7 Environmental Toxicity, Health Hazards, and Bioremediation Strategies for Removal of Microplastics from Wastewater

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Abstract

Microplastics (MPs) are minuscule plastic particles smaller than 5 mm in length that have become a significant threat to because of their toxicity in our natural environment and detrimental impacts on our water resources, aquatic life, and humans. Physical, chemical, ecological, and biological impacts are all possible ways of causing dangers posed by MPs. Microplastics also sorb and collect potentially toxic contaminants in aquatic environments. As a result, ingesting polluted microplastics may expose marine species and even the food chain to hazardous contaminants. However, wastewater treatment plants (WWTPs) are the primary source of microplastics that enter marine ecosystems. Microplastics in aquatic environments must be controlled to protect the environment and human health. This chapter examines the sources of microplastics in wastewater, their properties, ecotoxicity, and health risks, existing and newly developed methods for characterization of microplastics in wastewater, and for pollution prevention and control, bioremediation techniques for the removal of microplastics from wastewater have been developed.

Keywords

Wastewater · Microplastics · Toxicity · Health hazards · Characterization · Bioremediation

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7.1 Introduction

Industrial waste is the principal source of environmental pollution because of the existence of nutrients of environmental concerns, potentially toxic heavy metals, organic pollutants, and emerging contaminants that pose major ecotoxicological risks and environmental dangers (Chandel et al. 2022; Chaturvedi et al. 2021; Saxena et al. 2016, 2020a, b, c, d; Deb et al. 2020; Kumar et al. 2020; Bharagava and Saxena 2020; Mulla et al. 2019; Bharagava et al. 2017a, b, c, 2018; Goutam et al. 2018; Gautam et al. 2017; Saxena and Bharagava 2015, 2017). Among the environmental contaminants, the release of emerging contaminants along with industrial effluents is a major environmental concern. The extensive use of plastic goods in today's world eventually leads to emission of minute plastic particles into the environment. The diameter of the microplastic particles (MPs) is less than 5 mm (GESAMP 2015). The exact level of microplastics in the environment, including unidentified microplastics, is considered to be substantially higher than what uncontrolled plastic product flows predict (Kim et al. 2015). The microplastics (MPs) amount in seawater has been continuously increasing over the last decade, with a growing trend along the shorelines (Barnes et al. 2009), MPs pollution is a relatively new issue in the world, due to the growing use of plastics in practically all aspects of human activities and there is a lack of appropriate treatment of domestic and industrial wastewater (Bui et al. 2020).

At present, with the widespread use of MPs, particularly in the marine environment, marine life is unsheltered to MPs with broad range of effects which depends on the presence toxic chemicals from plastic additives and adsorbed pollutants such as pesticides, persistent organic pollutants, or metals leaching into the environment, particularly in the marine environment (Van Emmerik et al. 2018; Fossi et al. 2014). MPs are hazardous and can also serve as pathogen reservoirs, putting marine life in danger (Kor and Mehdinia 2020). MPs are found largely in coastal habitats, and their exact influence on human health has yet to be identified. Marine life, on the other hand, is at the centre of the food chain and provides a significant portion of the nutrients consumed daily by human beings (Bui et al. 2020). The growing presence of MPs in the environment and biota has attracted the curiosity of scientists and the general public, with emerging evidence of microplastics' detrimental effects (de Sá et al. 2015; Jeong et al. 2016). Surface runoff, wind advection, and WWTPs effluent are just a few of the ways MPs enter water bodies (Dris et al. 2015). Thousands of microplastic particles are deposited in WWTPs every day (Okoffo et al. 2019). Although there is no direct link between MPs concentrations and population density in WWTPs intake streams, agriculture and industrial activities appear to be important factors (Long et al. 2019). To determine the amount of microplastics that enter and exit WWTPs, it is essential to develop a precise and repeatable experimental approach for counting the microplastic particles in sewage influent and effluent.

MPs have been removed using a variety of processes, including grit chamber and primary sedimentation, coagulation, sand filtering, dissolved air floatation and fast (gravity) sand removal (Wang et al. 2020; Hidayaturrahman and Lee 2019; Chen et al. 2018; Bayo et al. 2020; Lares et al. 2018; Murphy et al. 2016). According to a study, microplastics concentration in the WWTP influents was found to be in between 15 and 640 particles L^{-1}, while it was significantly lower in case of the effluent, although varied over four orders of magnitude (Kang et al. 2018). As a result, it is unclear if the discrepancies in microplastic concentrations in wastewater are related to variances in plastic pollution levels or differences in sampling and analytical procedures (Kang et al. 2018).

MPs are currently identified and/or quantified using scientific analytical techniques such as spectroscopy, microscopy, and/or thermal analysis. The most common characterization methodology mentioned in the literature is the use of spectroscopic techniques such as Raman spectroscopy (Peñalver et al. 2020) and Fourier transform infrared (FTIR). MPs have been characterized using scanning electron microscopy based techniques such as, SEM-energy dispersive X-ray spectroscopy(SEM-EDS) and other techniques like Environmental Scanning electron microscopy-EDS (ESEM-EDS) (Rocha-Santos and Duarte 2015). Microplastics thermal analysis is a new technology for MPs characterization. This method is based on identifying the polymer based on the degradation products it produces pyrolysis gas chromatography–mass spectrometry (py-GC-MS), thermogravimetry (TGA), hyphenated TGA such as TGA-differential scanning calorimetry (DSC), TGA–thermal desorption–gas chromatography–mass spectrometry (TGA-TD-GC-MS), TGA–mass spectrometry (TGA-MS), and DSC are some of the other techniques used to characterize (Peñalver et al. 2020).

