental materials is critical such as water interaction to mitigate the problem as effectively as possible. The primary goal of this study is to integrate plastic waste management and current statements of green treatment technologies for microplastic pollution to help sustain the environment and furthermore to improve sustainability by meeting our communal requirements without causing further harm or depletion of the remaining natural resources and developing alternative production methods to replace those that have been shown to harm human health and to decrease environmental plastic pollution. The sorption capability at the regolith showing a progressive bond with the concentration of microplastics as

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concentration increasing the sorption capacity also increases, and desorption study with water showing the microplastic particles is easily absorbed. The microplastics waste green treatment technologies such as to reduce and conserve plastic usage and its associated nonrenewable energy sources and to safeguard biodiversity, habitats, and biotas to confirm that future generations will be able to fulfill their own needs.

Keywords Microplastics · Green treatments · Water purification · Waste management · Environmental monitoring · Environment's 4Rs

21.1 Introduction

Water is an essential resource and is required for life to exist on Earth. Nowadays, the microplastic contamination problems dominate the use of natural resources. Microplastics—Source is micro but the problem is macro, with no let-out boundaries. Beyond the existing problematic scenario, namely, economic, political, border security, international trade, climate change, water crisis, poverty, racism, etc., the alarming treats on microplastics remain static with no streamlined solutions. Accounting for the review of microplastics, there is an infinite number of reviews and research on this particular component about the sources, origin, properties, identification methods, etc., but there are no solid earmarks on the type or methodology to treat the same or find the solution on eradication or minimization. Policies and regulations made on the same were also not appealing. In the current situation, to tackle global climate change and minimize plastic pollution, many countries are enacting policies and strict rules on the usage and alternatives of plastics. Though there are strict rules in actions, people were not much aware of the future of plastic products and the secondary impact concerning the usage of plastic products. Green treatment technology refers to the process of removing pollutants and undesirable components from household, industrial, and contaminated waters in order to safely restore them to the environment for consumption, agricultural, industrial, and other activities. Green technology is environmentally friendly due to waste reuse (particularly bioremediation and biomass production), its manufacturing process, or its supply chain. When these compounds appear as a mixture, the sorption-desorption behaviors of microplastics on soil clays, which are the most reactive particle components of soils, may change significantly. Even surface water is the most extensively used resource, due to the scarcity of water, the world is more concentrating on the largest freshwater reservoirs that are groundwater, especially in arid and semiarid regions.

21.1.1 Plastic Waste as a Major Problem of the Environment

Plastic is a synthetic organic polymer that is produced from the polymerization of monomers and plays an essential role in the day-to-day activities of human life (Manikanda Bharath et al. 2021a). The main properties of plastic which render its essentiality are durability, less weight, non-corrosion, and lowest price (Ivleva et al. 2017). Since plastics are widely used and it has been turned into a recalcitrant pollutant in a variety of ecosystems such as soil, fresh water, marine water, etc., during the early 1970s, plastics are reported in the marine ecosystem, and according to the Greenpeace report of 2006, there are 267 diverse marine species that have suffered due to embarrassing situations and breakdown of plastic debris (Guern 2017). There are varieties of sizes of plastics ranging from macro- to microparticles (Erni-Cassola et al. 2019). Microplastics are described by the National Oceanic and Atmospheric Administration (NOAA) as abundant plastic particles smaller than mm in size (Arthur et al. 2009), because there are two categories of microplastics: direct and indirect (Cole et al. 2011). Primary microplastics are the products generated at the microscale size and to be used for industrial as well as native products such as hand cleansers, face cleansers, exfoliators, etc. (Zitko and Hanlon 1991; Andrady 2011). Secondary microplastics are produced from the degradation of the macroplastics items in sea and land through a variety of biogeochemical processes such as erosion, abrasion, photooxidation, corrosion, and biodegradation activities which resulted in the fragmentation of microplastics (Zettler et al. 2013).

21.1.2 Occurrence and Pathway of Microplastics in Environments

Microplastics have occurred in various matrices such as soil, sediments, water, food, etc. Generally, they have a larger transformation to various places due to their durability, weightlessness, buoyancy, etc. The transport pathway of microplastics is from the terrestrial region to the marine region (Avio et al. 2017; Horton et al. 2017a, b). As a consequence, microplastics are potentially aggregating in larger concentrations in the water ecosystems and harming marine species (Foley et al. 2018). The exposure route of microplastics into the aquatic creatures is ingestion, absorption, scattering via the circulatory system, entry into tissues as well as cells, and then potentially causing adverse effects on the organisms (Chae and An 2017).

Over the last few decades, the quantity of anthropogenic waste in aquatic and terrestrial ecosystems has increased significantly, with plastic accounting for 60–80% of the total (Derraik 2002). Plastics manufacturing started in the 1950s and has already reached 280 million tons worldwide (Plastics Europe 2017). Every year, approximately 4.8 and 12.7 million metric tons of inadequately handled plastic waste was estimated to enter the waterways from coastal regions (Jambeck and colleagues 2015). Microplastics are most commonly found in cosmetic and medical products containing polypropylene (PP), polyethylene (PE), and polystyrene (PS) particles (Horton et al. 2017a, b). Due to the extreme harmful environmental consequences of microplastics, the sale of cosmetic items using them has been restricted in a number of countries, particularly North America and Europe (Ballent et al. 2016). Physical, chemical, and biological processes that cause plastic waste to fragment produce secondary microplastics (Ryan et al. 2009). Plastic photooxidizes when exposed to ultraviolet (UV) radiation. It becomes unstable and fragments into microbes as a function (Thompson 2006; Harshvardhan and Jha 2013). Though temperature, sun, and also well conditions are excellent for frequent shattering ability to generate microplastics, cold and aerobic conditions in aquatic habitats and soils can cause plastic particles to disintegrate extremely slowly over periods (Zhang 2017). The discharge of sewage discharge containing synthetic fibers or different challenges microplastics from service users or household products to land is another significant primary source of microplastics. The most often observed type is fibers because of the constant abrasion of synthetic textile clothes and furniture, as well as the recycling of washing machine discharge (Napper and Thompson 2016). The synthetic fibers, which are mostly comprised of nylon, acrylic, and polypropylene, are released into the atmosphere alongside primary microplastics (Manikanda Bharath et al. 2021b). According to Browne et al. (2011), activities for industrial effluent and wastewater can release up to 1900 fibers per item into aquatic and terrestrial environments. Textile mills may be a point source of emissions in this context; however, this has not been studied. According to Noren and Naustvoll (2010), areas surrounding plastics manufacturers are projected to be hotspots; concentrations of over 1.2 million plastic particles/m^3 of seawater have been observed in a Swedish port region next to a polyethylene (PE) manufacturing company. Owing to the vast volume of macroplastic waste, another sources of microplastics particles are known as significant contributors to microplastic contamination. Anthropogenic activities such as littering generate secondary microplastics, which are emitted during landfill sites and waste management's zones. Surface runoff from agricultural fields and urban areas is another important source of microplastic accumulation in surface streams. Wind dispersal, soil erosion, and surface runoff can all introduce large plastic products and their degraded components into aquatic environments (Kole et al. 2017; Horton et al. 2017a, b).

There is also evidence suggesting tyres and road signs may contribute to plastic contamination, with water pollution acting as a significant transport channel for tyre and road wear particles (TRWP) to surface waterways (Dris et al. 2017; Unice et al. 2019). In addition, current study has found that a considerable quantity of fibers has been carried by air debris, particularly in highly populated areas (Cai et al. 2017). Synthetic materials from clothes and households, artificial turf, dumps, and waste incineration all contribute to microplastics in the atmosphere (Magnusson et al. 2016; Dris et al. 2017). Small particles in the airborne can be transported by the wind and deposited on ground, or they can be released into the atmosphere and deposited in the water bodies. As a consequence, physical processes driven by climatic factors, including such wind, waves, streamflow, and floods, alter the spatial patterns of microplastics within environmental media (Zhang 2017). Plastic particles are becoming a significant issue in aquatic ecosystems due to the ecotoxicological dangers they represent. Microplastics can exhaust energy reserves, bioaccumulate, and biomagnified across the food chain when consumed by a number of species. A look-at-green treatments that have been studied, validated, and deployed as clean water treatment alternatives to fill the void produced by inadequate conventional technology. Many urban areas in India were more polluted with plastic. Family units, as a subset of the general population, consume more plastic items and generate an enormous amount of waste as a result. Today, the rise in microplastic pollution, as well as increased government regulation, has cast doubt on some traditional green treatment technologies. This study aims to highlight the current status of microplastics research, to provide a wide overview of the complexity of microplastics, and to provide a broad view of the diversity of microplastics while categorizing exploration gaps to plan future research goals. The objective of this research is to use recycling methods and green treatment technologies to create jobs for unemployed youth and to improve our environment, further to promote the Environmental R's (reduce, recycle, and reuse) based on mitigating the use of plastics, to make our countries clean and serene to prevention of epidemic, and finally to create awareness of plastic pollution carrying out educational campaigns and to encouraged to reduce the use of plastics.

21.2 Microplastics, Wastewater Treatment Plants and Future Developments

The removal of microplastic contaminants and other toxic components from household, agricultural, and polluted waters in order to properly restore them to the environment for drinking, agriculture, industrial, and other purposes is referred to as green treatment methods. The first step of the process is to separate the micro- and macroplastics from the liquid water. This is accomplished by using density, gravity, and floating methods to remove plastic waste from water. Through using floating approach, other solid components that are less dense than water may be removed from the surface of water, and particles of concern can be clearly removed. The stream effluent is next filtered to eradicate any nanoparticle suspensions, toxic particulates, and contaminants. After removing toxicity of plastics and other suspended pollutants from wastewater, it was filtered and released into the environment.

Advanced green technologies, in particular to being environmentally friendly, provide numerous benefits over standard wastewater treatment methods. Variations or intermittent loading, for example, are less likely to cause hydraulic shock in biofilters. It must be included in wastewater treatment. Furthermore, their operating expenses are probably lower than those of alternative techniques, such as wastewater treatment and the green treatment technologies for microplastic pollution as shown in Fig. 21.1. Bioremediation techniques are also less expensive since they do not need excavation, combustion, or clean strategies like "pump and treat" used in

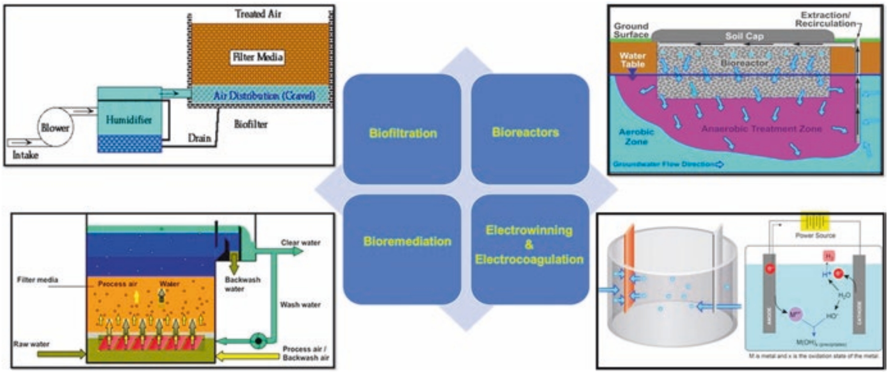

Fig. 21.1 Advance green treatment technologies for microplastic pollution

water treatment. More significantly, instead of storing or spreading pollutants, modern green technologies typically break to the molecular level. Other techniques, such as biosulfide extraction and electrowinning, can produce stable waste material, but electrocoagulation produces clear, colorless, odorless water that can be discharged. A variety of effective treatment techniques and materials based on plastic pollution in the environment were explored in this article.

21.2.1 Current Scenario on Treatment of Microplastics

Microplastics are supplementary by-products/end products of the major global components, namely, cookery, pet bottles, microwave containers, medical supplements, personal care products, household articles, etc. (Gregory 2009), enriching their presence in multivarious facets from rich to poor with no partiality. Though the production technology reached peaks with high-end variations, the solution for treatment and reduction remains idle (Graham and Thompson 2009). This minute 5 mm particle stamped a strong foundation globally and remains challenging with no solid solution. Horton et al. Horton et al. 2017a, b suggested the marked terrestrial environments and fresh waters as key contributors with the marine environment as a major sink. The energy level in the aquatic environment plays a major role in microplastics deposition (Manikanda Bharath et al. 2021a). Accordingly, low energy in the aquatic environment induces a high sedimentation rate and high accumulation of plastics, whereas high-energy levels act vice versa (Corcoran et al. 2015). Accordingly, Padervand et al. 2020 marked the limited version of microplastics removal from polluted systems. They warn about the severity of plastics and microplastics and their role in collapsing the biological environment of the aquatic systems (Cole et al. 2011). So it becomes mandatory to check with the possible treatment strategies.

21.2.2 Potential Impact of Microplastics in Nature

Microplastics occurrence in soil environments is gaining importance nowadays since from the soil, it transports to the plant tissues, and thus it enters into the human food chain (Manikanda Bharath et al. 2021b). On the other side, there are great chances for the contamination of food items by microplastics. It also acts as a carrier for other pollutants which potentially risk human and other organisms (Amiard-Triquet et al. 1993). Polyethylene, hazardous metals, phthalate, bisphenol A (BPA), polychlorinated biphenyls (PCB), and polycyclic aromatic hydrocarbons (PAHs) may all be attributed to the adsorption of microplastics and transferred readily from one site to another (Hahladakis et al. 2018). The poisonous chemicals linked with microplastics may cause biomagnifications in environments (Barboza et al. 2018). Hence, it increases the potential toxic risk of those chemicals to the consuming living organisms either through microplastics or the associated contaminants (Kelly et al. 2007; Hermabessiere et al. 2017). Hence, it is forecasted that the microplastics will increase the risk of emerging new contamination or infection through introducing new pathogens and vectors by acting as a carrier for those infections or contaminants (Keswani et al. 2016).

Another troubling aspect of microplastics is the impact on human health caused by the consumption of food and seafoods contaminated with microplastics. There are a lot of studies that revealed that food and seafoods contain considerable ranges of microplastics, but the adverse impact on human health remains unexplored. Many people believe that microplastics will cause cancer in marine animals and humans and that due to their size, they will enter cell membranes, the placenta, and secondary tissues such as the brain, liver, and muscle, among other places (Barboza et al. 2018).

Hence, a detailed assessment is essential to explore the impacts of micro- and nano-plastics on humans. Its study should be based on the average daily dietary exposure of microplastics through a variety of foods taken up by human beings. Furthermore, the parameters related to microplastics like particle size, polymer composition, surface area, density, persistence, adsorbed contaminants, and additive contents, etc. should be considered during the risk assessment (Hale 2018).

21.3 Green Strategies to Control Microplastics

The microplastics pollution in an aquatic and terrestrial ecosystem is considered more dangerous when compared to other pollutants due to their severe effects on marine organisms through ingestion and bioaccumulation or biomagnifications (Cole et al. 2011). The persistence and degradability of microplastics are determined by their nature and chemical structure, as are the methods of removal (Verschoor 2015). The primary strategies for controlling microplastic pollution are prevention, mitigation, removal, and behavioral changes, among others (Graham and Thompson 2009). Some traditional wastewater treatment systems are being called into service today as a result of increased environmental awareness and government regulation. Green treatment solutions are being studied, validated, and deployed as clean alternatives for wastewater treatment to fill in the gap left by inadequate conventional networks.

A number of steps are included in every wastewater treatment procedure. The first stage is to separate solids from liquid water. Gravity is utilized to achieve this since sediments are heavier than fresh water. Other solid materials that are less dense than liquid water, such as oils and woods, might be removed from the surface water. Beyond that, any fine solid colloidal suspensions, chemical particles, and contaminants in the stream effluent are cleaned (Gregory 2009). After filtering, the water is oxidized, which lowers or removes the toxicity of any residual contaminants and disinfects the effluent before it is discharged into the environment.

Water recycling, in addition to some water conservation techniques, is a readily available option for effectively meeting industrial, domestic, and environmental demands on a daily basis. The use of treated water relieves pressure on the freshwater supply. Wastewater treatment is a sanitation process that eliminates waterborne diseases. It provides public health and safety protection. Recycled water is said to be pathogen-free, so it can be used publicly, even for bodily contact. Even though it will not be consumed by humans, it must be chemical-free to avoid causing other harm to the environment.

21.3.1 Prevention of Microplastic Pollution

To address the problem caused by various types of plastics that are constantly entering the marine environment, a diverse set of prevention measures is required (Ryan et al. 2009). The most important steps toward reducing plastic waste and moving toward circularity are modifying plastic products for circularity, minimizing preproduction plastic waste, trying to extend capacity available, preventing certain types of single-use plastic products, and promoting the reprocessed plastics industry. Other method to reduce plastic pollution at the source includes improving laws and policies related to plastic pollution control, relevant standards for plastic pollution control, and plastic waste management policies, among others (Zhong and Li 2020).

21.3.2 Mitigation of Microplastic Pollution

Microplastic mitigation is another strategy to reduce its entry into the environment via different pathways. The disposal and dumping of plastic debris should be regularized and maintained scientifically. The treatment technologies for wastewater should be standardized further, and waste discharged outside should be free from microplastics.

21.3.3 Removal of Microplastics

The microplastics present in the soil and water ecosystem can be removed by employing a variety of methods. Each method has advantages and disadvantages, and because microplastics are being considered as emerging pollutants, the use of methods should be speculated more clearly in the future. Waste-free from harmful substances can be allowed to discharge in the marine environment, and it may be prohibited in ecologically sensitive areas (Bhuyan et al. 2021).