MPs traversed by the stream eventually enter the sea; hence, WWTPs that discharge their effluents into rivers contribute to ocean pollution. On the other hand, river mouths are the major area for MP contamination (Leslie et al. 2017). To avoid marine MP contamination, it is critical to find effective and environmentally benign methods of removing MPs in WWTPs. Biological methods using bacteria, fungi, and lower eukaryotes have been the focus of most investigations for MPs removal (Masiá et al. 2020). It is still difficult to use living organisms in MPs bioremediation. The key issue with these microscopic creatures is containing them within WWTPs to avoid inadvertent introduction of these organisms in the ecosystem (Nuzzo et al. 2020). Larger organisms, for instance, higher eukaryotes, may be simpler to contain in theory, but their practical use in MPs bioremediation is currently a niche market (Masiá et al. 2020). This chapter examines the sources of microplastics in wastewaters, their properties, ecotoxicity, and health risks, approaches that are already in use and those that are being developed for characterization of microplastics in wastewater, and bioremediation strategies for the removal of microplastics from wastewater for pollution prevention and control.

7.2 Sources of Microplastics in Wastewater

Microplastics are produced from a variety of land-based sources and eventually end up in wastewater treatment plants, which are thought to be the link between contaminants and natural habitats (Rochman et al. 2015). Primary microplastics are those that have been made intentionally, whereas secondary microplastics are those that have been produced by a different type of physical, chemical, or biological degradation (Cooper and Corcoran 2010). Microplastics discovered in wastewater treatment plants primarily consist of fibres and microbeads. Microbeads with a size of 250 μm are found in around 0.5–5% of cosmetics (Bowmer and Kershaw 2010). Exfoliants and toothpaste have been shown to release 4500–95,500 microbeads and 4000 microbeads, respectively, with each use (Carr et al. 2016). Synthetic textile washing, on the other hand, releases around 35% of fibre microplastics into the oceans (Boucher and Friot 2017). A load of roughly 5–6 kg, for example, was found to release 6,000,000–700,000 fibres from polyester and acrylic fabrics, respectively (Boucher and Friot 2017). The number of liberated fibres, however, is dependent on the washing conditions, textile qualities, use, and softener and detergent type (Cesa et al. 2017).

Other domestically produced consumer goods, such as contact lens cleaners and jewellery, have also been found to leak MPs. On the other hand, non-domestic sources, have been reported to leak MPs, including (a) air blasting, (b) transportation and manufacturing, (c) Styrofoam products, (d) textile sector, and (e) dust from the drilling and cutting plastics (Prata 2018). Bayo et al. (2020) recently revealed that seasonal variability is also a significant influence, with the highest amounts of MPs seen during warmer periods, as temperature accelerates plastic degradation and fragmentation. Furthermore, due to urban runoff, large amounts of microplastics have been detected during rainy events (Masiá et al. 2020).

7.3 Properties of Microplastics

Microplastics are a polymer blend that comes in a variety of shapes. Microplastics' form is a key criterion for classification. Microplastics in nine different shapes were identified in the influent and effluent of WWTPs: rod, fragment, film, pellet, foam, ellipse, line, and flake (McCormick et al. 2014). Pellets can be cylindrical, circular, flat, ovoid, and spheruloids, while fragments can be rounded, subrounded, subangular, and angular, to name a few. Microplastics, on the other hand, have uneven, elongated, deteriorated, rough, and broken edges as their most common morphologies. MPs in the environment are shown in terms of their sources, transport, accumulation, and fate in Fig. 7.1. MPs are non-biodegradable, water-insoluble synthetic polymers with a high proclivity for fragmentation and microbial ingestion (Beiras et al. 2018). MPs are bioaccumulated by bacteria, fungi, phytoplankton, and zooplankton in many ways in both terrestrial and marine environments (Paul-Pont et al. 2018). Bioadsorption, biouptake (cellular uptake), and biodegradation are the three main mechanisms through which MPs interact/accumulate in microorganisms (MOs) (Avio et al. 2017). MP bioaccumulation has been shown to alter the growth and metabolism of microorganisms (fungi, bacteria, phytoplankton, and zooplankton) (Xu et al. 2019; Sun et al. 2018).

MP bioaccumulation is a serious concern since if swallowed, it can destroy aquatic life. Because of their minute size, microplastics are easily eaten by different marine organisms (Ferreira et al. 2016). Because microplastics are disseminated at

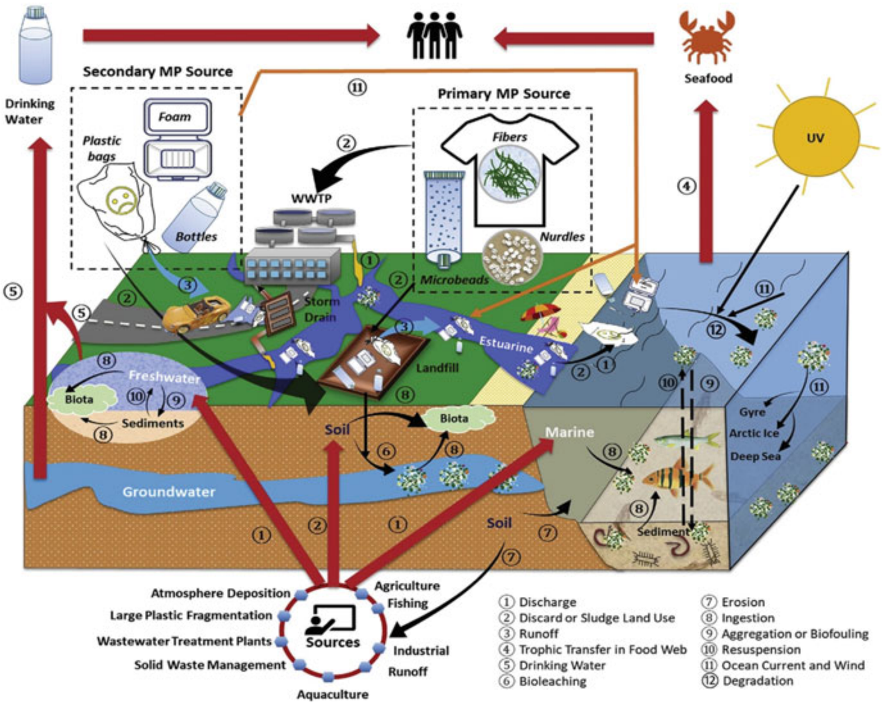

Fig. 7.1 Sources, transport, accumulations, and fate of MPs in the environment (adapted from Wu et al. 2019)

diverse trophic levels, microplastic concentrations in the body may grow as a result of bioaccumulation at higher trophic levels. Microplastics penetrate the food chain and eventually reach humans (Nelms et al. 2018). This shows that the most serious consequences of microplastic poisoning may be experienced by people. There is currently minimal knowledge about the effects of microplastics on food webs, and no laboratory trials on bioaccumulation toxicity induced by microplastics at higher trophic levels have been conducted (Anagnosti et al. 2021). As a result, whether or if any size of plastic may be transmitted to higher trophic levels is unknown. Many persistent organic pollutants (POPs), such as dioxins, polybrominated diphenyl ethers and PCBs have been well-documented occurrences of trophic transfer within marine food webs (Ogata et al. 2009; Hu et al. 2005).

Biological availability refers to the small percentage of the total number of particles/chemicals in the environment that are accessible for absorption by an organism. Because smaller particles have a larger volume ratio, stronger penetrating power, and greater ability to be taken up by marine species, MP bioavailability is known to improve as particle size decreases (Botterell et al. 2019). Microplastics density in the water column may alter their bioavailability. Low-density plastics like PE on the sea surface, for example, are likely to come into touch with filter planktivores, feeders, and suspension feeders in the upper water column (Kooi et al. 2017). Other factors influencing microplastic bioavailability in aquatic habitats include colour, shape, ageing, and abundance (Wright et al. 2013; Crawford and Quinn 2017). The binding affinity of MPs particles with other pollutants has an impact on their bioavailability (Bhagat et al. 2020).

Bioaccessibility and bioavailability are critical principles for calculating the risks of exposure to environmental pollutants. The bioavailability of MPs affects their overall effects on organisms (Cole and Galloway 2015). MP bioavailability to be directly absorbed by a wide spectrum of species is enhanced by their small size (Law and Thompson 2014). A planktivore may confuse MPs for natural food during normal eating behaviour, since their size % is comparable to that of planktonic organisms and sediments (Wright et al. 2013). Scherer et al. (2017) discovered that *C. riparius* can uptake 90 μm MP particles is much lower than that of 10 μm MP particles, despite intraspecific variability in feeding rates *($p < 0.01$)*. As a result, it was found that as MP size drops, their potential bioavailability in the food chain increases.

Microplastics can function as vectors for harmful chemical pollutants, and because they are most exposed in the marine ecosystem, many marine species inadvertently consume them (Fred-Ahmadu et al. 2020). PAHs, for example, have high partition coefficients when it comes to plastics, indicating that they have a significant affinity for polymers (Fred-Ahmadu et al. 2020). Because some of the most often observed environmental plastics have a lesser density than seawater (density 1.02 g/cm^3), they float in water bodies' surface microlayers and may sorb contaminants (SML) (PerkinElmer 2019; Sundt et al. 2014). The contaminant-laden plastics floating in the water can be eaten by marine creatures and seabirds in the epipelagic zone. Even when additive effects are taken into account, polymers like PS, PVC, and PU, as well as plastics with fouling surfaces, have a higher density than seawater or freshwater. The process of "microbial fouling" aids the adsorption of various contaminants onto the surface of microplastics in confined lakes (Neto et al. 2019). As a result, contaminant-sorbed microplastics fall to the bottom of the ocean, where they are available for ingestion by benthic creatures (Teuten et al. 2007).

Chemical contaminants that have been absorbed by microplastics can desorb and biomagnify their way up the food chain, from lower trophic species to fish (Bakir et al. 2014; Rochman et al. 2013). Sorbed pollutants on microplastic particles are easily leached by digestive juices (Voparil and Mayer 2000). MPs that have been ingested for a longer period of time are more effective. remain in an organism's intestines, the more likely pollutants may translocate into bodily tissue. Polybrominated diphenyl ethers (PBDEs) smeared on microplastics were discovered to be incorporated in the tissue of marine amphipods (Chua et al. 2014). As a result of being near to the sources and consumption of these chemicals, the adsorption of various POPs such as polychlorinated biphenyls, hexachlorobenzenes (HCBs), heavy metals and PBDEs to Hydrophobic plastic particles with a large surface area to volume ratio is more prone in freshwater ecosystem than in marine ecosystem (Dris et al. 2015). Freshwater organisms may thus be exposed to increased levels of contaminants, particularly in areas near industrial and populated areas, where increased level concentrations of hydrophobic pollutants, as well as a higher presence of microplastics, may exist, and in areas near agricultural areas, where both POPs (i.e. pesticides) and plastic products are used.

7.4 Ecotoxicity and Health Hazards of Microplastics

Marine animals such as zooplankton (Desforges et al. 2015), mussels (Qu et al. 2018), oysters (Leslie et al. 2017), corals (Hall et al. 2015), and microplastics in the environment may be consumed by fish (Collard et al. 2015). Once swallowed by marine species, microplastics constitute a threat to them. Health hazards posed by MPs to aquatic biota are presented in Table 7.1. Physical, chemical, ecological, and biological impacts are all possible ways of causing dangers (Provencher et al. 2018). Microplastics cause mechanical damage to organisms. Microplastics, for example, have the capacity to block the intestines and cause harm to the gut (through villi cracking and enterocyte splitting), and even affect organism filtering activity and phagocytosis, resulting in organism death (Canesi et al. 2015; Lei et al. 2018). Furthermore, MPs could build up in food web as a result of predation. Microplastics, for example, were discovered to be fed through the pelagic food web by Satlewal et al. (2008), from zooplankton to mysid shrimps. MPs were also observed to move from algae to zooplankton to goldfish, according to Cedervall et al. (2012).

Microplastics would sorb and collect contaminants in aquatic environments chemically. As a result, ingesting polluted microplastics may expose marine species and even the food chain to hazardous contaminants (Santana et al. 2017; Brennecke et al. 2016). In this case, microplastics act as conduits for hazardous pollutants (Carbery et al. 2018). However, little evidence of the influence of trophic transfer of microplastics and pollutants from the food chain on human health exists at this time, necessitating additional investigation. According to Koelmans et al. (2016), the proportion of total hydrophobic organic contaminants (HOCs) deposited on microplastics was modest in contrast to other media in marine ecosystems, and ingestion of microplastics by marine animals may not provide a HOC risk. According to Wang and Wang, PHE sorption capabilities on PE, PS, and PVC microplastics were higher than sorption capacities on sediment samples (Wang and Wang 2018).