21.3.4 Physical Methods for the Removal of Microplastics

Microplastic contamination in the marine environment is mostly caused by sediments. Cleanup actions should be conducted to remove microplastics from the seafloor, and marine debris monitoring programs may be implemented to reduce microplastic contamination. Moreover, the main technologies used to remove the microplastics during water/wastewater treatments are filtration and membrane technologies which are discussed below.

21.3.4.1 Filtration Methods

According to studies on wastewater treatment methodology, the treatment includes a variety of processes ranging from primary, secondary, and tertiary or advanced treatment techniques. Denitrification, ultrafiltration, ozonation, and UV irradiation are all part of the advanced treatment process. The main aim of filtration or ultrafiltration is to remove the microplastics from wastewater or sewage water. Though the conventional sewage treatments are not designed with the aim of removing microplastics, it holds good for the removal at some extent. During the primary processes, the removal efficiency would be 71.67%, whereas at the end of advanced steps, it reached up to 99.9% removal of microplastics (Talvitie et al. 2017). Various types of filters such as disc filter and membrane bioreactors have been employed in the filtration process based on the size and quantity of microplastics present in water. Furthermore, with the filtration combined with the other processes such as biological and sedimentation, the efficiency of microplastic removal was good enough (Lares et al. 2018). Several researches reported microplastics removal effectiveness by various technical advancements, such as Li et al. (2018) for dynamic membranes and Ersahin et al. (2017) and Horton and Dixon (2018) for dynamic membrane and reduced turbidity. Ward (2015) describes the usage of polymer coatings as an expanded mesh screen (Talvitie et al. 2017; Gurung et al. 2016). According to the Membrane bioreactor (1994) combine parts of a system with biological systems of the membrane bioreactor, disk filtration, fast sand filtration, and dissolved air floating are all used in this system. Based on the literature study strongly recommend membrane bioreactor's porous membrane along with biological efficacy removes 99.9% of microplastics from the aquatic environment.

21.3.4.2 Membrane Technology

A dynamic membrane developed from diatomite supporting mesh of 90 μm has an efficient removal of microplastics within a time of 20 min (Ersahin et al. 2017). This was observed due to the reduction in turbidity and influent water from 195 NTU for it has been reduced to less than 1 for the wastewater (Horton and Dixon 2018). Another efficient polymer-coated elongated mesh screen was defined by Ward (2015) for efficient removal of microplastics which has many advantages such as good durability, easily fabricated, utilized without electricity, etc. (Gurung et al. 2016). Membrane bioreactors, on the other hand, have been found to be more efficient than dynamic membranes in the removal of pollutants as polymeric debris and microplastics. Generally, membrane technologies have greater efficiency in the subtraction of microplastics from fresh water, and their efficiency depends on many factors such as the quantity of microplastics present in a matrix, membrane size, and durability, changes in influent and effluent water flow, etc. There may be an enhanced impact on the removal of microplastics when the porous membranes combined with biological processes (Padervand et al. 2020).

21.3.5 Biological Methods for the Subtraction of Microplastics

There are many biological techniques involved in the eradication of microplastics from the freshwater environment through potentially greenways. Some such methods are biosorption, microbial degradation of microplastics, microbial ingestion, plant uptake, etc.

Microplastics with lower half-lives (days) in seawater >60, fresh water >180, fresh or estuarine sediment >120 are considered degradable without posing a risk to the surrounding environments (Verschoor 2015). The certain algal groups have the capacity of adhering microplastics, and this was studied by Sundbæk et al. 2018 giving a positive hope by Fucus vesiculosus, seaweed, and an edible marine microalga with the behavior of adhering fluorescent microplastics onto their surface. Interestingly, the cut regions of the seaweeds have a gelatinous polysaccharide (alginate) which enhances the adherence of the polystyrene particles to their surface (Martins et al. 2013). The microplastic particles also a major role in getting adsorbed on the microalgae. Research on the Pseudokirchneriella subcapitata, unicellular green algae, by Nolte et al. (2017) highlighted greater adsorption by positively charged polystyrene compared to negatively charged microplastics.

Dawson et al. (2018) discovered Antarctic krill (Euphausia superba), a planktonic crustacean, fragmenting and resizing microplastics, with evidence of biologically mediated microplastic to nano-plastic transformation. Paço et al. (2017) studied the fungus Zalerion maritimum's (a naturally occurring fungus in marine ecosystems) capacity to decompose microplastics. Auta et al. (2017) examined the breakdown of different microplastic composition by Bacillus cereus and Bacillus gottheilii, namely, polyethylene, polystyrene, polyethylene terephthalate, and polypropylene. According to Arossa et al. 2019, the Red Sea giant clam and Tridacna maxima removed 66.03 percent of microplastics from wastewater, indicating a high potential for microplastic sorption.

21.3.5.1 Biosorption of Microplastics

Some studies revealed the adherence capacity of microplastics on the surface of microorganisms present in the marine or terrestrial ecosystem. The sorption of microplastics on marine microorganisms mainly depends on the surface charges of the microplastic particles. Since the chemical structure of microbes is with anionic polysaccharides, positively charged microplastics may easily adsorb on the surfaces of microorganisms. Sundbæk et al. (2018) described the pattern of fluorescent plastic debris adhesion on the top of the seaweed Fucus vesicular. Nolte et al. (2017) studied the sorption of polystyrene particles ranging in size from 20 to 500 nm on the surfaces of Pseudokirchneriella subcapitata algal species and concluded that positive-charge polystyrene microplastics are more effectively adsorbed.

21.3.5.2 Microbial Degradation of Microplastics

Though the biodegradation of plastics is still unclear in many cases, number of researchers provided evidence for the fragmentation of macroplastics into micro- or nano-plastics by a variety of microorganisms. The fragmentation of polyethylene microplastics and the results proved that the size altered due to the ingestion by Antarctic krill *(*Euphausia superba) (Dawson et al. 2018). Cocca et al. found two indigenous marine communities such as the Agios consortium and Souda consortium for the removal of high-density polyethylene from marine water. Both microscopic and FTIR images confirmed the fragmentation and degradation of microplastics in the above studies. Paço et al. (2017) discovered that the fungus Zalerion maritimum may biodegrade polyethylene microplastics in a bioreactor.

21.3.5.3 Microbial Ingestion

The ingestion of microplastics by the microorganisms present in the marine environment is also considered as one of the removal strategies. Cole (2013) studied the microplastic ingestion had a negative impact on zooplankton function and health but that it has a high capacity for the removal of 1.7–30.6 m polystyrene microplastics through ingestion. There is also evidence that scleractinian corals consume polypropylene microplastics. Red Sea giant clam such as *Tridacna maxima* have also the capability of removing 53–500 μm polyethylene microplastics (Arossa et al. 2019). The *Tridacna maxima* have also the capability of removing 53–500 μm polyethylene microplastics (Arossa et al. 2019). Based on previous research, it is possible to conclude that because microplastics ingested by marine organisms affect physiological functions, they can be used for the removal of microplastics present in low concentrations (Martí et al. 2017).

21.3.5.4 Plant Uptake of Microplastics

Any remediation process aims to remove the pollutant from the environment without affecting its holistic nature. Phytoremediation is an eco-friendly technique that is predominantly employed (Qi et al. 2018). The usage of plants, bio amendments, soil biota individually or together the successful remediation of polluted in soil. Similarly, the recent findings of various researchers (Li et al. 2019) confirmed the accumulation of microplastics by plants. However, various phytoremediation techniques such as phytoextraction, phytofilteration, and phytostabilization can be employed based on the type and microplastic contamination in particular site. The root zone of the plants is the first contact point of microplastics present in the contaminated soil. The microbeads of polystyrene (0.2 μm) size were found in the root cap to lettuce plants (Li et al. 2019), and hence, it was proved that the rhizosphere is the first contact region in any phytoremediation techniques. The selection of plants for phytoremediation should be based on a number of criteria such as tolerance for microplastics, accumulation capacity, biomass production, bioaccumulation factor, profuse root system, etc. Apart from various advantages, the major disadvantage is the risk of entry of microplastics into the human food chain. Hence, care should be given for the plant selection based on mobilization or immobilization of microplastics in the soil environment (Ebere et al. 2019).

21.3.5.5 Behavior Changes

Social awareness regarding the demerits and environmental impacts of microplastics should be imparted to the public through education campaigns and noneducational meetings. The behavioral changes of people toward the handling, usage, and disposal of plastic materials should be achieved through economic/incentive tools. Young minds should be nurtured about the proper handling and management of plastic waste through school and college curriculum. It will lead to positive changes in the behavior of the common people in a scientific way.

21.3.6 Application Fields of Green Treatment Technologies for Microplastics Pollution

The development of alternative fuels is one important application field for green treatment technologies of microplastics waste. Wind turbines, solar cells, and bioreactors are examples of clean, sustainable, and productive new energy sources that are being created and applied. Unlike traditional fossil fuels, these energy sources create electricity without damaging the environment. Monitoring is the third area of operation for green treatments, which involves weather forecasting, enabling remote discharge monitoring utilizing spatial cognition, etc. These methods can help minimize energy waste and greenhouse gas emissions when combined with building monitoring. Municipalities, industries, and environmental authorities can use remote, online monitoring systems integrated with the Internet of Things (IoT) technology to measure pollutants and discharges in real time while making process or other modifications to maintain compliance.

The following processes can be used in water reclamation:

Solutions for membrane bioreactors: This method combines biological, secondary, and tertiary wastewater treatment in a single step. It seeks to reduce the carbon footprint associated with sludge sewage treatment. It employs a high level of organic, microorganism, and nutrient removal.

Ultrafiltration solutions: Ultrafiltration solutions are commonly used when water is treated for drinking purposes. It employs membrane filtration, in which a force similar to pressure separates particles from liquid or gas mixtures. Water viruses, bacteria, protozoa, and other pathogens are effectively removed because it is intended for human consumption.

System of reverse osmosis: This is a supporting treatment that is used after the water has been pretreated to remove unwanted particles. For safer use, the water is desalinated using a reverse osmosis system, which creates an excellent barrier against pathogens. It employs a semipermeable membrane with small pores to ensure that only pure water passes through.

Electrodialysis reversal: This is a desalination process in which electricity is used against electrodes to separate salt and other particles. It is self-cleaning and thus suitable for turbid wastewater. In water-stressed areas, electrodialysis reversal provides one of the highest recoveries.

Thermal evaporation and crystallization systems: Evaporation and crystallization are two methods of wastewater treatment that are commonly used in brine, streams, and seawater. It collaborates with other processes like reverse osmosis to create Zero Liquid Effluent Discharge systems. This is a low-cost disposal method that is popular among businesses that have a recycling system.

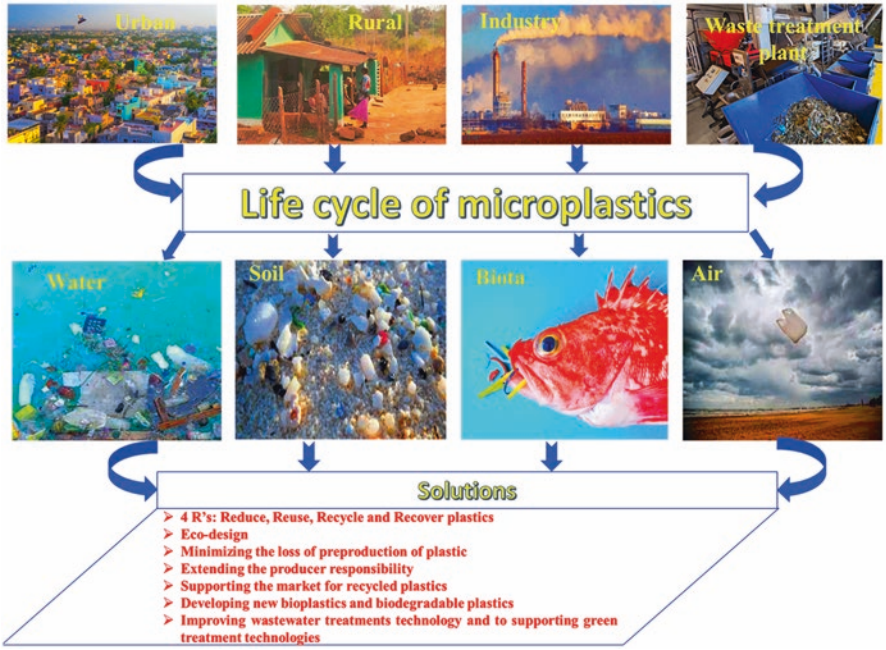

Fig. 21.2 The life cycle of microplastic impacts on environments and solutions of mitigating plastic pollution

21.3.7 Cleaning and Remediation of the Environment

The second major use of advanced green technology is environmental sanitation and remediation. The life cycle of microplastic impacts on environments and solutions were shown in Fig. 21.2. This includes water and air purification, waste management, environmental cleanup, and wastewater treatment. Cleaning is done using a combination of green physical and chemical methods.

21.3.8 Environmental 4Rs Concept

Accounting the 4Rs (reduce, reuse, recycle, and recover) mechanism for waste management plays a major role in awareness campaigns, meetings, pamphlets, topics of discussion for webinars, theories, and other IEC components (Solis and Silveira 2020). But practically, not much improvement is experienced nationally when compared with the international outlook. The 4Rs concept should be impounded from the childhood itself with practical applicability. If not, the same will remain only as a concept rather than reducing the killer impact marking their footprints on mother nature. Though there are policies and laws for the restriction of plastic usage, it is not creating much improvement because of the liberalism maintained in the practical applicability of policy violators and the lack of affordable alternatives.

21.4 Conclusion

Recycling methods were used in green treatment solutions like minimising and conserving plastic usage and its associated non-renewable energy sources, as well as protection of biodiversity, habitats, and ecosystems to ensure that future generations can satisfy their own goals. Sorption capacity of the regolith showing a positive relationship with the concentration of microplastics as concentration increasing the sorption capacity also increases and desorption study with water showing the microplastic particles is easily absorbed with a 100% absorption. The management of any plastic waste should rely on the 4Rs strategy. Effective waste management rules and policies may pave the way for scientific treatment and disposal of plastic wastes, and in turn, it reduces the emerging microplastic pollutant from various sources. There are so many ways and methods to treat microplastic-contaminated soil and water, but all are in very primitive stages. Hence, a detailed experiment and analyses should be executed to confirm the outcome of each method for its efficiency and suitability. Though the chemical methods are easily adaptable and efficient, there is the possibility of further contamination by the chemicals employed in those methods. Hence, we may opt for suitable physical or biological methods for the green treatment of microplastics. This review demonstrates the possibilities and potential of advanced green technology in wastewater treatment and environment monitoring, which will become increasingly important as the world shifts to clean renewable energy and waste resource recovery. Water scarcity is becoming more common as demand for it grows. The need to develop strategies for long-term sustainability is becoming more pressing. Experts are now looking into wastewater processing as technology advances. Wastewater treatment is a process that removes unfavorable compounds and contaminants from water. Polluted waters from home and industrial sectors will be treated with green technology and safely returned to the environment for irrigation, industrial usage, and drinking. Effective urban water management is essential to attain these goals. It will treat contaminated waters from domestic and industrial areas with green technologies before returning them to the atmosphere for irrigation, industrial use, and drinking. To achieve these objectives, effective urban water management is required. We'll find out how green technology can help with wastewater treatment. Though there are strict rules in actions, people were not much aware of the future of plastic products and the secondary impact concerning the usage of plastic products.

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Chapter 22
Chemical Technologies to Degrade Microplastic Pollution

Asifa Nasrullah, Hadiqa Basharat, Muhammad Zaffar Hashmi, and Muhammad Ashfaq

Abstract Microplastics (MPs) are semisynthetic plastics having diameter less than 5 mm and considered as one of the most abundant pollutant in environment. MPs become part of the environment by the breakdown of larger plastics or through the release of plastic feedstock like pellets, nurdles, and microbeads from industries into the environment. MPs have also been identified in every marine habitat (beaches, surface water, deep seafloor) around the world. There are various chemical techniques used for the degradation of MPs which include advance oxidation processes such as photocatalysis, photodegradation, chlorination, and coagulation, agglomeration, flocculation, and chemical weathering. Photocatalysis is most extensively used because undesired products are not formed. Photodegradation involves the exposure of material to lighten ambient conditions resulting in the formation of free radicals of microplastics. Degradation of microplastics depends on intensity of light as well as nature of environment in which photodegradation occurs. MPs can be degraded using chlorination. Chlorination breaks the bond and introduces the new bonds between chlorine and hydrogen. Coagulation/flocculation/agglomeration processes entail the formation of large-sized particles of MPs by using salts of Fe and Al and other coagulants. Electrocoagulation is robust, is cost-effective, is energy efficient, is environmentally friendly, produces minimum sludge, and is resilient to automation. Chemical weathering processes cause fragmentation, alter the chemical composition along with decrease in molecular weight, and destroy thermal and mechanical properties of microplastics.