Microplastics can also serve as a microbe's artificial substrate in addition to serving as carriers of linked chemical burdens to aquatic species. This has sparked concerns about the biological consequences for freshwater ecosystems as they provide important advantages and services, including as habitat for a diverse range of native plants and animals, drinking water, and recreational opportunities (Meng et al. 2020). In terms of ecology, this might have a significant influence on how microplastics interact with freshwater biotas, such as colonized creatures floating over greater distances and microplastics becoming vectors for poisonous bacteria/algae, diseases, and even invading species. The taxonomic composition of bacterial assemblages colonising microplastics in a heavily urbanized river in Chicago,

Table 7.1 Studies on toxic effects of microplastics on aquatic biota

Species	Class	MP type	Size (μm)	Concentration	Exposure	Effects	Reference
Dunaliella tertiolecta	Microalgae	PS	0.05–6	25 and 250 mg/L	72 h	Algae growth is significantly slowed, but photosynthesis is unaffected	Sjollema et al. (2016)
Skeletonema costatum	Microalgae	PVC	1	0–50 mg/L	4 days	Algal growth inhibition; chlorophyll concentration and photosynthesis	Zhang et al. (2017)
Chlorella pyrenoidosa	Microalgae	PS	0.1 and 1	0–100 mg/L	30 days	Unclear pyrenoids, deformed thylakoids, reduced photosynthetic activity and the suppression of algal development from the lag to the early logarithmic stages results in damaged cell membrane	Mao et al. (2018)
Scenedesmus obliquus	Microalgae	PS	~0.07	44–1100 mg/L	72	Algal growth inhibition; Chlorophyll-a content reduction	Besseling et al. (2014)
Pseudokirchneriella subcapitata	Microalgae	PS	0.055 and 0.1	0.1–1 mg/L	72	Algae growth is slowed	Casado et al. (2013)
Centropages typicus	Copepod	PS	7.3	4000–25,000 beads/mL	24 h	Algal intake is significantly reduced	Cole et al. (2013)
Nephrops norvegicus	Crustacea	PP	3000–5000	5 fibres per 1.5 g feed	>8 months	The rate of feeding, body mass, metabolic rate, and fat catabolism all decrease	Welden and Cowie (2016)
Tetraselmis chuii	Microalgae	PE	1–5	0.046–1.472 mg/L	96 h	Microalgae population expansion is being slowed	Davarpanah and Guilhermino (2015)
Daphnia magna	Crustacea	PS	0.05–10	5 mg/L	14 days	The bioaccumulation of phenanthrene was enhanced by 0.05-m PS, whereas 10-m PS had no impact	Ma et al. (2016)

Arenicola marina	Annelida	PS	400–1300	0–100 g/L sediment	28 days	Body weight, feeding activity, and energy efficiency are all reduced	Besseling et al. (2013)
Perna viridis	Mollusca	PVC	1–50	21.6–2160 mg/L	2 h/day for a period of 91 days	Filtration behaviour, respiration rate, and byssus production all decreased significantly With increased pollution, survival time has decreased	Rist et al. (2016)

PP polypropylene, *PS* polystyrene, *PVC* polyvinylchloride, *PE* polyethylene, *PA* polyamide, *HDPE* high-density polyethylene

Illinois, differed significantly from those colonising suspended organic matter and water column, and that various taxon, such as pathogens and plastic-decomposing organisms, were more abundant on microplastics, according to McCormick et al. (2016). Several research works have looked into the impact of MPs on marine animal reproduction in the ecosystem (Sharifinia et al. 2020). Sussarellu et al. (2016) describe an emerging perspective that MPs reduce reproductive output by altering organism food consumption and energy allocation. According to Lei et al. (2018), MP particles from various sources, such as (Polyamide, Polyethylene, Polypropylene, Polyvinyl Chloride) PA, PE, PP, and PVC, considerably lower reproductive success in the nematode *Caenorhabditis elegans*, although only PE- and PVC-MPs had a significant impact on brood size.

Microplastics are biologically sensitive to colonization by microorganisms. Microplastics may influence microbial community evolution and gene exchange (such as antibiotic resistance genes and metal resistance genes) among bacteria (Yang et al. 2019). The antibiotic resistance gene profile is determined by the microbial community composition, according to Yang et al. (2019). Freshwater invertebrates, water fleas (*Daphnia magna*), and various fish species all actively feed on microplastic particles (1–100 μm), according to laboratory research, and MP particle intake causes critical immobilization of these animals (Besseling et al. 2019; Oliveira et al. 2013; Rehse et al. 2016), as well as affecting predator-prey relationships (Besseling et al. 2019; Oliveira et al. 2013; Rehse et al. 2016; Rochman et al. 2017). However, several studies have revealed that MPs have no effect on ecosystem processes (Krause et al. 2020), making predictions about ecosystem-level consequences more difficult. Furthermore, intergenerational effects on *Daphnia magna* revealed no impacts in the first generation, while neonates exposed to the same concentration of MPs were extinct after two generations (Martins and Guilhermino 2018). Many freshwater benthic consumers (e.g. Oligochaeta worms, Chironomidae larvae, gammaridae, and amphipods) act as ecosystem engineers in sediments and are heavily exposed to MPs, chemical additives, sorbed pollutants, and possible microbial diseases (Frère et al. 2018; McCormick et al. 2014), posing a serious risk of broad range of impacts, particularly on benthos (Izvekova and Ivova-Katchanova 1972; Ward and Ricciardi 2007). For example, lugworms (*Arenicola marina*) that ate MPs had less bioturbation, which decreased the primary productivity of bioturbated substrate and changed lugworm respiration (Wright et al. 2013; Green et al. 2016). PVC microplastics were found in the diet of African freshwater catfish in a new study (Iheanacho and Odo 2020a, 2020b). In this case, the microplastic caused neurotoxicity, oxidative stress, and lipid peroxidation, all of which had an impact on the fish's physiological status. The majority of Microplastics are found in waterways surrounding big cities, particularly in poorer nations with inadequate waste management systems (Xu et al. 2020).

MPs have been found in a variety of places where, all kinds of marine life exists ranging from microscopic species (such as phytoplankton and zooplankton) to enormous predators (mammals and fish) (Anagnosti et al. 2021; Wang et al. 2020). Microplastics have been demonstrated to alter the reproduction, mortality, development, cellular response, behaviour, life span, assimilation efficiency, regeneration, oxygen consumption, egestion, metabolism, nutrition, neurotoxicity, carcinogenicity, and gene expression of aquatic creatures (Haegerbaeumer et al. 2019; Xu et al. 2020). Ingestion of microplastics has a direct impact on small creatures at the bottom of the food chain, producing malnutrition and the inability to eliminate microplastics causes mechanical stress called saturation (Cole et al. 2013; Wright 2015). Microplastic absorption by phytoplankton has been shown to disrupt photosynthesis and, as a result, organism growth (Kalčíková et al. 2017; Bhattacharya et al. 2010). MPs long-term exposure caused considerable alterations in energy stores in two sediment-dwelling bivalve species, *Abra nitida* and *Ennucula tenuis*, but did not affect burrowing activity survival, condition index (Bour et al. 2018). The size as well as number of particles were connected to the outcomes, with larger particles and higher concentrations having more severe consequences. At greater concentrations, microplastics caused oxidative stress; damage to the gut, liver, and gill tissues; increased heart rate; and impeded development and motility in goldfish larvae, resulting in oxidative stress; damage to the intestine, liver, and gill tissues; and increased heart rate (Yang et al. 2019).

Because agricultural plastic film and plastic particles are used extensively in industrial production, MPs pollution on land could be more problematic than in the marine environment (Ramos et al. 2015). MPs also represent a threat to terrestrial creatures, as well as human health, through the food chain and other channels (Sharma and Chatterjee 2017). Plastic films or irrigation water containing MPs can both introduce MPs into the soil system (Rillig 2012). MPs have been found in some studies to have an impact on soil organisms, such as altering the isotopic composition of soil collembolans and perturbing the microbiome (Zhu et al. 2018). Earthworms in the soil can be harmed by polystyrene MPs, which can even kill them (Cao et al. 2017). These findings imply that MPs pollution in soils is harmful to soil organisms and that MPs pose an ecological danger in terrestrial ecosystems. In mice, polystyrene MPs were found to cause dysbiosis of the gut microbiota, intestinal barrier failure, and metabolic problems, according to a study (Jin et al. 2019; Lu et al. 2018). MPs could be consumed by micro- and mesofaunas such as mites, collembola, and enchytraeids, accumulating in the soil detrital food web (Rillig 2012). After seeding and planting *Lolium perenne* (perennial ryegrass) in soils containing MP-clothing fibres, shoot lengths, dry root biomass, dry root–shoot ratio, and chlorophyll a–b ratio high-density polyethylene (HDPE), biodegradable polylactic acid (PLA), and all altered dramatically hence concluded, In the presence of MP-clothing fibres or PLA, seed germination was lower than in control soil (Boots et al. 2019).

Water and nutrient absorption by plant roots is also hampered by the presence of MPs. Plant biomass, root characteristics, tissue elemental composition and soil microbial activity have all been shown to be strongly affected by soil MPs, according to current research (de Souza Machado et al. 2018). Humans consume a wide range of plant and animal products that may include MPs, posing a variety of health hazards. Microplastics are mostly absorbed by ingestion, inhalation, and skin contact (Prata et al. 2020). Microplastics have been detected in beer (Liebezeit and Liebezeit 2014), seafood (Smith et al. 2018), honey and sugar (Liebezeit and Liebezeit 2013), sea salt (Kim et al. 2018), and drinking water (Mintenig et al. 2019). On average, humans consume 4000 MPs each year from water to drink, 37–1000 microplastics from edible sea salt (Van Cauwenberghe and Janssen 2014; Kosuth et al. 2017) and 11,000 microplastics from shellfish. Microplastics (specifically nanoplastics) might reach agricultural fruits/seeds and consequently goes inside the human body through food consumption, according to studies by Sun et al. (2020). Eventually, plant uptake of microplastics may impact human health as well as food security and safety. MPs can also be inhaled through the respiratory system. Airborne microplastics are the most common cause of respiratory exposure. According to a study (Vianello et al. 2019), humans can absorb up to 272 particles each day from indoor air.

The length of time inhaled airborne microplastics travel through the lungs is determined by their size (Enyoh et al. 2019). MPs with a diameter of less than 2.5 m will settle in the lungs first, allowing them to penetrate past the respiratory barrier. Inhalation for a long time Low-level exposure to tiny particles can potentially result in gene mutations (Kingsley et al. 2017). After 10–20 years of being exposed to polypropylene fibres, synthetic textile workers had a higher cancer incidence rate. Workers who worked with polyvinyl chloride had a higher risk of lung cancer as they got older, worked more years, and spent more time in the factories (Prata et al. 2020).

Another form of exposure is dermal touch, but this is a less important route (Prata et al. 2020). Because only particles smaller than 100 nm can be absorbed directly via the skin due to stratum corneum penetration, most microplastics are difficult to absorb (Revel et al. 2018). Microplastics are resistant to chemical breakdown in vivo when they reach the body (Wang et al. 2020). The inhibition of acetylcholinesterase by microplastics could also lead to neurotoxicity (Jeong and Choi 2019). According to a simulated digestion research, microplastics might affect lipid digestion after being consumed by humans by forming microplastics-oil droplet heteroaggregates and inhibiting digestive enzyme activity (Tan et al. 2020), providing a threat to human digestion health.