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22.1 Introduction

In the "plastic age" where we use plastic in every field of life (Rillig 2012), due to the excessive use of plastic all over the world, its annual production has reached to 335 million tons in 2016 (Peixoto et al. 2019). Plastic is extensively used due to its low weight, easy to manufacture, temperature and chemical resistant, and low cost (Sharma and Chatterjee 2017). Microplastics (MPs) are the emerging pollutants, produced by the fragmentation of larger plastic and are used in cosmetics, textile, and pharmaceutical industries. Effluents of these industries mix with the water bodies and cause water pollution. The first researchers detected plastic pellets on the surface of "North Atlantic Ocean" in 1972 and stated that "at present, the only known biological effect of these particles is that they act as surface for the growth of hydroids diatom and probably bacteria" (Carpenter and Smith 1972). MPs are basically of two types: (1) primary MPs formed by the release of plastic feedstock, i.e., pellets, nurdles, and microbeads and (2) secondary MPs produced by the breakdown of larger plastic through different processes (Kershaw and Rochman 2015). Recent studies have shown four basic sources of MP pollution: (1) medicines (microplastics from pharmaceutical industries and medicine used for drug delivery enter into water through waste), (2) cleansing products (enter into the water bodies through sewage and stormwater) (Zitko and Hanlon 1991), (3) larger plastic litter (comes into water bodies through shipping and fishing activities), and (4) textile industry (Browne et al. 2011). MPs are identified in soil, beach sediments, marine animal bodies, human bodies, and air employing different analytical techniques like pyrolysis GC/MS, Raman spectroscopy, and IR spectroscopy (Bergmann et al. 2015). MPs are found in the gills, gut, and liver of fishes, turtles, zooplankton, and marine mammals and may cause oxidative stress (Peixoto et al. 2019) along with lethal effect on human through utilization of contaminated seafood. In addition, MPs have also found in sugar, beer, salt, honey, and mineral water (De-la-Torre 2020). Synthetic MPs are polluting the air in different ways which in turn has very harmful impact on the health of living organisms (Verla et al. 2017). The airborne MP toxic substances cause the DNA destruction, oxidative stress, cancer, and defect of immune cell (Enyoh et al. 2019). MPs also reduce the fertility of soil (Stubenrauch and Ekardt 2020).

This chapter includes type, sources, and impact of MPs on marine environment, soil, air, and food. Our main focus will be on the chemical technologies used for the degradation or removal of MPs.

22.2 Microplastics (MPs)

Microplastics (MPs) are considered as the heterogeneous mixture of many particles of different sizes ranging from microns to several millimeters in diameter. Two terminologies are used for MPs on the basis of its sizes: larger MPs having size of 1–5 mm and smaller microplastics with size ranging from 1 to 1000 μm (Hanvey et al. 2017). Generally, MPs are called as semisynthetic plastics which have diameter less than 5 mm (Free et al. 2014) and appear in various shapes from spherical to elongate along with different colors; for example, red and blue fibers are most common (Bergmann et al. 2015). Synthetic plastic was produced first time in 1907 by reacting phenol with formaldehyde (Ng et al. 2018) and is a very versatile material (can be molded in different shapes), inexpensive, corrosion resistance, having high electrical and thermal insulation values, very strong, and having lightweight (do Sul and Costa 2014). Most commonly used synthetic fibers are polyethylene, polypropylene, polyvinyl chloride, nylon, and polystyrene (Andrady and Neal 2009). MPs are widely used in textiles, medicines, pharmaceuticals, cosmetic industries, drug delivery, and dentist tooth polish (Sharma and Chatterjee 2017). In cosmetics, it is commonly used in hand cleansers and scrubbers (Thompson et al. 2004). The main ingredients of the compact face powder are polypropylene, polystyrene granules, and polyethylene (Beach 1972).

22.3 Types of Microplastic

Two types of microplastics are present in environment, i.e., primary microplastics and secondary microplastics.

22.3.1 Primary Microplastics

Primary microplastics are also called microscopy fragments of plastic and are formed by the release of plastic feedstock, i.e., pellets, microbeads, and nurdles. The plastic feedstock is the by-products of the plastic-based materials (Kershaw and Rochman 2015), whereas microbeads are used in the personal care items (Hanvey et al. 2017). In the manufacture of plastic items such as plastic bags, plastic pellets are used as the precursors. Plastic pellets consist of polypropylene, polyethylene, polyolefin, and polystyrene. They all are lipophilic in nature and can absorb the harmful chemicals from the surface of the marine water. Different aromatic hydrocarbons have been detected on the surface of pellets, i.e., polychlorinated biphenyl (PCBs), dichlorodiphenyltrichloroethane (DDT), and polycyclic aromatic hydrocarbons (PAHs) (Van Cauwenberghe et al. 2015).

22.3.2 Secondary Microplastics

Secondary microplastics are the fragments of bigger plastic items which are formed by the breakdown of larger plastics. Breakdown of larger particles into tiny fragments may be caused by weathering. Different processes are used to degrade the larger plastics into the smaller fragments. A very important process of fragmentation is photodegradation by ultraviolet radiation, and as a result of which, the chemical bond between the matrix of polymer is broken. Other processes include biodegradation (action of living organisms, i.e., microbes), thermal degradation (under high temperature), hydrolysis (reaction with water), etc. (Sharma and Chatterjee 2017).

22.4 Sources of Microplastic Pollution

There are various sources of MP pollution. These may be land-based or water-based sources.

22.4.1 Textile Industry

Textile washing process is a major contributor of MP pollution in the aquatic environment. Estimated 5.6 million tons of synthetic fibers were produced all over the world in 2016 which may enter the water bodies. Polyester microplastics are found in the aquatic environment (Deng et al. 2020). It is estimated that the synthetic fibers may contribute up to 90% microplastics in aquatic environment (Barrows et al. 2018). The processes, i.e., washing, transportation, and packaging of the textile material, are responsible for the release of microplastics. All synthetic fibers such as nylon, polyester, and acrylic, shed the microfiber on washing, but the polyester fleece shed the highest amount of the microfibers on average 7360 fibers/m^2/L[1] in one wash (Barrows et al. 2018). Yapingsai and coworkers (2020) identified the different microplastic pollutant in water by washing various times the synthetic fibers of 12 different textiles. The 12 textiles showed the variability in microplastic fiber release. The results indicated that shedding ranges from the 210 to 72,000 microplastic fibers per gram textile per wash (Cai et al. 2020). Highly processed textile fibers release the more microplastics fiber than the less processed textile material on washing. Household and industrial washing of synthetic textiles shed the microplastics due to the mechanical and chemical stresses that are faced by material during the washing in addition to the nature of detergent which may also affect the microplastic fibers release from the textile fiber (De Falco et al. 2018).

22.4.2 Cosmetics and Personal Care Products

Cosmetics and personal care items are also a source of MP pollution because the personal care products contain the plastic microbeads as exfoliate agent. Plastic microbeads have been added into the toothpastes, scrubs, shampoo, and facial soaps, shower gel, hand sanitizer, sunbath cream, and shaving cream. MPs derived from the personal care products are less than 350 μm in diameter (Sun et al. 2020; Lei et al. 2017).

22.4.3 Wastewater Treatment Plants

"Wastewater treatment plants" are the major source of MP in aquatic environment. Although the WWTPs remove the considerable amount of microplastics, the effluents entering water bodies contain the microplastics. In a research work reported by the Talvitie et al., effluents from a WWTP contain 4.9 ± 1.4 fibers while 8.6 ± 2.5 synthetic particles. The discharged sediments contained 1.7 ± 1.0 fibers, 7.2 ± 4.9 synthetic fibers, and 1220 ± 160 black carbons (Talvitie et al. 2015). In a study by (Gündoğdu et al. 2018), 6 days of treatment results showed that influent wastewater carried up to 6.5 million plastic particles in a day while the effluent carried 1.5 million microplastics. The removal efficiency of WWTPs was in between 73 and 79%.

According to Kay et al. (2018), the amount of microplastics was more in downstreams than upstreams of WWTPs. Although modern treatments can remove the microplastics efficiently, the effluents from the secondary treatment contain the considerable amount of microplastics. In a study by Murphy (Murphy et al. 2016), secondary wastewater treatment work effluents consist of 65 million microplastics. Sludge from the wastewater treatment plant also carries the considerable amount of microplastics (Rolsky et al. 2020). White sludge-based microplastics were abundant in eastern China with $22.7 \pm 12.1 \times 10^3$ particles of microplastics per kg of dry sludge (Li et al. 2018). In another study conducted in Australia, it was found that effluent may contribute up to 22.1×10^6 to 133×10^6 microplastics/day, while sludge may add 400×10^6 to 9100×10^6 MP particles/day into the environment (Ziajahromi et al. 2020).

22.4.4 Landfills

About 21–42% plastic is deposited in the landfills all over the world. Under the influence of different factors, plastic is converted into the microplastics and carried to debris by leachates. Landfilling is a slow process, and the system goes through complex and dynamic changes such as temperature change, variation in pH, and amount of oxygen (Kjeldsen et al. 2002). Recently, it was investigated that landfills are the source of microplastic pollution in oceans (He et al. 2019). The results showed that leachate consists of seven types of plastics.77.48% microplastics were in between 100 and 1000 μm^2. In the landfilling process, anaerobic process occurred which was responsible for the breakdown of plastic into microplastics (Mahon et al. 2017). Size and distribution of microplastics depend upon age of landfills. In an aerated bioreactor landfills, ventilation can be the source of microplastics into the environment. Seepage from the landfills can be ways for the introduction of microplastics into the aquatic ecosystem (Foose et al. 2001). Soil application can be responsible for the microplastics pollution in the terrestrial environment via runoff from the constituents of landfills (Zubris and Richards 2005).

22.4.5 Vehicle Tire Wear

Tire wear particles, which are formed by the rubbing of tires with road surface, are considered as primary source of MPs (Sundt et al. 2014). There are various ways by which the tire wear particles can be transported, i.e., may settle on the road, washing with water, dispersed by wind allow another route (Gnecco et al. 2005). In a study by Verschoor et al. (2016), about 10% of plastics become part of surface water, 40% dumped into soil, 5% enters in air, while remaining 45% come on contact with road surfaces. In the recent study of World Ocean (Evangeliou et al. 2020), tire wear particles and brake wear particles had the high transport efficiency and were abundant in arctic region. It is estimated that per capita production of microplastics derived from the tires is 0.23–1.9 kg/year in India and Japan, respectively, while in America, it is 4.9 kg/year (Kole et al. 2017).

22.4.6 Paints

The composition of paints includes coloring matter, filer, additives, and solvents. The paints are used for the coating of ships, aircrafts, automobiles, etc. There are different kinds of paints depending upon their film-forming material such as natural, alcoholic, amino, alkyl, nitro, epoxy, acrylic, polyurethane, organosilicon and silicone-based paints. Recent research has revealed that paints are source of MP pollution due to erosion, aging, and scrapping from the surface of different material. In 2015, the production of industrial and architectural paint was 115.624 and 5.624 million tons, respectively. It is estimated that the annual paint production of China reached up to 14% of global production from 2006 to 2016. Due to excessive production and usage, the shedding of paints in the environment ultimately takes MP pollution in the environment (An et al. 2020) of which architectural coating is its primary source (Wang et al. 2019).

22.4.7 Municipal Debris

Municipal debris include plastic bags, plastic bottles, plastic packages, and plastic tableware. All these things are made of plastic along with additives. Due to low cost, high competency, and easy storage, plastic is excessively used by mankind. Plastic bottles and bags packages are mostly made up of polyethylene and polypropylene (An et al. 2020). It is estimated that 25% of packaging materials include plastic bags, and 37.5% of plastic production is used in the formation of plastic bags (Kershaw 2016).

22.4.8 Fishing Waste

Fishing waste is composed of fishing nets, lines, cables, tanks, rods, boxes, and buoys. It is estimated that 0.13 to 135,000 tons of fishing gadgets is thrown out annually (Merrell Jr 1980). The polystyrene and Styrofoam are the major component of aquaculture equipment. The fishing boats also incorporate the fishing nets and fishing ropes into the water which undergo the degradation, converted into small fibers, and increase the amount of plastic in water bodies that is why the fishing waste is a major contributor of MP pollution in the environment (An et al. 2020).

22.5 Impacts of Microplastic Pollution

22.5.1 Marine Environment

Nowadays, MP is considered as one of the most abundant pollutant in marine environment. MPs have been detected in seawater and sediments of Pacific Ocean, Atlantic Ocean, European seas, Mediterranean Sea, Indian Ocean, Southern Baltic Sea, Caribbean Sea, and marginal seas at different contamination level as shown in Table 22.1 and have been identified in every marine habitat (beaches, surface water, and deep seafloor) around the world which is an alarming situation for the aquatic life. It affects the fishes, invertebrates, turtles, and mammals. Microplastics greatly affect the organisms at cellular level (Bergmann et al. 2015). Excessive exposure of MPs may cause the weight loss, increase the phagocytic activity, and decrease the feeding activity. Ingestion of MPs may cause the blockage of digestive system, inhibition in secretion of gastric enzyme, and infertility (Sharma and Chatterjee 2017). Microplastics accumulated in the liver, gills, and gut may cause the inflammation and oxidative stress (Peixoto et al. 2019). They are more dangerous than the macroplastics because of their smaller size. They can also be easily ingested and enter the food chain through different pathways as shown in Fig. 22.1. MPs are most commonly entered into the food web of marine through absorption. It has been observed when polystyrenes were absorbed by the marine alga (*Scenedesmus app.*) which undergoes oxidative stress due to the inhibition of photosynthesis (Bhattacharya

Table 22.1 Microplastic ingestion level of different coastal and marine biota of the coastal and marine ecosystems in the world (Adapted from Thushari and Senevirathna 2020)

Location	Contamination level	References
Seawater		
French-Belgian-Dutch coastline	0.4 parts/L	Van Cauwenberghe et al. (2015)
Hong Kong, China	3.973 pieces/m^3	Cheung et al. (2019)
Guanabara Bay, Rio de Janeiro, Brazil	1.40–21.3 particles/m^3	Glaucia et al. (2019)
Western English Channel	0.27 particles/m^3	Cole et al. (2014)
Northwestern Mediterranean Basin	0.116 particles/m^2	Collignon et al. (2012)
North Pacific Gyre	0.334 particles/m^2	Moore et al. (2001)
Caribbean Sea	0.001 particles/m^2	Law et al. (2010)
Gulf of Maine	0.002 particles/m^2	
North Atlantic Gyre	0.020 particles/m^2	
Atlantic	<0.1 particles/m^2	Doyle et al. (2011)
Mangrove Creeks, Goiana Estuary	3.4 items 100/m^3	Lima et al. (2016)
Río de la Plata Estuary	139 items 100/m^3	Pazos et al. (2018)
Madu Ganga estuary, Sri Lanka	40.06 ± 1.84 items/m^3	Praboda et al. (2020)
Sediment		
French-Belgian-Dutch coastline	6 parts/kg dry	Van Cauwenberghe et al. (2015)
Irish continental shelf	85% fibers (blue: 72%/red: 28%), 15% fragments	Martin et al. (2017)
Mediterranean Sea, SW Indian Ocean and NE Atlantic Ocean (across subtropical to subpolar waters)	1.4–40 pieces/50 ml	Woodall et al. (2014)
Subtidal region, United Kingdom	0.2–1 pieces/50 ml 6 pieces/50 ml	Thompson et al. (2004); Browne et al. (2011)
Southern Baltic Sea	0–27 particles/kg of bottom sediment d.w.	Graca et al. (2017)
Belgian coast	390 particles/kg	Claessens et al. (2011)
Arctic deep sea from the HAUSGARTEN Observatory	4356 particles/kg	Bergmann et al. (2017)
Belgium shelf	100–3600/kg	Leslie et al. (2017)
Dutch North Sea coast	54–3146/kg	Hall et al. (2015)
Guanabara Bay	8766 particles	Carvalho and Baptista Neto (2016)
Northern Gulf of Mexico estuaries, NA	13.2–50.6 items/m^2	Wessel et al. (2016)
Madu Ganga estuary, Sri Lanka	5.88 ± 1.33 items/100 g	Praboda et al. (2020a)

et al. 2010). Lusher et al. examined 507 fishes, and different kinds of microplastics were found in 37% fishes of which most common microplastics were polyamide and rayon (Lusher et al. 2013). Microplastics have also been found in all species of marine turtles (Tourinho et al. 2010) and in stomach and intestine of marine mammal (harbor seals) (Rebolledo et al. 2013).

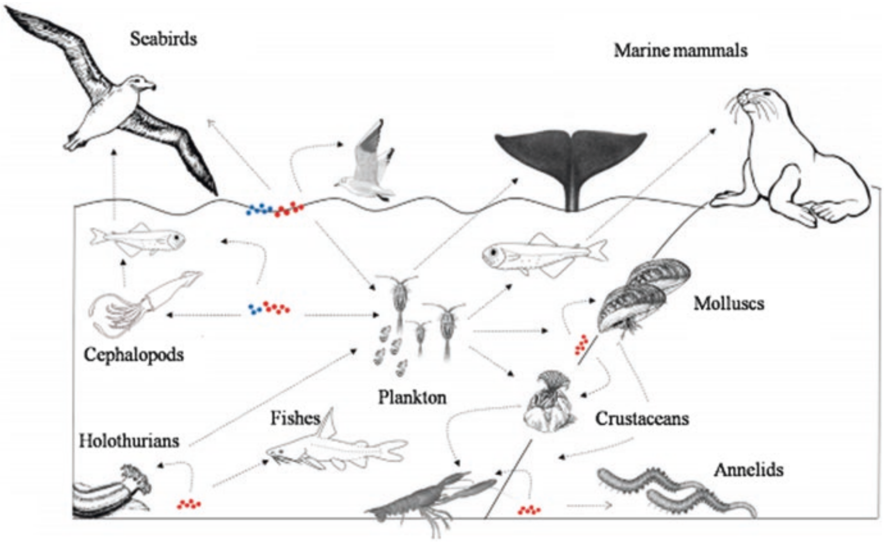


Fig. 22.1 An overall representation of the different pathways of microplastics entering into the food chain of vertebrates and invertebrates. Blue dots are the less dense microplastics (PE and PP) and red dots represent the more dense microplastics (PVC). (Adapted from (do Sul and Costa 2014)

22.5.2 Food

Digestion, inhalation, and skin contact are the main sources of human exposure to MPs (Prata 2018). When these microplastics enter the marine environment through different sources (Bergmann et al. 2015), they are ingested by fishes and other seafood. Fish and shellfish are extensively consumed by human beings due to their high protein contents. MPs decrease the nutritive quality of these seafood. Consumption of these contaminated food have very adverse effect on human health. MPs have also been found in all food products (De-la-Torre 2020). Every one of us ingest adequate amount of salt because it is an essential nutrient in our daily diet and enter into food web shown in Fig. 22.1. Salt is used in cooking and preservation of different items such as cheese, pickle, fruits, etc. Salt is also used in cosmetics in industries and in personal care products (Westerhoff et al. 2008). Yang et al. (2015) collected 15 different brands of salts from China market and found sufficient amount of MPs in all types of salts. Result of this research showed that microplastics were found in high amount in the sea salt than the lake and rock salt. The most common plastics were polyethylene, cellophane, and polyethylene terephthalate. MPs have the ability to translocate between the human organ systems. About 90% of the microplastics taken by the human beings are excreted through feces and the remaining 10% absorbed into the bloodstream (Smith et al. 2018). MPs cause the tissue damage, tissue necrosis, oxidative stress, nausea, diarrhea, infertility, cancer, chromosomal change, and obesity (Sharma and Chatterjee 2017).