Microplastics can also be absorbed by human tissues via endocytosis (gastrointestinal tract and airway surface) and paracellular persorption, which is influenced by surface charge, microplastic size, surface functionalization, generated protein corona, and hydrophobicity, among other factors (Wright and Kelly 2017). Increased permeability of the gastrointestinal mucosa can be caused by malnutrition and diets containing high-fructose carbohydrates (due to alterations in the flora of the intestine) and high saturated fats (West-Eberhard 2019). Inhalation and ingestion of MPs in rats resulted in microplastics being discovered in the circulation as well as liver and spleen are examples of distant tissues (Eyles et al. 1995; Jani et al. 1990). A placental perfusion model in humans indicated that 240 nm particle size can cross the placental barrier (Wick et al. 2010). Damage to the DNA replication and repair machinery, as well as DNA damage caused by ROS or particle translocation into the nucleus, can all contribute to MP particle genotoxicity (Rubio et al. 2020).

Microplastics disrupts nuclear membranes, causes oxidative stress, produces damage-related molecular patterns, and activates downstream inflammatory and apoptotic/necrotic pathways in mammalian cells (Yong et al. 2020; Hwang et al. 2020). According to relevant animal model research, these MPs can be transported from living cells to the circulatory systems and lymphatic, where they can gather and harm the cells and immunity of humans (Brown et al. 2001; Browne et al. 2008). Tissue distribution in mice demonstrated MPs accumulation in the kidney, stomach, liver and also the symptoms of energy balance disruption, oxidative stress, and neurotoxicity, after oral administration of fluorescent 5 and 20 m particle sizes at 106 and 104 mg/mL, respectively (Deng and Zhang 2019). After exposure to particulate matter, in vivo neurotoxicity has been reported, possibly due to oxidative stress and activation of the brain's microglia (immune cells) from direct contact with translocated particles or the action of circulating pro-inflammatory cytokines (from other inflammation sites), resulting in neuron damage (Mohan Kumar et al. 2008).

Several studies have linked microplastics to abnormalities in energy homeostasis. Microplastics, for example, may decrease energy intake (a) by causing a decrease in feeding activity (e.g. in crabs, marine worms, and clams) (Xu et al. 2017; Watts et al. 2015); (b) due to decreased predatory performance (e.g. in fishes) (Wen et al. 2018); and (c) due to alterations in digestive enzymes, thereby causing a loss in digestive capacity (Wen et al. 2018).

Additives and monomers from the microplastics matrix may seep into the body, leading to exposure of tissues to endocrine disruptors including phthalates and bisphenol A, which interfere with endogenous hormones even in minute amounts (Cole and Galloway 2015). Changes in the gut microbiome could have negative consequences, such as the spread of dangerous bacteria, a rise in endotoxemia and intestinal permeability (West-Eberhard 2019). Human's inhale, ingest, and eat microplastics through the air (Gasperi et al. 2018), bottled water (Zuccarello et al. 2019), seafood and table salt (Nelms et al. 2018; Zuccarello et al. 2019). Recent studies have shown microplastics in excreta of humans (Yong et al. 2020), indicating that microplastics have been eaten. Plastic toxins were found in every human tissue analysed from Alzheimer's patients in a recent study, which linked toxicity and neurological impairment to lifelong exposure to microplastics (Manivannan et al. 2019).

7.5 Factors Affecting Toxicity of Microplastics

Plastic toxicity varies depending on the polymer type. Polyurethane, PVC, polyacrylonitriles, styrene-based copolymers and epoxy resins, categorized as the most dangerous (category 1A or 1B mutagen or carcinogen) because of the hazard division of monomers (Lithner et al. 2011). It is crucial to keep in mind that the higher toxicity of smaller particles is not always apparent, since it depends on a variety of elements such as exposure time, charge, cell type, dose, and polymer type. Larger particles necessitate the use of specialist cells to phagocytose them (Alberts et al. 2002). Endocytic and passive uptake mechanisms can take up smaller particles. Particle size and toxicity are usually inversely proportional. The toxicity of 500 nm PS particles (IC50 12.6 g/mL) was found to be higher than that of 50 nm (IC50 $>$

100 g/mL) in NIH/3T3 and ES-D3 mouse embryo cultures, for example (Hesler et al. 2019). Because of their small size and high surface–volume ratio, MP/NP can absorb additional contaminants such as heavy metals, persistent organic pollutants (POPs), and viruses (de Souza Machado et al. 2018; Yu et al. 2019). Plastics can have persistent organic pollutants (polycyclic aromatic hydrocarbons, polychlorinated biphenyls, DDT), heavy metals (Cd, Cr, As, Hg, As, Br, Zn, Cu, Sb, Sn, Ti, Mn, Co, Ba), and pathogenic *Vibrio* spp. (Campanale et al. 2020; Brennecke et al. 2016; Prinz and Korez 2020; Kirstein et al. 2016; Velzeboer et al. 2014; Rodrigues et al. 2019).

The absorption, transport, and toxicity of particles can all be influenced by the surface charge of MPs (Yacobi et al. 2010; Fröhlich et al. 2012; Loos et al. 2014a, 2014b). Plasticizers, stabilizers, dyes, lubricants, and flame retardants are among the leachates/plastic additives, which account for around 4% of MPs content and potentially pose health hazards (Bouwmeester et al. 2015; Campanale et al. 2020; EFSA CONTAM Panel 2016). Hahladakis et al. (2018) show that the existence and release of additives, on the other hand, does not always imply a health risk, as toxicity is dictated by the plastic composition and the rate of leachate migration, as well as the amount and solubility of leachate in the surrounding environment. The migration of additives is in large amounts from plastics in fatty foods and when stored at high temperatures or for long periods (Hahladakis et al. 2018).