22.5.3 Atmosphere

Air is a vital component for all living organisms on earth. Therefore, air must be pure and fresh. Due to the increasing population of the world, the demand of the plastics in different fields is increasing, which are polluting the air in different ways. The polluted air has very harmful impact on the living organisms (Verla et al. 2017). The airborne MPs are recently identified in the atmosphere in the indoor and outdoor environment. Due to the small size and low density, these airborne MPs can easily travel from one place to the other place in the atmosphere (Enyoh et al. 2019).

Many types of MPs (synthetic and natural) are detected in different forms. Different types of microplastics found in atmosphere include Synthetic (nylon, PVA, polyester, polyethylene, epoxy resins, polyurethane, and polyacrylamide), whereas natural plastic includes wool and cotton. These are present in different shapes or forms such as foam, fiber, microbeads, and granules. All these forms are produced through two ways: (1) by the degradation of larger plastic by UV and (2) by the degradation of clothing nurdles and health-care products (Enyoh and Verla 2019). Fiber is classified into synthetic and natural. Synthetic fiber includes polyester, polyamide, and polypropylene rayon. Natural fiber includes wool, cotton, silk, and asbestos (Gasperi et al. 2018).

Researchers observed that the distribution of MPs change with the climate. Zhou et al. (2017) observed that the percentage of MPs are very high in summer, winter, and spring, and in autumn, the percentage is very low. Human exposure to the MPs can be through inhalation, skin contact, dermal, and open meal. Small-size MPs can easily translocate within the body than the larger ones. Airborne microplastics present in air enter human body through inhalation of this polluted air. These small-size MPs after entering through the nostrils and mouth deposit on the upper airway, and some of them deposit in the lower air way (deep lungs) causing respiratory infections (Gasperi et al. 2018). Workers in the plastic industry also suffered severe interstitial lung diseases such as dysphonia, coughing, and decline in lungs functioning (Boag et al. 1999). Airborne MPs can enter the body through penetration whose size is equal to the skin pores and have more chances of penetration (Flament et al. 2015). MPs may also enter the body through the exposed meal. Catario et al. found that human may take MPs ranging from 13,731 to 69,415 through open meal (Catarino et al. 2018). Airborne MPs have ability to absorb the toxic substance (metals due to its hydrophobic nature) which in turn causes DNA damage, oxidative stress, cancer, and damage of immune cells as shown in Fig. 22.2 (Enyoh et al. 2019).

22.5.4 Soil

MPs are emerging soil pollutants which alter the soil structure and soil functioning (Boots et al. 2019) along with decrease in soil fertility that directly affect the human health (Stubenrauch and Ekardt 2020). Every year, 63,000–430,000 tons of MPs

Fig. 22.2 Effect of microplastics on the human health. (Adapted from Prata et al. 2020)

have been estimated in the land of Europe (Nizzetto et al. 2016). An estimated 700–4000 plastic particles per kilogram of soil are observed (Barnes et al. 2009; Briassoulis et al. 2010) out of which about 30 to 40 percent enter via organic fertilization (Stubenrauch and Ekardt 2020). Sewage sludge is scientifically proven highest source of MP in soil, and it is proposed that in Europe, about 0.2–8 mg of microplastics per hectare per inhabitant enter in the soil through sewage sludge annually (Nizzetto et al. 2016).

22.6 Chemical Technologies for the Degradation of MP Pollution

Degradation is the process of change in the physical and chemical properties of any substance. This process can be chemical or physicochemical. The causes of chemical degradation of polymers are hydrolysis or oxidation (Smith 2005). Chemical degradation of MPs depends upon class of polymer, chemical composition, presence of supplements (UV stabilizers), environment, and depositional settings (Corcoran 2020). It is observed that MPs on the littoral surface disintegrate more as compared to MPs in depth of ocean because they are directly exposed to the UV radiations (Leonas and Gorden 1993). The disintegration rate of microplastics in sea water and stimulated water is much more higher than fresh water due to variation in the various properties such as alkalinity, brininess, and biomass of two mediums (Weinstein et al. 2016; Da Costa et al. 2018). There are various technologies such as advance oxidation processes (photocatalysis, chlorination, UV radiations), coagulation, agglomeration, flocculation, etc., that are used for the degradation of MPs.

22.6.1 Advance Oxidation Processes

Microplastics can be degraded by the advance oxidation processes, i.e., photocatalysis, chlorination, photooxidation, ozonation, etc.

22.6.1.1 Photocatalysis

Various methods have been used for the elimination of microplastics from the environment, i.e., filtration, ozonation, incineration, etc. The drawbacks of these methods include generation of large amount of undesirable by-products along with high energy requirements. But the photocatalysis is budget friendly, feasible, energy efficient, and surface phenomenon (Tofa et al. 2019). Photocatalysts have been extensively used for the eradication of pollutants in the wastewater due to their sustainability and cleaning nature (Shi et al. 2020). The development period has four stages 1960s–1993, 1994–2000, 2001–2010, 2011, and recent (Long et al. 2020). In the photocatalytic removal of contaminants, semiconductors are used which, on the encounter with light, produces the electron and holes pairs. A large number of photocatalysts have been prepared and used throughout the history. These species interact with water and produces highly reactive oxygen species. The highly reactive species include superoxide anion and hydroxide radicals which degrade the pollutants from water (Yang et al. 2020). Electro-Fenton process was investigated for polyvinyl chloride degradation (Miao et al. 2020) and carbocatalytic oxidation and hydrothermal hydrolysis for the high-density polyethylene microplastics degradation (Kang et al. 2019) which however showed lower efficiency. Previously, it was thought that only the large-sized plastic can be removed by the photocatalysts (Shang et al. 2003). However, it is now thought that all types of MPs can be degraded by the photocatalysts. A photocatalyst should be selected according to the nature of pollutants (Ariza-Tarazona et al. 2020). Photocatalysis of the plastic produces low molecular weight compounds which can be used for the formation of new compounds. Photocatalysis in the presence of sunlight is very attractive because the sunlight is renewable source of light and is economical, active, and environmentally friendly (Bratovcic 2019). The process of photocatalysis is surface phenomenon which occurs by the interaction of light with photocatalyst and pollutant. When the photon has the energy greater than the band gap, the transfer of electron into conduction band occurred, and hole is produced at the place of electron. Water molecules adsorbed on surface of photocatalyst react with holes and form hydroxyl radicals. Superoxide anion radicals formed by interaction of electrons with oxygen. The pollutants are mineralized by these reactive species into water and carbon dioxide (Llorente-García et al. 2020).

Zinc oxide nanorods in the presence of visible light used for the degradation of low-density polyethylene microplastics and about 30% increase in the photooxidation of low-density polyethylene was observed in the presence of visible light (Talvitie et al. 2015). Hydroxy-rich bismuth oxychloride (BiOCl-X) has more degradation capacity than alone BiOCl (Jiang et al. 2020). Ariza-Tarazona suggested the use of feasible photocatalytic process to degrade HDPE microplastics which also reduce its chances of entering into the aquatic environment. The removal of HDPE microplastics present in facial scrubs was investigated by using TiO_2 modified with nitrogen on contact with visible light that showed remarkable sustainability and degradation capacity than the conventional ones at low temperature (0 °C) and low pH (3) (Ariza-Tarazona et al. 2020).

Polyamide microfibers can be removed from wastewater treatment plant by TiO_2 in presence of ultraviolet light, where 97% polyamide mass loss occurred within 48 h of treatment (Lee et al. 2020). The shape and the size of the MPs have considerable effect on the degradation properties of photocatalyst. Low-density polyethylene degraded more than high-density polyethylene in the presence of N-TiO_2 in oxygenated, and less illuminated medium and high surface-to-volume microplastics degraded more than low surface-to-volume microplastics (Llorente-García et al. 2020). Efficiency of photocatalysis depends upon the different factors such as light intensity, initial pH, flow rate, structure, and particle size. Degradation of polymethyl methacrylate and polystyrene from the wastewater using TiO_2-P_{25}/β SiC in presence of UV-A at low pH is higher than higher pH, and the 140 nm polystyrene degraded much more faster than 508 nm polystyrene (Allé et al. 2020).

Pyrrole/TiO_2 nanocomposites prepared by sol gel method in the presence of sunlight have the ability to remove polyethylene plastic. Weight loss of polyethylene plastic occurred on irradiation with sunlight for 240 h due to the strong bond between the pyrrole/TiO_2 and polyethylene plastic interface (Li et al. 2010). ^{14}C radiotracer technology in the UV light was used for the mineralization and degradation of polystyrene. ^{14}C styrene was used as precursor for the synthesis of ^{14}C polystyrene. The effect of water and air on photodegradation of polystyrene was checked by exposing the polystyrene to UV light in air or water. The characterization showed that cross-linking in polystyrene occurred in air, while in water, there was no cross-linking, and the mineralization was greater in water as compared to air. The greater photoactivity in water was the strong evidence of photodegradation of polystyrene in aqueous environment (Tian et al. 2019). TiO_2 nanoparticle films prepared from the Triton X-100 causes the complete mineralization of MPs than the UV-irradiated TiO_2 nanoparticle films (Nabi et al. 2020).

22.6.1.2 Photodegradation

Photodegradation is a process where substance/material is decomposed by light. It is one of the prime sources for the degradation of substrate under ambient conditions. Mostly, the degradation of synthetic polymers is initiated by the ultraviolet or visible light. Near UV light (400–299 nm) have the energies of 72–97 kcal/mol which is sufficient for breaking most of bonds (Rånby 1989).

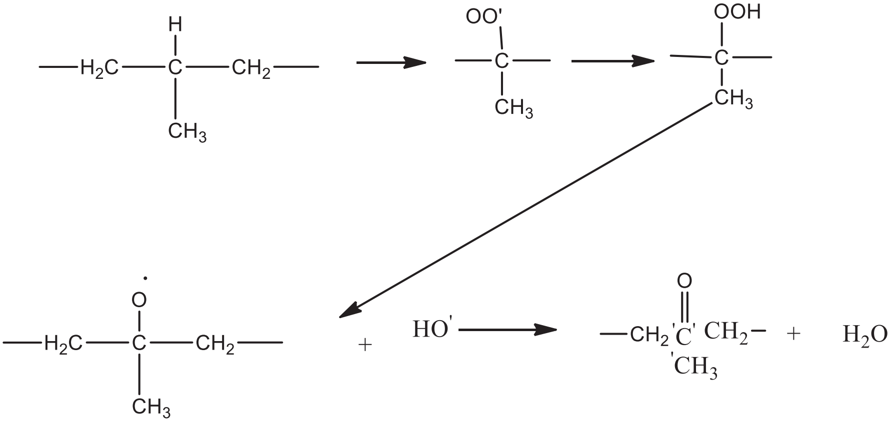
Normally, MPs converted into short chains on contact with oxygen and UV light, and presence of additives also enhances the degradation of MPs (Chamas et al. 2020). In the presence of UV light, the most common MPs such as polyethylene and polypropylene undergo C-H bond breakage resulting in the formation of free radical, and hence, molecular weight of MPs decrease (Gewert et al. 2015). Mechanical and chemical changes also occur on MPs' surface (Cooper and Corcoran 2010). The disintegration of MPs is relatively high in air than aqueous environment due to the presence of oxygen. Intensity of ultraviolet light also affects the degradation of microplastics in the environment. Cracks usually appeared on the top of plastics on degradation (Cai et al. 2018).

Degradation of Polypropylene

Polypropylene (PP) is a thermoplastic polymer that is used in packaging, labeling, textile, etc. PP is most extensively used because of its low cost and high processability. At moderate temperature, PP is resistant to photo and thermal oxidation. Under high temperature, the tertiary carbon is susceptible to attack by oxygen in the polymer (Zhao and Li 2006). Ultraviolet radiation over 290 nm initiates the degradation of PP and causes destruction of color. Bonds of PP under direct sunlight are responsible for high degradation of PP because bonds become weak (Yang and Ding 2006) as shown below in Scheme 22.1.

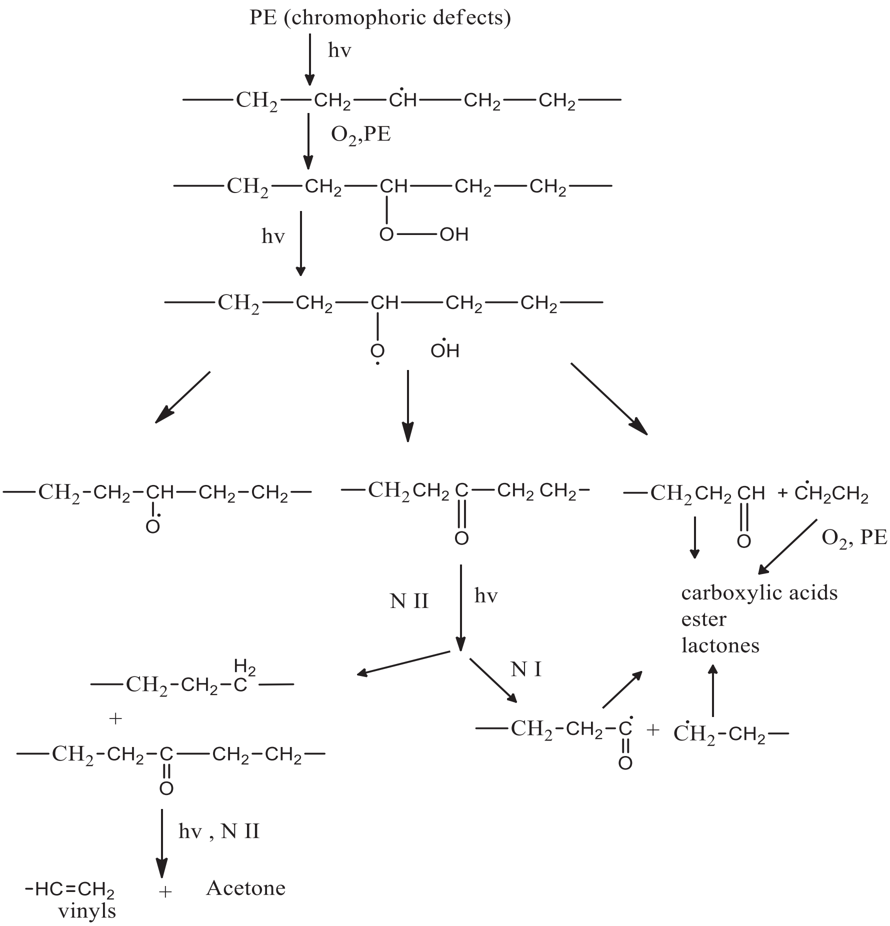
Photodegradation of Polyethylene

Polyethylene (PE) is the thermoplastic polymer that has been used in various applications, i.e., plastic bags, bottle caps, milk crates, fuel tanks, etc. There are different factors affecting the degradation of PE such as molecular weight, crystallinity, morphology, and branching of polymers (Smith 2005). The absorption of light by PE



Scheme 22.1 Polyester degradation. (Adapted from Rånby 1989)

molecule in the existence of oxygen results in the production of free radicals (macro-alkoxy and hydroxyl radical) which can further react in following pathways: cleavage of macromolecule to form aldehyde and removal of hydrogen atoms from macromolecule to form ketone (Carpentieri et al. 2011; Edge et al. 1991). In the first step, ultraviolet radiation is absorbed by the material which causes the formation of free radical followed by addition of oxygen into free radical and form hydroxyl radical along with carbonyl group. Additional exposure of UV light to this carbonyl resulted in two types of reactions: (i) The C-C bond undergoes cleavage, and two free radicals are formed, one of which is carbonyl radical which further participates in chain reactions, and (ii) the carbonyl molecule undergoes breakage and formation of C-H to form diradical moiety of the molecule, which then forms two smaller chains shown in Scheme 22.2.



Scheme 22.2 Degradation of PE. (Adapted from Gardette et al. 2013)

22.6.1.3 Chlorination

In the WWTPs, chlorine is used extensively as disinfectant due to its sterilization properties. Chlorine, as strong oxidant, has the ability to change the physical and chemical nature of MPs and can degrade it (Kelkar et al. 2019). Chlorination process can break the bond and introduce the new bonds into the MPs, and formation of bond between chlorine and hydrogen enhances its lifetime and toxicity (El-Shahawi et al. 2010). The concentration of MPs can be increased by chlorination due to their splitting (Lv et al. 2019). Aged MPs have more ability to absorb the chemicals than less aged microplastics (Wang et al. 2018). Kelkar et al. (2019) proposed the physical and chemical changes in microplastics due to chlorination. The results indicated that polystyrene degraded immediately while high-density polyethylene and polyethylene showed resistance at normal dosage. Chemical changes occurred in the high-density polyethylene and polyethylene on long-term exposure.

22.6.2 Coagulation/Agglomeration

Coagulation is employed by innumerable WWTPs to prepare the large-sized contaminated particles which can easily be separated (Hu et al. 2012). The coagulation process involves the use of iron and aluminum salts that bind to the waste material via ligands exchange mechanism (Chorghe et al. 2017). Coagulation is a technique to lower the concentration of MPs in the water bodies. Coagulation and flocculation processes produces large-sized particles that can be separated easily (Hu et al. 2012; Lee et al. 2012). Sedimentation and air flotation are two processes involved in the coagulation step for removing contaminants in the water (Enfrin et al. 2019). The chemical coagulants such as aluminum and iron slats are used because of their high efficiency and low cost (Shen et al. 2020). Nano−/microplastics agglomerates due to their uneven surface and size are unstable and readily dispersed in the water (Sumitomo et al. 2018). MP agglomerates are stabilized by using coagulating agents which can then be separated from water by skimming. The coagulation process involves the complex formation between the microplastics contaminants and coagulant by the ligand exchange (Chorghe et al. 2017).

Ariza-Tarazona et al. (2019) demonstrated the removal of polyethylene microplastics by coagulation and ultrafiltration. The experiment was conducted with different parameters such as concentration of coagulants, pH, and size of microplastics. The results showed that the aluminum was better coagulant than iron. The removal efficiency of Al^{+3} decreased by increasing the pH for the microplastics having smaller size. Skaf et al. (2020) recently studied the removal of microplastics using alum, the results of which showed that high doses of alum had better performance. The presence of surfactant did not cause the coagulation of microplastics. Rajala et al. (2020) compared the use of organic and inorganic coagulants for the elimination of microplastics from the water. The elimination efficiency was 99.4%, 90%, and 76.7% by polyaluminum chloride, ferric chloride, and polyamine, respectively. pH had no significant effect on the microplastic removal.

22.6.3 Electrocoagulation

Electrocoagulation is a process that provides a budget-friendly tertiary treatment process for microplastic removal (Shen et al. 2020). Electrocoagulation makes the coagulation simple due to the formation of coagulants by electrochemical means. Electrocoagulation has several advantages over the conventional coagulation, i.e., rapid process, independent of pH, budget friendly, and requirement of very small quantity of chemicals (Garcia-Segura et al. 2017). Electrocoagulation produces the less sludge because of in situ production of coagulant by electrolytic oxidation of anodic material (Moussa et al. 2017). Involvement of electricity in electrocoagulation causes the fusion of the tiny particles of oil resulting in the formation of large particles which can be separated (Mhatre et al. 2015). Electrocoagulation is complicated process in which application of electricity produces cations which ultimately form the flocs. From the cations to flocs, there are successive stages. First step is generation of cations for the production of microcoagulant of Al^{+3} and Fe^{+3} at the anode. Second step involved the loss of stability of pollutants and suspended particles in water under the influence of coagulants. Third step involves the coalescence of pollutants and microcoagulants to form the large flocs which undergo froth flotation or sedimentation (Shen et al. 2020).

Parren and coworkers used the electrocoagulation for the elimination of polyethylene microbeads from the contaminated water with the help of bench-scale reactor. The elimination efficiency was studied on the basis of different characteristics such as pH, concentration, size, conductivity of microplastics, and current density. Electrocoagulation begins by the production of metal ions in water from the electrodes by electrolysis. The reactions occurred at the cathode and anode. The removal of microplastics begins by formation of coagulant by reaction of metal ions which are released from the metal surface in water due to electrolysis with hydroxyl ions which are present in the system (Perren et al. 2018).

Collides get broken by the coagulant and stabilize the microplastics surface. The coagulant traps the microplastics. 90% removal of microplastic beads occurred by electrocoagulation. 99.2% removal occurred by optimization of conditions: pH of 7.5, NaCl concentration 0–2 g/L, and current density of 11 Am^{-2} (Perren et al. 2018).

22.6.4 Chemical Weathering

Chemical weathering is a way for the degradation of microplastics in the environment. Chemical weathering can be chemical and surficial. Weathering process changes the physical properties of microplastics. The microplastics undergo change in color, weight, and appearance and causes fragmentation (Corcoran 2020). Weathering action of environment alter the physicochemical characteristics of microplastics and converted into the form which is totally different from their initial form with which they released in the environment (Liu et al. 2020). For example, Song et al. (2017) irradiated the polyethylene, polypropylene, and polystyrene to UV light for 12 months along with 2 months mechanical abrasion. The results indicated that polystyrene and polypropylene undergo more fragmentation than polyethylene. Weathering causes the destruction of thermal and mechanical properties of microplastics because the polymer backbone undergoes the cleavage (Iñiguez et al. 2018). Lv et al. (2017) demonstrated that molecular weight of polypropylene decreases due to loss in tensile strength after the 1.5 years of outdoor weathering conditions such as light temperature and presence of oxygen. Weathering processes alter the chemical composition of microplastics. Recently, Dong et al. (2020) performed the micro Raman spectroscopy of weathered and nonweathered microplastics (polyethylene, polyvinyl chloride, polypropylene, polyethylene terephthalate). The Raman spectrum of weathered microplastics consisted of C and O bonds which appeared due to the oxidation of microplastics. Weathering of microplastics causes its fragmentation. It was investigated by Mailhot and Gardette (1992) that polystyrene on light exposure of wavelength greater than 300 nm produced the many oxygenated products.

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Chapter 23
Bibliometric Analysis of Emerging Trends in Research on Microplastic Pollution in Post-Paris Agreement and Post-COVID-19 Pandemic World

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Abstract Microplastic pollution has emerged as a severe transboundary threat to natural ecosystems, marine environments, and human and nonhuman health. Microplastic pollution and its consequent impacts on natural ecosystems and habitats have attracted the attention of experts, environmentalists, researchers, academia, decision-makers, and the governments across the globe. It is imperative to examine and analyze the existing trends and themes in microplastic pollution-related research. It is also important to identify the most productive countries, organizations, and journals focusing on microplastic pollution and its impacts. The analysis is also needed to pinpoint the keywords and thematic evolution in research on

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microplastic along with the most influential and effective research in the area. This study serves this purpose. The study uses a systematic bibliometric approach to trace out the most productive countries, organizations, sources, and documents on microplastic related research. The study also provides detailed analyses regarding collaborations among countries and organizations in research on microplastic pollution. It also provides a detailed analysis of keywords and thematic evolution regarding microplastic research. This chapter also finds some emerging trends in research regarding the COVID-19 pandemic and microplastic pollution. This chapter pinpoints prospects of research on microplastic pollution and its implications on natural ecosystems, marine environment, human and nonhuman health, and approaches for the effective control and management of microplastic pollution. The conclusion of this analysis also stresses the need for collective strategies and frameworks to manage the microplastic pollution in the COVID-19 outbreak and post-pandemic world.

Keywords Microplastic pollution · Bibliometric analysis · Marine environment · Wastewater · Fresh water · Health effects · COVID-19 and microplastic pollution · Bibliometric analysis

23.1 Introduction

The research on mitigation and adaptation got unprecedented momentum in the wake of the unveiled synthesis report on climate change (IPCC 2014) and the global Paris agreement (UNFCCC 2015). The efforts and policy interventions to reduce Greenhouse Gas (GHG) emissions were a major focus to achieve the targets set in the climate agreement (Ali et al. 2019) and the sustainable development goals (SDGs) (United Nations 2020). In addition to GHG emission control, plastic(s) and microplastic(s) pollution has been a trending area of research that attracted the attention of researchers, experts, and academia all over the globe. Plastic waste has been burgeoning due to an increase in demand for plastic products with an increase in population and economic growth (Ali et al. 2021). The annual global plastic production is about 150 million tons as of 2020. The plastic waste may possibly reach 12 billion metric ton by 2060 (Cox et al. 2019; Zhang et al. 2020). Due to the increase in consumption of plastics, the quantity of plastic items released in the environment is burgeoning. Microplastics are plastic particles less than 5 mm in size. The concept of microplastic was first presented in Thompson et al. (2004). The publications (articles) on the subject started in 2007 and has got momentum in following years.

The plastic items in the environment break down into millions of microplastics. The fragmentation occurs as a result of many routine activities such as mechanical degradation, the physical withering of large items, washing of synthetic textile, chemical degradation, UV degradation, and biological degradation (Horton and

Dixon 2018). The fraction of microplastics in global plastic accumulation is predicted to be 13.2% by 2060 (Sharma et al. 2021). Microplastics are pervasive and harmful pollutants. Microplastics originate from multiple sources, of which the fragmentation of large plastic materials is the primary source. Microplastics are discharged into the environment due to the mismanagement of commercial and industrial wastes. The mechanical processes and UV radiation also release microplastics. Other sources include air pollution, urbanization, ineffective plastic waste management, consumption, and production activities. Microplastics are ubiquitous and affect all kinds of environments due to chemical additives. The connectivity of environments leads to the movement of microplastics. The terrestrial environment is the dominant source of microplastics and contributes more than 70% to marine microplastic debris (Kumar et al. 2021). Besides, the airborne transportation of microplastics carries the microplastics emitted from vehicles and industrial machinery.

The number of studies on microplastic had grown rapidly due to the growing concerns of potential risks related to microplastic exposure. Such risks include marine microplastics, the presence of microplastics in a freshwater system (Reid et al. 2019; Su et al. 2016), soil pollution (Veerasingam et al. 2020), food contamination (Kedzierski et al. 2020; Toussaint et al. 2019), and biological ingestion (Rotjan et al. 2019; Zhong and Li 2020). Lately, much attention has been given to microplastics in aquatic environments (Murphy et al. 2016). Addressing the issue of microplastic pollution is an urgent need as COVID-19 has exacerbated the situation. The rise in global production and consumption of face masks and other personal protective equipment to prevent the transmission of the global pandemic led to huge plastic waste during COVID-19 (De-la-Torre and Aragaw 2021; Patrício Silva et al. 2021). The surgical face masks are made of polymeric materials which are a significant source of microplastic (Kumar et al. 2021). Such plastic particles waste is released into the environment and ends up in oceans posing a threat to aquatic lives.

The chapter fulfills multiple objectives: The first objective of the study is to identify the most productive countries regarding research on microplastic pollution. It also focuses on the quantity of research published and its impact measured by the citation index. The chapter also aims to explore how countries collaborate in research on microplastic pollution. Second objective is to provide a comprehensive analysis of the most productive organization and how these organizations collaborate with other organizations in the field of research on microplastic pollution. The third objective is to examine the most productive journals publishing research on microplastic pollution and its sources, composition, and impacts on human, nonhuman, environments, ecosystems, marine life, and fresh water. The fourth objective is to provide a comprehensive analysis of the most productive and highly influencing research in terms of their citations. The fifth objective is to provide a bibliometric analysis of the most frequently used keywords, thematic evolution, and factorial analysis of conceptual structure(s) of microplastic pollution-related research. The study also provides a comprehensive discussion on microplastic pollution, evolutional analysis of related research, and recent trends in the field of microplastic pollution.

23.2 Methodology for Bibliometric Analysis

23.2.1 Database Selection

The 1st step of a bibliographic study is to select a suitable database to collect the relevant articles published on a specific area of research. In the present bibliometric study, the Scopus database has been chosen to serve this purpose. Another reason to use the Scopus database is that it is larger than the Web of Science (WOS) and also includes the Medline that makes the Scopus far better than Medline (Sweileh 2020). Moreover, it is one of the largest databases with more than 23 thousand journals in every field (Falagas et al. 2008). It is very much convenient to search and export data from the Scopus database. It is also very easy to import and perform analysis on data exported from Scopus. It provides two techniques of search—a basic and advanced search. Complex and long search queries can be carried out to fulfill the goal with high levels of validity. It is worth noting that the Scopus allows the use of terms in titles or titles and abstracts or journal names or authors' names or affiliations (Sweileh 2020), making the search more detailed and comprehensive.

23.2.2 Search Strategy to Retrieve the Required Data

The second important step in a bibliometric study is to construct a reasonable search query that enables the data retrievers possible to gather as many relevant documents as possible but with minimum false-positive search results (Sweileh 2020). Following Sweileh (2020), several articles published as "bibliographic analyses" or "systematic reviews" or "bibliographic analysis" have been reviewed to develop a search query for microplastic pollution. The keywords used were "microplastic pollution" and "microplastic" or "COVID-19" or "surgical masks" or "environmental occurrence" or "sustainable waste management" or "bioremediation" or "single-use plastic" or "biomedical plastic waste" or "nanomaterials" to reach the relevant publications. The search generated 3188 documents including all types of documents from the year 2007 to 2021. The study was limited to only research articles published on microplastic pollution. There were 2617 research articles. Since the articles from the year 2007–2014 were less than 50 in each year, the articles published in these years were excluded from the bibliometric analysis in this study. The analysis is based on 2549 research articles during 2015–2021 till June 6, 2021. Overall, the trend of research in the field of microplastic is on the rise.

23.2.3 Validation of the Search Queries for Retrieval of Relevant Articles/Documents

Validation of the search queries is indispensable to confirm the relevance of the data retrieved from the database. To serve this purpose, two approaches were used. Firstly, following Sweileh (2020), the top 50 articles published in microplastic pollution were reviewed to ensure whether they fit within the scope of microplastic pollution. Using this approach, the false-positive results were excluded from the data file manually. Secondly, following Sweileh et al. (2018) and Sweileh (2020), the actual number of each authors' research articles is compared, through information obtained from Scopus profiles of the authors, with the number of articles obtained by the search query for active authors researching on microplastic pollution. Pearson Correlation Test was carried for this comparison. A robust and significant correlation is deemed to be the confirmation of search query validity (Sweileh 2020; Sweileh et al. 2018). The data was collected from the Scopus database and exported as CSV file. Microsoft Excel, VOSviewer Software program (van Eck and Waltman 2010), and Bibliometrix R-package software were used for data analyses.

23.3 Results of the Bibliometric Analysis

Bibliometric analysis of the articles published on microplastic pollution during 2015–2021 shows that 2549 research documents are published in 354 journals till June 6, 2021, as classified in the Scopus database. It is evident that the research publications on microplastic pollution have been increasing over the years, and it got momentum after the Paris agreement as to the sources of pollution, and its impacts on human life, natural environments, and ecosystems have attracted the attention of the experts and researchers in succeeding years. The number of research documents on microplastic pollution increased by 82% from 2015 to 2016. This increasing trend also got momentum in subsequent years as it increased to 105.5% during the 2019–20 period. This trend is still likely to continue as 619 research articles have been published in less than 6 months in 2016. Articles published in 2015 have the highest mean total citations per article (MTCA) of 192.2 and mean total citation per year (MTCY) of 32.03 as articles published in 2015 got a maximum of 6 years (see Table 23.1).

Table 23.1 The number of research articles on microplastic pollution published during 2015–2021 (June 6, 2021)

Year	*N*	MTCA	MTCY	CY
2015	50	192.20	32.03	6
2016	91	127.43	25.49	5
2017	141	102.68	25.67	4
2018	258	63.53	21.18	3
2019	455	31.08	15.54	2
2020	935	10.31	10.31	1
2021	619	1.59		0

N: total number of documents published in a country; *MTCA:* Mean TC per article; *MTCY:* mean TC per year; *CY:* citable years

23.3.1 Most Productive Countries in Microplastic Pollution Research

23.3.1.1 Bibliometric Analysis of the Most Productive Countries in Research on Microplastic Pollution

Bibliographic analysis of countries with at least one publication concluded 122 countries globally. Out of these countries, 47 are meeting the threshold of at least five publications during the sampled period with 1840 research articles published in these countries. More than 80% of these articles on microplastic pollution were published in the top 20 countries listed in Table 23.2. China is a leading county with 356 research articles which is 23% of 1501 articles published in the top 20 countries, whereas the United States, Germany, and the United Kingdom stand 2nd, 3rd, and 4th, subsequently, in the top 20 nations published on microplastic pollution. Out of 47 countries with at least five documents, the top 4 countries published more than 41% of the total published during 2015–2021 which is more than 51% published in the top 20 countries.

It concluded seven clusters with China as the leading country. The data retrieved from the Scopus database indicates that out of 122 countries in which the microplastic-related research documents published, 51 countries met the criteria of a minimum of four documents. The overlay visualization of the TLS of the bibliographic coupling with other countries is displayed in Fig. 23.1.