Chemical adsorption on MPs can be influenced by some circumstances. MPs type, size, environmental salinity and pH, and plastic ageing are only a few of the variables (Mammo et al. 2020). For the same size (200–250 mm), different kinds of microplastics, such as PP, PVC, polyethylene terephthalate (PET), and PE, have varied surface areas and distribution coefficients (Teuten et al. 2007). At differing pH levels, the sort of charge on microplastics surface and chemicals influences whether adsorption increases or decreases. Adsorption is enhanced when the MP surface and chemicals have opposite charges, but adsorption is reduced when the MP surface and chemicals have identical charges (Karlsson et al. 2017). According to Seidensticker et al. (2018), due to repulsion between comparably charged polar compounds and plastic surfaces, non-polar molecules have stronger sorption on PE and PS than polar compounds. The influence of salinity on chemical sorption on MPs can be assessed using changes in the partition coefficients of a chemical with a change in salinity. Log KMP-SW in saltwater and log KMP-W in the same chemical water are different, according to Wang et al. (2020), suggesting that salinity impacts chemical sorption on MPs.

Weathering or ageing of MPs has been reported to increase the rate of chemical sorption (Endo et al. 2005; Rios et al. 2007). Due to environmental interactions such as long-term exposure to the sun, which can cause photo-oxidation, aged plastics have rough surfaces. This causes plastics to degrade into smaller sizes, increasing their surface area and sorption capacity (Brennecke et al. 2016). Adsorption is linked to several sorption sites that are dependent on crystallinity (Joshi et al. 2017). Higher crystallinity produces a clean surface with fewer sorption sites, lowering adsorption. Previous research has found that the crystallinity of MPs influences HOC partitioning, which affects adsorption (Guo et al. 2012). Guo et al. (2012) found that lowering the crystallinity of PE from 59 to 26% increased the sorption of phenanthrene, naphthalene, and lindane. Liu et al. (2019) studied the differences in ciprofloxacin sorption on PVC (low crystalline) and PS (high crystalline) and found that ciprofloxacin sorption on PS was lower than that on PVC.

7.6 Techniques for Characterization of Microplastics in Wastewater

The sample of wastewater is the initial step in its characterization, and it has a direct impact on the MPs study's outcomes. Filtration collection is the most popular method for taking samples from wastewater in WWTPs (Kang et al. 2020). Input and effluent samples, on the other hand, can provide information on microplastics sources, total MP removal rate, and pollutant loading. After sampling, the sample must be predigested to remove contaminants and increase extraction efficiency, as well as to minimize MP loss and damage to the greatest extent possible. Purification is another crucial stage in MPs characterization since it demands the removal of the largest amount of organic materials while causing the least amount of damage to MPs. The digestive regents utilized and their concentration, as well as reaction variables such as temperature and duration, might affect the purifying effect (Kang et al. 2020) MPs must be removed from pre-treatment samples after purification to be detected and analysed. Flotation (Imhof et al. 2013) and Elutriation (based on an upward gas or liquid flow to separate MPs) (Mahon et al. 2017) are two unique procedures that have been invented but not generally implemented. Density separation and filtration are two popular ways of extracting MPs (Kang et al. 2020).

Microplastics analysis can be bifurcated into two categories: chemical characterization and physical characterization. Chemical characterization, on the other hand, is primarily used to investigate the composition of microplastics. Several analytical techniques are currently being used to characterize microplastics, including polarized light optical microscopy (PLOM) (Sharifinia et al. 2020), energy-dispersive X-ray spectroscopy (EDS) (Li et al. 2018; Mahon et al. 2017), Raman spectroscopy, and Fourier transform infrared spectroscopy (FTIR) (Sun et al. 2019). The term "physical characterization" refers to the process of determining the distribution of size of microplastics and also other physical characteristics, for instance colour and shape. Several studies performed by authors on the detection and characterization of MPs using different analytical techniques are listed in Table 7.2.

7.7 Bioremediation Strategies for Microplastics

7.7.1 Bacterial Degradation of Microplastics

Numerous investigations on the use of microbes for MP breakdown are now underway. A list of microorganisms reported for the degradation of MPs is presented in Table 7.3. The characteristics features of candidate microbial species for

Table 7.2 Microplastics are detected and characterized using analytical methods

Country	Habitat	Size	Abundance	Sampling	Extraction	Quantification	Reference
UK	Estuarine/ subtidal sediment; beach ($n =$ 17)	>1.6 mm	NA	Strandline, subtidal; $n = 5$ at each location	Separation by density (1.2 kg L^{-1} NaCl); stirring; filtration (Whatman GF/A 1.6 mm)	Polymer identification and visual counting/ sorting (FTIR)	Thompson et al. (2004)
Singapore	Beach ($n = 8$)	>1.6 mm	NA	1 cm surface ($n = 4$); sub-surface (10–11 cm, $n = 4$)	Separation by density (hypersaturated solution, 1.2 kg L^{-1}); filtration (1.6 mm)	Polymer identification (FTIR)	Ng and Obbard (2006)
India	Marine sediment (10 locations)	>1.6 mm	81 mg plastics per kg sediment	Subtidal strandline; $n = 5$ at each site Surface (10–11 cm, $n = 4$); subsurface (10–11 cm, $n = 4$) ten sample sites; a) (between high tide a low water mark) (each site) 10 kg samples; depth (0–5 cm); sieved	Separation by density (1.2 kg L^{-1} NaCl); stirring; filtration (1.6 mm Whatman GF/A) Separation by density (hypersaturated solution, 1.2 kg L^{-1}); filtration (1.6 mm) Separation by density (30% NaCl); filtered (1.6 mm)	Polymer identification and visual counting/ sorting (FTIR) Identification of Polymers (FTIR) Counted and sorted visually (optical microscope); polymer identification (FTIR); surface characterization (SEM)	Reddy et al. (2006)
England	Estuary (3 locations, 2 sites)	<1000 mm, 1–10 mm	NA	Strandline; randomly placed quadrates (5 replicates 50 cm 50 cm); depth (0–3 cm)	Density separation (saturated NaCl)	Polymer identification (FTIR)	Browne et al. (2010)