23.3.1.2 Bibliometric Coupling Analysis Based on Countries

Figure 23.1 displays the bibliometric coupling analysis of top 20 countries. The analysis has been carried out, and the network is visualized by VOS-viewer software. The nodes, in Fig. 23.1, indicate countries. The larger the node, the greater the influence of the node (respective country) on other nodes (other countries), whereas the line shows the mutual relationship between the nodes (countries). The different colors in Fig. 23.1 represent the years. The bibliographic coupling analysis based on

Table 23.2 Bibliometric analysis of top 20 countries publishing research on microplastic pollution

R	Country	*N*	*N* (%)	TC	TC/N	TLS
1	China	356	23.7	10810	30.4	520623
2	United States	158	10.5	7913	50.1	303563
3	Germany	128	8.5	7074	55.3	258253
4	United Kingdom	123	8.2	8672	70.5	273158
5	Italy	89	5.9	3461	38.9	175929
6	Spain	79	5.3	2177	27.6	145412
7	Netherlands	60	4.0	4583	76.4	138698
8	France	54	3.6	4519	83.7	166968
9	India	54	3.6	860	15.9	118729
10	South Korea	52	3.5	1575	30.3	82412
11	Australia	51	3.4	2034	39.9	96202
12	Canada	44	2.9	1405	31.9	73999
13	Brazil	43	2.9	1559	36.3	101524
14	Portugal	41	2.7	2482	60.5	106788
15	Norway	34	2.3	728	21.4	58548
16	Denmark	30	2.0	1199	40.0	58493
17	Hong Kong	30	2.0	1714	57.1	62030
18	Japan	28	1.9	449	16.0	51066
19	Turkey	24	1.6	618	25.8	42045
20	Iran	23	1.5	634	27.6	57241

N: total number of research articles published in a country; *N (%)*: percentage of each country's research papers published in top 20 countries; *TC*: total citations of a country; *TC/N*: total citations per document; *TLS*: total link strength

the top 20 countries' weight ranking reveals that the top 5 most influencing countries are China, the United States, Germany, the United Kingdom, and Italy with TLS of 520623, 303563, 273158, 258253, and 175929, respectively. It is also notable that the proportion of TLS for these countries in the top 20 countries has been 17.8, 10.4, 9.3, 8.8, and 6%, respectively. The coupling analysis unveils that China, the United States, Germany, the United Kingdom, and Italy cite many the same literature on microplastic pollution. It means the research on microplastic pollution in these countries has the same literature reference foundation.

23.3.1.3 Bibliographic Analysis of Country Collaboration on Microplastic Pollution

Figure 23.2 represents the country's collaboration on microplastic pollution research all over the globe. The analysis reveals 1466 entries of collaborations among countries with a maximum of 60 to 1 collaboration. China has the lead in collaboration on microplastic pollution-related research having 10.57% of total collaborations globally. China and the United States are the top collaborating countries with 60 collaborations which are 4.09% of total collaborations globally. Among the top 20

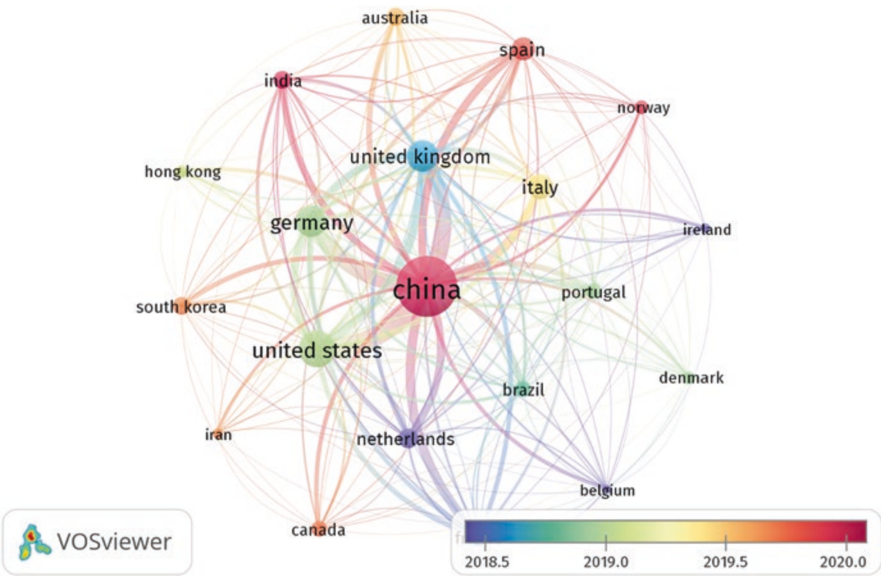

Fig. 23.1 Bibliographic coupling analysis of top 20 countries

No.	From-To	*f*	%
1	China-USA	60	4.09
2	China-Australia	28	1.91
3	China-Hong Kong	26	1.77
4	USA-Canada	21	1.43
5	China-United Kingdom	17	1.16
6	United Kingdom-Netherlands	17	1.16
7	USA-Australia	17	1.16
8	USA-United Kingdom	17	1.16
9	United Kingdom-Australia	15	1.02
10	China-Germany	14	0.95
11	Italy-Spain	14	0.95
12	Spain-Portugal	14	0.95
13	Italy-United Kingdom	13	0.89
14	Netherlands-Mexico	13	0.89
15	USA-Netherlands	12	0.82
16	France-Belgium	11	0.75
17	Germany-Norway	11	0.75
18	Germany-United Kingdom	11	0.75
19	Spain-France	11	0.75
20	China-Canada	10	0.68

Note: f = frequency; % = percentage of total global country collaborations

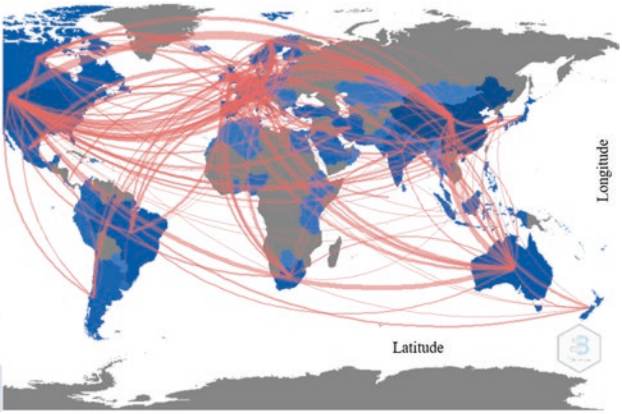

Fig. 23.2 Country collaboration map on microplastic pollution around the world

collaborating countries, China has the most effective collaboration with six major collaborations in the top 20 collaborations. Out of the top 20 collaborations, the top 3 collaborations come from China to the United States, Australia, and Hong Kong. The 4th leading collaboration on microplastic pollution-related research is between the United States and Canada with the frequency of 21 which is 1.43% of the collaboration worldwide. The United States has collaboration with four countries including Canada, Australia, United Kingdom, and the Netherlands. The United Kingdom has also been one of the leading countries in research on microplastic pollution in the world.

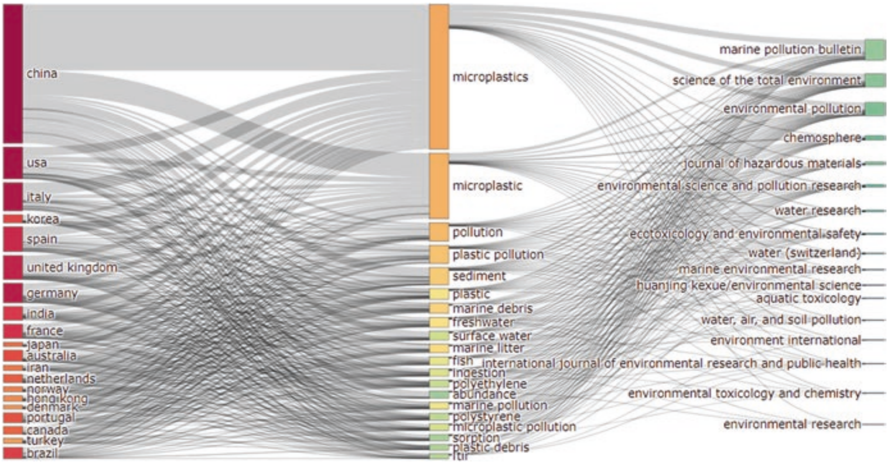

Fig. 23.3 Three-factor analysis of countries, keywords, and sources

23.3.1.4 Three-Factor Analysis of Countries, Keywords, and Sources

Figure 23.3 represents a three-factor analysis of the relationship among countries (left), keywords (middle), and the sources (journals) (right). The analysis shows that the top 20 countries all published "microplastics"-related research in the top 3 journals: *Marine Pollution Bulletin, Science of the Total Environment, and Environmental Pollution*. China has the lead in published "microplastics" and "microplastic"-related research articles followed by the United States and Italy.

23.3.2 The Most Productive Organizations/Institutions

23.3.2.1 Bibliometric Analysis of the Most Productive Organizations

Bibliographic analysis of organization shows that 15 out of 3588 organizations all over the globe meet the threshold of minimum five documents of an organization with four clusters. Table 23.3 summarizes the bibliographic analysis of the 15 organizations/institutions/universities publishing five or more than five documents. The analysis reveals that the University of Chinese Academy of Science (UCAS) has been a leading institution with 32 publications and a TLS of 6462 followed by the East China Normal University (ECNU), Shanghai, China, with 21 publications on microplastic pollution with 4988 TLS. However, publications from ECNU are leading in TC and TC/N of 3188 and 151.81 as compared to that of UCAS having 1743 TCs and 54.47 TC/N. It is interesting to note that 12 out of 15 top organizations are from China. Two institutions from South Korea and one organization from the Netherlands are included in the top 15 institutions publishing research on microplastic pollution.

Table 23.3 Top organizations publishing research on microplastic pollution

R	Organizations	Cluster	*N*	TC	TC/N	TLS
1	University of Chinese Academy of Sciences, Beijing, 100049 China	2	32	1743	54.47	6462
2	State Key Laboratory of Estuarine and Coastal Research, East China Normal University, Shanghai, China	1	21	3188	151.81	4988
3	College of Natural Resources and Environment, Northwest A&F University, Yangling, Shaanxi, China	3	9	125	13.89	1529
4	State Key Laboratory of Freshwater Ecology and Biotechnology, Institute of Hydrobiology, Chinese Academy of Sciences, Wuhan, China	1	8	869	108.63	2762
5	Wageningen Marine Research, Ijmuiden, Netherlands	2	7	873	124.71	1581
6	Key Laboratory of Plant Nutrition and the Agro-Environment in Northwest China, Ministry of Agriculture, Yangling, Shaanxi, China	2	6	12	2.00	1227
7	Shanghai Institute of Pollution Control and Ecological Security, Shanghai, China	2	6	24	4.00	701
8	University of Chinese Academy of Sciences, Beijing, 100039 China	1	6	563	93.83	2281
9	Agro-Environmental Protection Institute, Ministry of Agriculture of China, Tianjin, China	3	5	5	1.00	316
10	Guangdong Laboratory for Lingnan Modern Agriculture, South China Agricultural University, Guangzhou, China	1	5	5	1.00	1413
11	Key Laboratory of Aquatic Botany and Watershed Ecology, Wuhan Botanical Garden, Chinese Academy of Sciences, Wuhan, China	4	5	789	157.80	2527
12	Key Laboratory of Soil Environment and Pollution Remediation, Institute of Soil Science, Chinese Academy of Sciences, Nanjing, China	2	5	267	53.40	410
13	Laboratory for Marine Ecology and Environmental Science, Qingdao National Laboratory for Marine Science and Technology, Qingdao, China	1	5	316	63.20	1834
14	Oil and Pops Research Group, Korea Institute of Ocean Science and Technology, Geoje, South Korea	1	5	161	32.20	1036
15	Sino-Africa Joint Research Center, Chinese Academy of Sciences, Wuhan, China	4	5	789	157.80	2527

R: rank of the organization; *N*: total number of research articles published on microplastic; *TCs*: total citations; *TC/N*: total citation per document; *TLS:* total link strength

23.3.2.2 Bibliographic Coupling Analysis Based on Organizations

Figure 23.4 shows the bibliographic coupling analysis based on organizations. Nodes in Fig. 23.4 represent the organizations, and lines indicated the collaboration links. The thickness of the lines shows the total link strength of an organization on other organization(s). The visualization of the estimates of the overall strength of the bibliographic coupling links with other organizations shows that the CAS Beijing, China, is also leading in the collaboration with other organizations. The CAS has the strongest collaboration network with East China Normal University with a link strength of 1382.

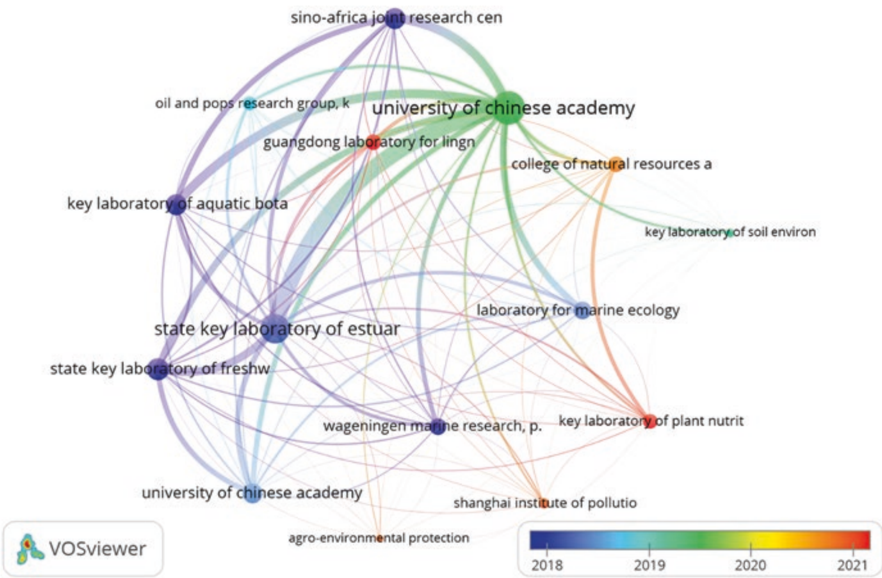

Fig. 23.4 Bibliographic coupling analysis based on organizations

23.3.3 The Most Productive Journals in Research on Microplastic Pollution

23.3.3.1 Bibliometric Analysis of Top 20 Journals Based on Published Articles on Microplastic Pollution

A bibliometric analysis of 2581 microplastic pollution-related publications published in 2015–2021 in 350 journals was carried out. The study finds out an average year from publication, average citation per document, and average citation per year per document of total articles published during the selected period as 1.54, 30.02, and 8.17, respectively. The analyzation of the top 20 journals publishing microplastic pollution-related research traced out that 1972 research articles published in these top 20 journals. The summary of the analysis is given in Fig. 23.5, and Table 23.4 shows the visual presentation of bibliographic coupling of the sources. Out of 1972 plastic pollution-related publications, *Marine Pollution Bulletin (MPB)* ranks first with 520 articles with 26.4% of the documents published in the top 20 journals during 2015–2021. The 2nd ranked journal is *Science of the Total Environment (STOTEN)* with 397 documents (20.1% of 1945) followed by *Environmental Pollution (EnP)* with 385 research articles related to pollution research producing 19.5% of the documents published in the top 20 journals during the selected period. Analysis of year-wise publications shows a higher number of documents published in *MPB* in each year from 2015 to 2021. It is interesting to note that 2/3 of the research papers have been published in the top 3 journals. *EnP*

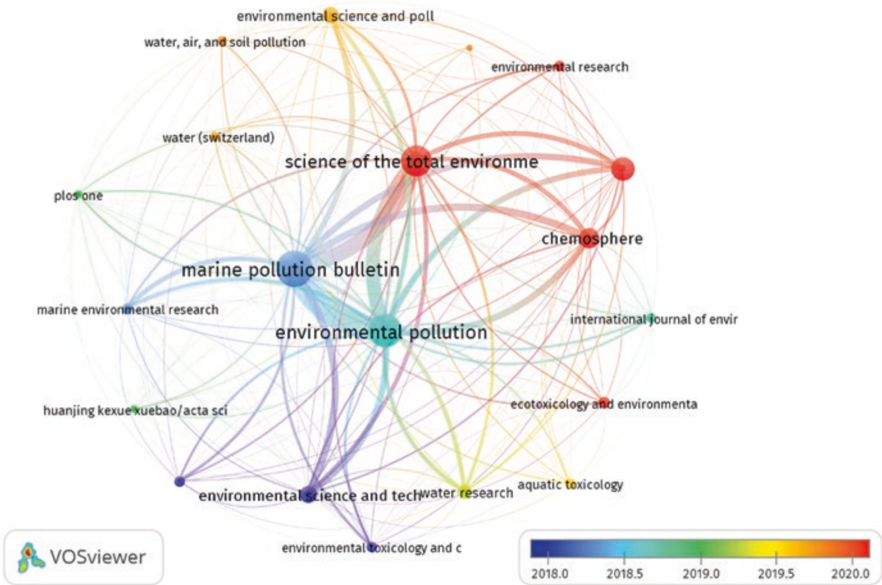

Fig. 23.5 Bibliographic coupling analysis of the sources

is a leading journal with respect to number of total citations (TCs) of 18452 followed by *MPB* with 15666 and *STOTEN* with 10382 TCs.

MPB is a leading journal started in 1970 with a 7.9 cite score and 4.049 impact factor (IF). It publishes research related to "rational use of maritime and marine resources."[1] Moreover, it also focuses on publishing research on marine pollution, effluent disposal, and pollution control. Since 1972, the journal *STOTEN* with a 10.5 cite score and 6.55 IF primarily focuses on research related to the total environment interfacing with "atmosphere, lithosphere, hydrosphere, biosphere, and anthroposphere."[2] In recent years, the journal has been publishing research related to nanomaterials, microplastics, and other emerging contaminants. *EnP* has been publishing documents focusing on environmental pollution and its impacts on ecosystems and human health. The journal has increased its publications and its quality of research published and increased the cite score and IF to 10.8 and 6.79, respectively.

[1] https://www.journals.elsevier.com/marine-pollution-bulletin

[2] https://www.journals.elsevier.com/science-of-the-total-environment

Table 23.4 Source dynamics of top 20 journals publishing articles on plastic pollution

R	Sources	*N* (%)	*N*	2015	2016	2017	2018	2019	2020	2021	H-Index	G-Index	TC	TC/N
1	Marine Pollution Bulletin	26.4	520	18	22	49	70	114	174	73	77	123	15666	30
2	Science of the Total Environment	20.1	397	0	3	4	32	72	160	126	67	102	10382	26
3	Environmental Pollution	19.5	385	8	22	28	60	78	113	76	58	86	18458	48
4	Journal of Hazardous Materials	6.3	124	0	0	0	2	5	37	80	38	78	1254	10
5	Chemosphere	6.0	118	1	3	2	5	19	41	47	27	48	2357	20
6	Environmental Science and Technology	4.5	89	3	5	13	11	19	32	6	24	45	6103	69
7	Environmental Science and Pollution Research	2.9	57	1	1	6	2	6	21	20	21	34	1192	21
8	Water Research	2.6	52	0	3	3	5	7	21	13	15	30	2984	57
9	Ecotoxicology and Environmental Safety	1.8	36	0	0	1	3	4	19	9	15	26	696	19
10	Water (Switzerland)	1.2	24	0	0	0	4	4	11	5	13	20	215	9
11	Environment International	1.1	21	0	0	1	0	4	14	2	13	17	846	40
12	Marine Environmental Research	1.1	21	2	3	1	1	1	8	5	10	16	1447	69
13	Scientific Reports	1.0	19	1	1	4	2	4	5	2	9	15	1553	82
14	Aquatic Toxicology	0.9	18	0	0	0	5	3	5	5	9	14	567	32
15	Environmental Toxicology and Chemistry	0.8	16	2	4	1	1	2	2	4	8	13	975	61
16	Plos One	0.8	16	0	1	0	2	1	10	2	8	13	320	20
17	Water Air and Soil Pollution	0.8	16	0	0	1	2	2	6	5	7	11	127	8
18	Huanjing Kexue/Environmental Science	0.8	15	0	0	0	0	4	7	4	6	11	133	9
19	Environmental Research	0.7	14	0	1	0	1	0	4	8	6	11	133	10
20	International Journal of Environmental Research and Public Health	0.7	14	0	0	1	5	1	5	2	6	10	262	19
Total publication[a]			1972	36	69	115	213	350	695	494				

Note: *R*: rank of the quantity ranking of the top 20 journals during 2015–2021 (May 30, 2021)
[a]Total publication per year by top 20 journals publishing papers on plastic pollution

23.3.3.2 Bibliometric Analysis of Topmost Cited Microplastic-Related Research Documents

Bibliometric analysis of the top 20 most cited research articles is summarized in Table 23.5. The mean value and range of TCs of the top 20 most cited journals are 427 TCs and 322–553 TCs. The most cite document also ranked 1st in the top 20 with the highest number of total citations 553 and 92 TCs per year is (Murphy et al. 2016) published in *Environmental Science and Technology.* The article examines the microplastic sources in the aquatic environment. Murphy et al. (2016) show that despite the efficient removal rates of microplastics using modern treatment technology, even a modest amount of microplastic per liter of effluent is being released into the environment and results in a huge amount of microplastic discharge. The second-ranked document with 552 TCs and 92 TCs per year is Sussarellu et al. (2016). The authors provide groundbreaking data on microplastic impacts providing help for the prediction of ecological effects on marine ecosystems. The study examines how exposure to polystyrene microplastics affects reproduction in oysters. The analysis reveals that microplastics cause feeding modifications, and reproductive disruptions in oysters also have a considerable effect on offspring. The study provides that strong reasons to believe that microplastics entered the marine environment are the source of concern, especially for filter feeders. Wright and Kelly (2017) (ranked 4th) examine how exposure to microplastic through diet and inhalation occurs and harms human health. The researchers review multidisciplinary scientific literature, gauge human health effects of microplastics, and outline some emergent areas of future research. Examining the accretion, particle toxicity, and chemical and microbial hazards, the corresponding existing fields reveal potential particle, chemical, and microbial hazards. The review anticipates chronic exposure to be a great source of concern. Van Sebille et al. 2015 (ranked 3rd) study the impact of microplastics debris in the ocean surface on marine life. The estimates reveal that in 2014, the accrued number of microplastic particles ranges between 15 and 51 trillion particles, weighing between 93,000 and 326,000 metric tons which makes it only approximately 1% of global plastic waste estimated entering the ocean in 2010.

The most of the top 20 studies consider municipal wastewater effluent as one of the major pathways for microplastic entrance into the aquatic environment, for instance, considering the engorgement that municipal wastewater treatment plants (WWTPs) are the major conduits of microplastic to the environment. Carr et al. 2016 (ranked 5th) investigate waste discharges from tertiary and secondary plants. The authors also probed the influent loads, particle size/type, transportation, and confiscation at WTPs. The researchers find out that existing wastewater treatment processes are effective in removing microplastic pollutants exerting into municipal WWTPs. The analysis does not unveil tertiary effluent as a significant source of microplastics as the latter is successfully removed during skimming and settling treatment processes. Mintenig et al. 2017 (ranked 8th) identified microplastic in effluents from four tertiary and eight secondary WWTPs. The polymer was identified of all microplastics down to the size of 20 μm through Micro-FTIR imaging. Microplastic was determined in all effluents of analyzed WWTPs. About 97% of

Table 23.5 Top 20 most global cited articles on microplastic pollution

R	Author(s)	Focus	TC	TC/Y	NTC
1	Murphy et al. (2016)	Examine how wastewater treatments work as a source of microplastic pollution in the aquatic environment	553	92	4
2	Sussarellu et al. (2016)	Assessment of the impact of polystyrene microspheres on the physiology of the Pacific oyster	552	92	4
3	Van Sebille et al. (2015)	The study represents the global estimates of microplastic abundance and mass using the largest dataset.	551	79	3
4	Wright and Kelly (2017)	Focuses on how microplastics affect human health	538	108	5
5	Carr et al. (2016)	Investigates the effluent discharge from seven tertiary and one secondary plant to examine microplastic in municipal wastewater treatment plants (WWTPs)	477	80	4
6	Avio et al. (2015)	The study analyzed how polyethylene (PE) and polystyrene (PS) microplastics adsorb pyrene.	470	67	2
7	Van Cauwenberghe et al. (2015a)	The study detects microplastics mussels and lugworms living in natural conditions.	456	65	2
8	Mintenig et al. (2017)	Identifies microplastic in effluents of WWTPs	420	84	4
9	Van Cauwenberghe et al. (2015b)	Reviews techniques, occurrence, and effects of microplastics in sediments	414	59	2
10	Klein et al. (2015)	Studies occurrence and special distribution of microplastic in the aquatic environment	408	58	2
11	Dris et al. (2015)	Investigates microplastic pollution in the urban environment	403	58	2
12	Brennecke et al. (2016)	Provides a deeper understanding of the microplastic vector for heavy metal contamination in the marine environment	393	66	3
13	Napper and Thompson (2016)	Examine the synthetic microplastic fibers released from washing machines	382	64	3
14	Lusher et al. (2015)	Analysis of microplastic in surface and subsurface samples from Arctic polar waters	369	53	2
15	Reid et al. (2019)	Discuss 12 major emerging threats to freshwater biodiversity	365	122	12
16	Su et al. (2016)	The occurrence of microplastics in the freshwater environment	354	59	3
17	Avio et al. (2017)	Attempted to examine plastic and microplastic pollutions in the oceans and emerging threats	347	69	3
18	Lei et al. (2018)	The study finds out how microplastics cause intestinal damage and other negative effects in *zebrafish Danio rerio* and *nematode Caenorhabditis elegans*.	346	87	5
19	Mason et al. (2016)	Detection of microplastic in municipal WWTPs	334	56	3
20	Leslie et al. (2017)	The study represents concentration data for the emerging contaminants in WWTPs, freshwater, and marine systems.	322	64	3

R: ranking of the papers based on citations; *TC*: total citations of the article(s); *Y*: number of year(s); *NTC*: normalized TC microplastic was removed with installed tertiary treatment. WWTP was found a possible source of microplastic but also the as sink as the latter was also detected in sewage sludge. Mason et al. 2016 (ranked 19th), analyzing the 17 different facilities across the United States, found fibers and fragments to be the most common type of particle within the affluent, but some fibers might be originated from nonplastic sources. The authors also observed inter- and intra-facility variation in discharge concentrations. Moreover, variations in the relative proportions of particle types were also observed. Leslie et al. 2017 (ranked 20th) clarified that treated municipal wastewater and solids were critical sources of microplastic pollution. "Riverine-suspended particulate matter" was found enriched in microplastic. Moreover, similar microplastic intensities were found in canal and treated wastewater. The authors observed that filter feeders and other benthos amassed microplastics in their bodies, and estuarine sediments were highly infested with microplastics.

Avio et al. 2015 (ranked 6th) examined the bioavailability and toxicological risk posed by the pollutants and examined how polyethylene and polystyrene microplastics effectively absorb pyrene. Further, pyrene absorbed on microplastics is readily available for mussels. It is also found that microplastics have several impacts on molecular and cellular pathways. Moreover, Avio et al. (2015) observe potential toxicological risks arising from virgin and contaminated microplastics. Van Cauwenberghe et al. (2015a) (ranked 7th) discussed the results of the study in the context of possible risks due to possible transfer of adsorbed pollutants detecting microplastics in mussels and lugworms living in natural habitats. The authors collected two species of marine invertebrates representing different feeding strategies from six locations along the French-Belgian-Dutch coastline. In addition, laboratory experiments were conducted to examine likely harmful impacts on ingestion and translocation of microplastic on the energy metabolism of the selected species. The analysis found microplastics in all collected organisms. It is also found that mussels and lugworms are exposed to high concentrations but no significant negative impact on the overall energy budget of the organisms. In another analysis, Van Cauwenberghe et al. (2015b) (ranked 9th) reviewed the technique, occurrence, and effects of microplastics in sediments and stressed the need for standardization and harmonization of abstraction techniques, occurrence, distribution, and impacts.

Klein et al. 2015 (ranked 10th), analyzing all sediments containing microplastic particles by infrared spectroscopy, found polyethylene, polypropylene, and polystyrene in abundance. Moreover, emerging pollution of inland river sediments with microplastic, and the rivers are vectors of transportation of microplastics into the ocean. Microplastic contaminations in urban areas are a source of great concern. Aspired from this argument, Dris et al. 2015 (ranked 11th) investigate the microplastic contamination in urban apartments and surface water in the continental environment. The pioneering study on the urban environment confirmed the microplastic in sewage, freshwater, and total atmospheric fallout and provided robust information on the type and size of the distribution of microplastics.

Most of the studies focus on the analysis of microplastics and their impressions on the marine environment, and the prevalence of microplastics in freshwater has been less explored. Reid et al. 2019 (ranked 15th) document 12 evolving intimidations to freshwater biodiversity including emerging contaminants, engineered nanomaterials, and micropollutants. Su et al. 2016 (ranked 16th) investigate microplastic pollution levels of freshwaters examining the Chinese Taihu Lake. The study found that abundance of microplastics in planktons was the highest in freshwater lakes worldwide. The prevalence of the highest levels of microplastics is found not only in water but also in organisms in the China's third largest lake. Lei et al. 2018 (ranked 18th) used zebrafish Danio rerio and nematode Caenorhabditis elegans as model organisms for microplastic exposure in freshwater pelagic and benthic environments. The researchers examined the toxic impacts of five common types of microplastics including polyethylene, polypropylene, polyvinyl chloride, and polystyrene. The results show that microplastic particles cause intestinal damage and other negative effects in organisms. The toxicity of microplastics is closely reliant on their size rather than composition.

There is only a single study examining the interaction between different classes of pollutants. Brennecke et al. 2016 (ranked 12th) attempted to examine the interactions between the two different classes of pollutants – heavy metal and microplastics. The authors examined the adsorption of two heavy metals leached from antifouling paints to virgin polystyrene beads and aged polyvinyl chloride fragments in seawater during the experimental manipulation of 14 days. The analysis showed a significant interaction between the microplastics and heavy metal showing implications its repercussions for marine life and environment. According to Napper and Thompson (2016) (ranked 13th), the release of synthetic microplastic plastic fibers is potentially a vital source of microplastic into the environment. It is also found that fiber release varied according to wash treatment with a variety of convoluted interactions, and it was concluded that washing clothing can a vital conduit of microplastic into the aquatic habitats.

In a pioneer study to examine microplastics in Arctic waters, Lusher et al. (2015) (ranked 14th) found microplastics in surface and subsurface samples in Arctic region. But the origins and corridors through which microplastic reached the Arctic region remained vague. The study recommended further research to develop a deeper comprehension of the microplastic sources and their impact on the environment. Avio et al. (2017) (ranked 17th) pointed out that plastic pollution has increased worldwide and a foremost risk to the marine environment. It has become ubiquitous, but there is a dire need for quantifiable estimates on the global abundance and weight of floating plastics particularly in the Southern Hemisphere and distant regions of the world. Even some large-scale convergence zones of plastic debris have been discovered, yet immediate standardized common methodologies are imperative to measure and quantify plastic in seawater and sediments. Moreover, plastic contamination has been affecting marine species that call for a more integrated ecological risk evaluation of these materials on a priority basis.

Bibliographic coupling of the documents shows that only 24 research articles fulfill the threshold level of a minimum of 300 citations. Estimations of the total strength of the bibliographic coupling links of the top 20 documents with other documents indicated two clusters of documents (Fig. 23.6).

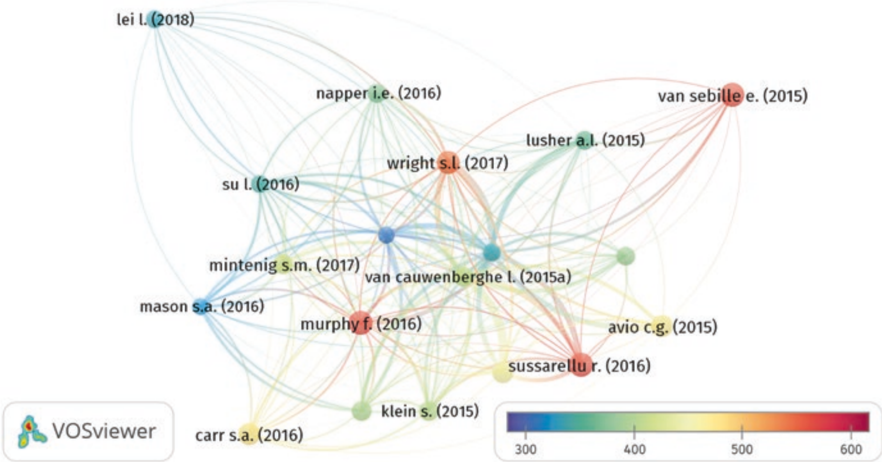

Fig. 23.6 Bibliographic coupling of documents

23.3.4 Bibliometric Analysis of Keywords, Thematic Evolution, and Factorial Analysis of Conceptual Structure

Out of 3262 author keywords, 139 met the threshold level of a minimum of 50 appearances. The visualization of the keyword clusters in Fig. 23.7 unveiled three major clusters. Cluster 1 mainly contains the words microplastic, microplastics, microplastic pollution, and composition and structure of microplastic pollution with microplastics as the most used word. Cluster 2 focuses on environmental monitoring, water pollutants, marine pollution, marine environment, geological sediments, sediments, plastic pollutions, and their impacts on the marine environment, whereas cluster 3 comprises the keywords related to impacts of organic pollutants, nanoparticles, ingestion and ingestion rate, toxicity, and their impacts on humans, nonhumans, and marine life. Risk assessment of exposure to microplastic pollution is also included in cluster 3.

Thematic evolutional analysis shows a very detailed process of evolution of keywords used and themes that emerged during the 2015–2021 period. The 7-year sampled period has been divided into four slices in 2015–2016, 2017–2018, 2019–2020, and 2021 to have a deeper insight into the thematic evolution of microplastic pollution-related research. Figures 23.8 and 23.9 show the visualized results of the thematic analysis. In 2015–2016, the researchers focused on microplastic, marine litter, persistent organic pollutants (POPs), abundance, plastic ingestion, and polystyrene. The research on polystyrene has been an emerging field in microplastic research in 2015–2016 as it can be seen in Fig. 23.9a. Research themes regarding microplastics, sediments, plastic ingestion, and polystyrene have the major themes attracting the attention of the researchers in Slice 2 (2017–2018) (Figs. 23.8 and 23.9b). The themes related to Fourier Transform Infrared (FTIR) spectroscopy,

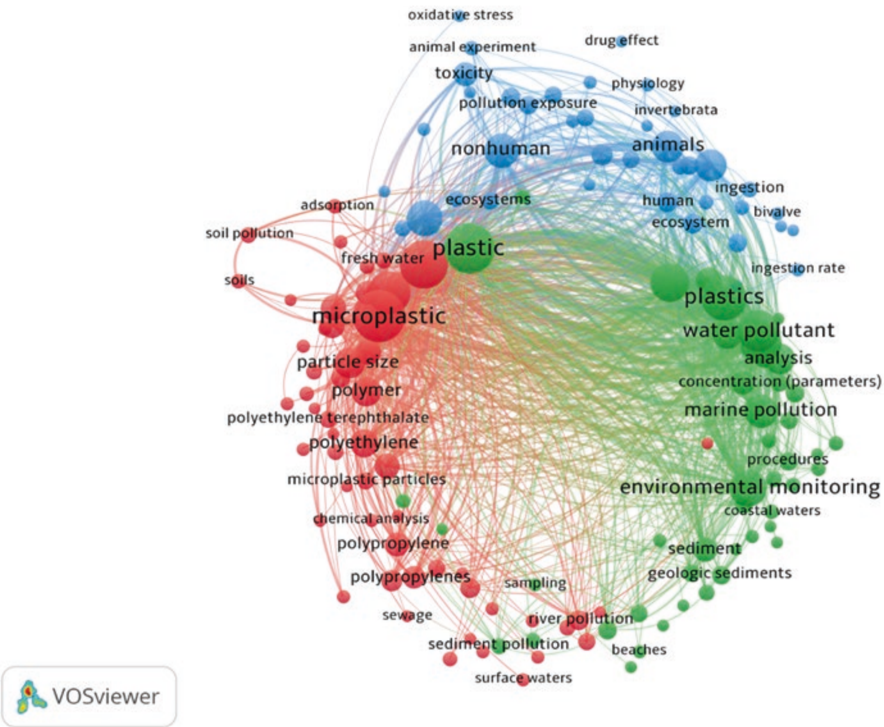

Fig. 23.7 Bibliographic visualization of keyword clusters

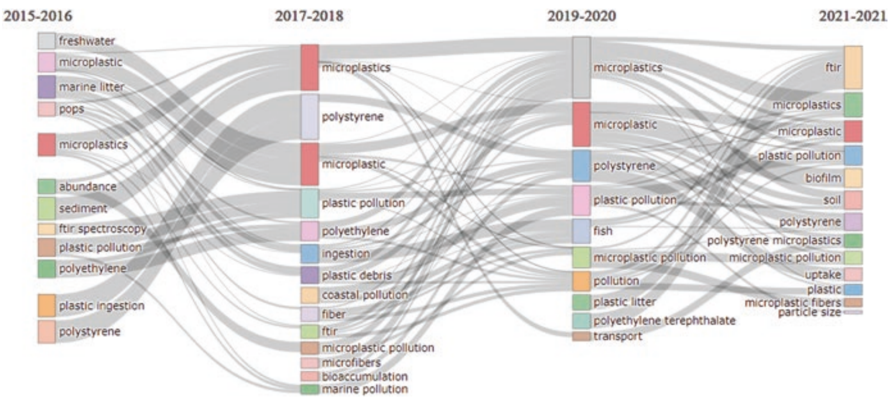

Fig. 23.8 Thematic evolution analysis

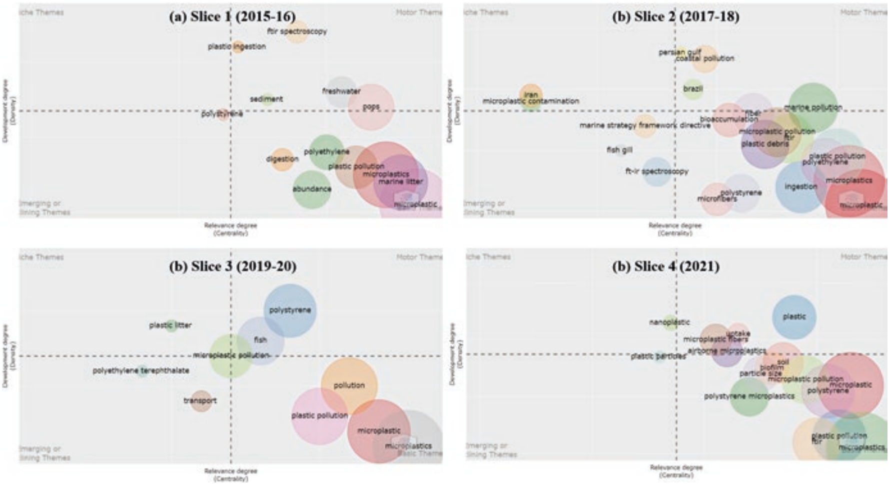

Fig. 23.9 Thematic evolution of the research on microplastic pollution: slice-wise analysis

polyethylene, microplastic debris, coastal pollution, microplastic pollution, microfibers, bioaccumulation of microplastic, and marine pollution increased in Slice 3 (2019–2020). Microplastic pollution has been the central theme of research in the same slice, whereas polyethylene terephthalate and transport have been the emerging field of research in Slice 3 (Fig. 23.9(3)). In Slice 4, the FTIR spectroscopy, microplastic(s), and plastic pollution have been leading themes of research in microplastic pollution research. FTIR spectroscopy has been an extensively used technology in research on microplastic pollution (Veerasingam et al. 2020).

Factorial analysis of conceptual structure of microplastic pollution (Fig. 23.10)-related research shows that major latest research on microplastic pollution research mainly focuses on FTIR spectroscopy, sediment, polypropylene(s), sediment(s), river pollution (fresh water), and polyethylene and is focused on microplastic research in China. Keywords and thematic analysis show that microplastic(s), microplastic pollution, marine litter, microplastic contamination, polystyrene microplastics, FTIR, ingestion, bioaccumulation, coastal pollution, pops, and analysis of particle size have been major themes of interest for the microplastic pollution-related research.

23.4 Discussion

The bibliographic analysis of the research articles published shows that the research on microplastic pollution got momentum after the publication of synthesis report on climate change (IPCC 2014) and the global Paris agreement (UNFCCC 2015).

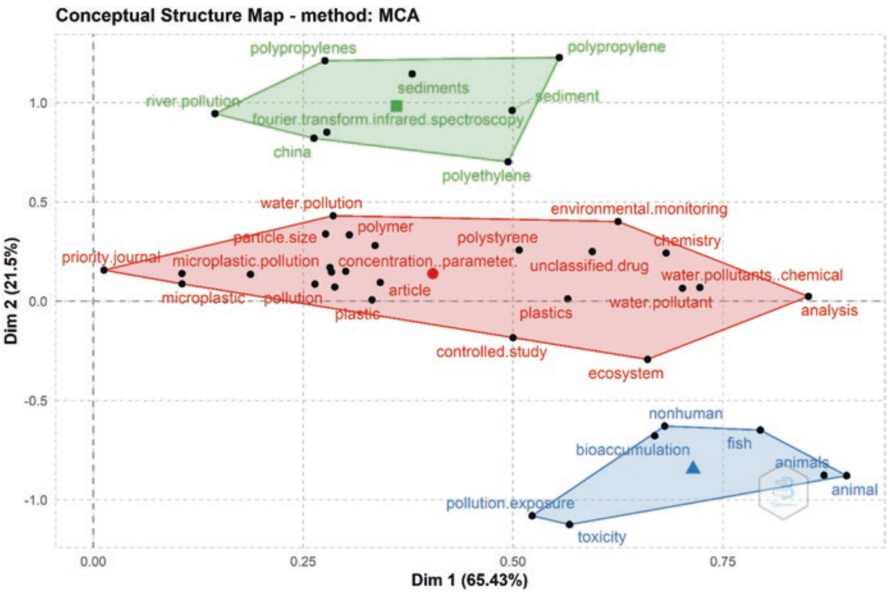

Fig. 23.10 Factorial analysis (conceptual structure map)

Moreover, the collective commitment of the global economies to sustainable development goals (SDGs) (United Nations 2020) also contributed to this momentum. The number of articles increased by 1770% in 2020 as compared to published in 2015, whereas this number of documents published on microplastic pollution is higher at 1138% in 2021 (on June 6, 2021) as compared to those published in 2015. The number of publications in less than 6 months of 2021 is higher than in 2015. However, it is likely to be much higher by the end of 2021. The results of the current study are in agreement with that in Sorensen and Jovanović (2021). The number of publications increased 2323.1% from 2009 to 2019 (Sorensen and Jovanović 2021).

The bibliometric analysis of the most productive countries shows that China is leading and the most effective country out of 122 countries in terms of research article publications on microplastic pollution during 2015–2021 followed by the United States and Germany, respectively. He et al. (2020) also find these three countries the most productive countries in research on microplastics in terrestrial ecosystems. Moreover, based on bibliographic coupling analysis of countries, China is also found to be more productive and collaborating with other countries. A bibliometric mapping and analysis of the management of plastic waste (de Sousa 2021) find a significant increase in research publications on plastic pollution, and China is found to be the most influencing in research articles published on plastic pollution. Moreover, analysis of the top organizations conducting and producing research on microplastic reveals the Chinese organizations/institutions having the lead in this research area as 12 of 15 top organizations have been publishing articles on microplastic pollution. It is important to note that 113 out of 130 research articles published in top 15 organizations worldwide are produced by the Chinese organizations/institutions which makes 86.92% of total publications by top 15 organizations. In addition, the Chinese organizations are also leading research impact measured by TC per research document published. The UCAC has the lead in publishing articles on microplastic pollution. This analysis produces the similar results as in de Sousa (2021) and He et al. (2020) which also find Chinese authors and organizations/institutions leading in the plastic pollution-related research. The analysis of the most productive journals reveals *MPB, STOTEN,* and *EnP* are deemed to be the top 3 productive journals in microplastic pollution-related research. He et al. (2020) also find these three journals leading the research related to microplastics in terrestrial ecosystems, but the ranking of these journals is *STOTEN, EnP,* and *MPB* as 1st, 2nd, and 3rd, respectively. The results of the current study are also supported by Pauna et al. (2019) and Sorensen and Jovanović (2021) that *MPB* has published the most documents on microplastic pollution. Pauna et al. (2019) conducted a bibliometric network analysis of global scientific literature on "marine microplastics."

Bibliometric analysis of the topmost cited microplastic-related research articles shows that some studies focused on sources of microplastic waste in the marine environment. For instance, Murphy et al. (2016) concluded that despite efficient removal of plastic waste from the wastewater, the effluents have been discharged and causing accumulation into the environment. Sussarellu et al. (2016), using a groundbreaking data on microplastic impacts, found microplastics affecting the reproduction in oysters. Wright and Kelly (2017) anticipates that chronic exposure to accumulated particle toxicity and chemical and microbial hazards affect human health. Some of the studies focused on the analysis of accumulation of microplastic debris in marine environments. For instance, Van Sebille et al. (2015) traced out how microplastic particles accumulated in the shapes of plastic marine debris. Most of the top research documents attempt to explore how municipal wastewater effluents work as a conduit of microplastic into the aquatic environment. Carr et al. (2016) find out that existing wastewater treatment processes are effective in the removal of microplastics from WWTPs through tertiary plants. Mintenig et al. (2017) focused on FRIR imaging to identify the size of the microplastics in tertiary and secondary WWTPs. Mason et al. (2016) observed inter- and intra-facility variation in discharge concentrations. Leslie et al. (2017) detected microplastic concentration in freshwaters such as canal water and treated water. Some studies attempted to examine toxical risks in marine inhabitants. Virgin and contaminated microplastics pose potential toxicological risks to marine mussels (Avio et al. 2015). In another study, Van Cauwenberghe et al. (2015a) reveal that mussels and lugworms are exposed to a high concentration of principle laboratory experiment but no significant impact on the organism's overall energy budget. Klein et al. (2015) find polyethylene, polypropylene, and polystyrene in abundance in river shore sediments. A pioneering study on the urban environment, Dris et al. (2015) confirm the existence of microplastic in sewage and fresh water. Su et al. (2016), Lei et al. (2018), and Reid et al. (2019) analyze microplastic pollution and its impact on freshwater. Brennecke et al. (2016) explore the interaction between heavy metals and microplastics. Synthetic microplastic plastic fibers are also a critical and potential source of microplastic into the environment (Napper and Thompson 2016). It is also imperative to analyze the microplastic in Arctic waters. Lusher et al. (2015) serve this purpose and found surface and subsurface samples, but the origins and pathways of microplastics to Arctic region still need to be explored. Pauna et al. (2019) unveiled that the research on marine microplastic primarily focused on toxicology and environmental chemistry. Pauna et al. (2019) stressed the need of adoption of interdisciplinary perspectives in marine microplastics-related research.

Since the inception of the COVID-19 pandemic, the problems related to plastic and microplastic pollution have increased enormously. In the wake of the COVID-19 pandemic and the preventive measures recommended by the World Health Organization (WHO) to slow down the infection, transmission has been a great source of plastic pollution. A substantial increase in the use and production of face masks and other elements such as gloves, face protectors, protective suits, and safety shoes manufactured with polymeric material including antiviral textiles has been ending as microplastic pools (Ardusso et al. 2021). In recent research, the consequent increase in plastic and microplastic pollutions due to the pandemic and issues related to it has attracted the attention of environmental researchers and experts.

The research themes and trends have shifted to COVID-19 and pollution. The unprecedented increase in the production of face masks during the COVID-19 pandemic has emerged as a new environmental challenge globally (Aragaw 2020). Disposable face masks (DFMs) have been used to slow down the transmission rate of COVID-19. Consequently, extensive use of single-use DMCs is playing a critical role in microplastic pollution. This is another source of concern amid the COVID-19 pandemic for the researchers and communities warranting the measures and policy interventions to address the microplastic pollution in pandemic situations (Fadare and Okoffo 2020). Chowdhury et al. (2021) also assert that face masks are a considerable nonrecyclable plastic material. Moreover, it is also a source of concern that wearing masks poses microplastic inhalation risk, and reusing the masks increases the risk (Li et al. 2021). Anastopoulos and Pashalidis (2021) analyze the role of face masks and subsequent mask-driven microplastic as pollutant carriers in the hydrosphere, biosphere, etc. Single-use face masks enter the uncontrolled environment, and safe disposal of the masks has been a challenging issue. Single-use surgical face masks can act as dye carriers (Anastopoulos and Pashalidis 2021).

Moreover, unprecedented and increased use of the PPEs during COVID-19 pandemic situations has worsened the plastic pollution issues in the marine environment (De-la-Torre et al. 2021; De-la-Torre and Aragaw 2021). It is stressed to address the key research needs regarding the occurrence and abundance of PPEs, sources, fate, and drivers of PPEs, PPE as sources of microplastics and vectors of invasive species and pathogens and source and vector of chemical pollutants in the marine environment (De-la-Torre and Aragaw 2021). Torres and De-la-Torre (2021) stress the need of using biodegradable face masks as an alternative to reduce nonbiodegradable plastic waste pollution. In a recent study, Ardusso et al. (2021) provide reflections and perspectives on how the pandemic causing aggravated plastic pollution on beaches and coastal environments consequently would be increasing devastation to the marine environments and ecosystems.

23.5 Conclusion

The research on microplastic pollution, its composition, and its impact on natural environments, ecosystems, marine environments, and human health has shown increasing trends since the global Paris agreement to deal with climate change and global warming. This chapter primarily focused on identifying the most productive countries, organizations/institutions, research journals, the most productive research articles, author keywords, and the evolution of various research themes during the sampled period of 2015–2021. The bibliometric analysis concluded China, the United States, and Germany the most productive in producing research on microplastic pollution. In addition, these countries are also found to be more productive and effective in terms of collaboration in the field of research on microplastic pollution. The analysis of the most productive organizations/institutions has revealed that the Chinese organizations/institutions are the most productive in research on microplastic pollution. The UCAS has been the leading organization/institution in producing research on microplastic pollution.

While analyzing the major sources (journals) publishing research on microplastic pollution and its composition, sources, and impacts on natural environments, ecosystems, human and nonhuman life, and marine environments, the journals *Marine Pollution Bulletin, Science of the Total Environment,* and *Environmental Pollution* have been the leading journals in the research area. The examination of the best papers in terms of research influence measured by the total citation per research article shows that the top publications focused mainly on marine environments, sources of the microplastic pollution, the composition of microplastic, and their impacts on natural environments and ecosystems with a special concentration on its impacts on human and nonhuman health. Most of the productive research articles addressed the microplastic pollutions in marine environments and fresh waters. Some studies focused on microplastic pollutants in wastewaters. The author keyword and thematic evolution analysis show that the frequent keywords and themes in research during the sampled period have been microplastic(s), plastic(s), water pollutants, concentration, particle size, polyethylene, polymer, polypropylene, sentiments pollution, coastal waters, fresh waters, plastic ingestion, bioaccumulation, marine pollution, FTIR, polystyrene microplastics, to name a few. A new research area that emerged in microplastic pollution-related research has been the microplastic pollution in the COVID-19 pandemic since the inception of the global pandemic. The recent studies in this area are primarily focused on COVID-19 and microplastic pollution, face masks, PPEs, and plastic wastes during COVID-19.

However, a few studies have focused on the strategies and policy frameworks to control microplastic pollution. The focus of the researcher has the identification of the microplastic(s), their composition, sources, conduits, and their impacts on the natural environments, ecosystems, human and nonhuman health, and marine life. The current bibliometric analysis finds it imperative to represent multiple research prospects that can be useful in setting future research trends on strategies to control and effectively manage the microplastic(s) pollution and issues related to it. An important future research area may be the analysis of microplastic pollution in far-off regions such Arctic region. It is important to examine the sources, conduits, and composition of microplastic pollution in far-off areas such as the Artic region. Moreover, it is also pivotal to evaluate the measures to control microplastic pollution and frame out strategies to deal with it in the less developed and developing countries. There is a need to adopt hybrid approaches to manage fresh water as crucial ecosystems for human life as well as essential hotspots of biodiversity and ecological function. Future research may focus on bridging the gaps between conservation of biodiversity and accelerated rate of species endangerment and stimulate the efforts to reverse the global trends in freshwater degradation (Reid et al. 2019).

It is imperative to put forth resources and strategies to mobilize and increase awareness on COVID-19 prevention, but it is also indispensable to increase public awareness on the use and disposal of the waste and its management. It is also imperative to direct research to look for eco-friendly alternatives along with enhanced effective waste management systems. The future research directions are likely to focus on the development of waste management systems, framing, and implementation of strategies for integrated coastal management. Since the COVID-19 pandemic has been adversely affecting all economies on the globe, the collective global strategies toward the developments in environmental research on COVID-19 and post-pandemic would be productive to deal with the various global environmental risks.

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