Brazil	Beach (1 location)	>500–1000 mm	NA	Strandline; 100 m transect; depth (0–2 cm); 9 replicates; quadrat (988 cm); wire cloths field sieving (0.5, 1 mm)	Density separation (filtered seawater)	Visually counted/ sorted	Costa et al. (2010)
Belgium	Marine sediment (three harbours, 3–4 sites at each)	>63 mm	170 (49–390) pieces per kg	3 sampling points (strandline, intertidal, subtidal zone; parallel); sediment cores (2–7 cm)	Density separation, 1 kg sediment, 3 L conc saline solution; stirred; settle 1 h; sieved (38 mm)	Visually counted/ sorted (binocular microscope); polymer identification (FTIR)	Claessens et al. (2011)
Portugal	Beach (5 locations)	>1 mm	190 (29–393) pieces per m^2	Strandline; two quadrats (50 × 50 cm; 2 × 2 m) in duplicate; depth (0–2 cm); samples sieved in situ (2.5–3.5 mm)	Density separation, concentrated NaCl 140 g L^{-1}, stirred vigorously; filtered (Whatman GF/C 1 mm)	Polymer identification (m-FTIR)	Martins and Sobral (2011)
Canada	Freshwater sediment	<5 mm	NA	Parallel to shoreline; 60 m transect; visible debris sampled (within 1 m); two replicates	Separation into three categories (<5 mm, >5 mm and PS); ultrasonicated in DI water for 4 min	Polymer identification (FTIR); surface characterization (SEM)	Zbyszewski and Corcoran (2011)
Germany	Beach ($n = 2$)	>0.45 mm	NA	2 random samples; depth (0–2 cm)	Density separation, NaCl (1.2 g cm^3); manual shaking; filtered (0.45 mm nitrocellulose)	Pyr-GC combination with mass spectrometry (MS), Stereomicroscope; SEM	Fries et al. (2013)
South Korea	Beach ($n =$ 10)	>2 mm	980 pieces per m^2	Strandline line; 50 m transect; 10 replicates; quadrat (50 cm); depth (0–5 cm)	Field sieved (2 mm)	Visual identified and sorting	Heo et al. (2013)

(continued)

Table 7.2 (continued)

Country	Habitat	Size	Abundance	Sampling	Extraction	Quantification	Reference
Canada	Freshwater lake (3 lakes; 26 locations)	<10 cm	NA	Parallel to shoreline; 60 m transect; 10 m interval; visible <10 cm particles collected	Cleaned (ultrasonic bath, DI water); separated by hand into four categories	Surface characterization (SEM); visual counting/sorting; polymer identification (FTIR)	Zbyszewski et al. (2014)
Hong Kong	Beach (n = 25)	315–5000 mm	5595 (106–15,554) pieces per m^2	Tide line; parallel; 30 m transect; depth (0–4 cm); quadrat 50 cm	In field sieve (315 mm); density separation (tap water, sonication); wet sieved (315 mm)	Visually counted and sorted (microscope); gravimetrically	Fok and Cheung (2015)
South Africa	Beach sediment (21 sites)	65–5000 mm	670 particles per m^2	Strandline; triplicate composite samples; depth (0–5 cm); 1200 mL subsample taken	Density separation; saturated saline solution; repeated five times	visual sorting (fragments and fibre, colour; visually counted (dissecting microscope)	Nel and Froneman (2015)

Table 7.3 Microorganisms reported for the degradation of microplastics

Microbes	Habitat	MP type	Exposure and degradation	Reference
Bacillus	Mangrove sediment	PP	40 (d) and 4.0% weight loss	Auta et al. (2018)
Rhodococcus	Mangrove sediment	PP	40 (d) and 6.4% weight loss	Auta et al. (2018)
Bacillus gottheilii	Mangrove ecosystems	PE, PET, PP, PS	40 (d) and 6.2, 3.0, 3.6, 5.8% weight loss	Auta et al. (2018)
Enterobacter asburiae	Plastic-eating waxworms	PE	28 (d) and 6.1 ± 0.3% weight loss	Yang et al. (2014)
Bacillus	Plastic-eating waxworms	PE	28 (d) and 10.7 ± 0.2% weight loss	Yang et al. (2014)
Aspergillus tubingensis	Marine coastal area	HDPE	30 (d) and weight loss; VRKPT1 was found to be effective at degradation of HDPE; virgin polyethylene was used as a carbon source	Devi et al. (2015)
Aspergillus flavus	Marine coastal area	HDPE	40 (d) and a weight reduction of 4.0% 40 (d) and a weight reduction of 6.4% 40 (d) and a weight decrease of 6.2, 3.0, 3.6, and 5.8% 28 (d) and a weight reduction of 6.1, 0.3% VRKPT1 was found to be successful at degrading HDPE; virgin polyethylene was utilized as a carbon source. 30 (d) and weight loss; VRKPT2 was found to be effective at degrading HDPE; virgin polyethylene was used as a carbon source	Devi et al. (2015)
Penicillium simplicissimum	Soil and leaves	PE irradiated for 500 h with UV light	Polyethylene with molecular weights ranging from 4000 to 28,000 showed after 3 months, the molecular weights of polyethylene were reduced, and functional groups added to the polyethylene assisted biodegradation	Yamada-Onodera et al. (2001)
Penicillium pinophilum	Purchased from a strain centre	LDPE powder	The biologically treated MPs exhibited substantial morphological and structural changes after 31 months, including 0.37% mineralization	Volke-Sepúlveda et al. (2002)

(continued)

Table 7.3 (continued)

Microbes	Habitat	MP type	Exposure and degradation	Reference
Zalerion maritimum	Marine	PE pellets	28 days and molecular modifications; Z. maritimum was able to use PE, resulting in a reduction in pellet bulk and size	Paço et al. (2017)
Bacillus sp. and *Paenibacillus* sp.	Municipal landfill sediment	PP	60 (d) and 14.7% weight loss	Park and Kim (2019)
Bacillus cereus, *Arthrobacter* and *Bacillus pumilus*	Soil beds	HDPE/ LDPE	14 (d) and 21.7–22.41% weight loss	Satlewal et al. (2008)
Exiguobacterium	Plastic-eating mealworms	PS	28 (d) and 7.4 ± 0.4% weight loss	Yang et al. (2015)
Bacillus sphaericus and *Bacillus cereus*	Marine	Nylon 66 and nylon 6	3 months and 2–7% weight loss	Sudhakar et al. (2008)

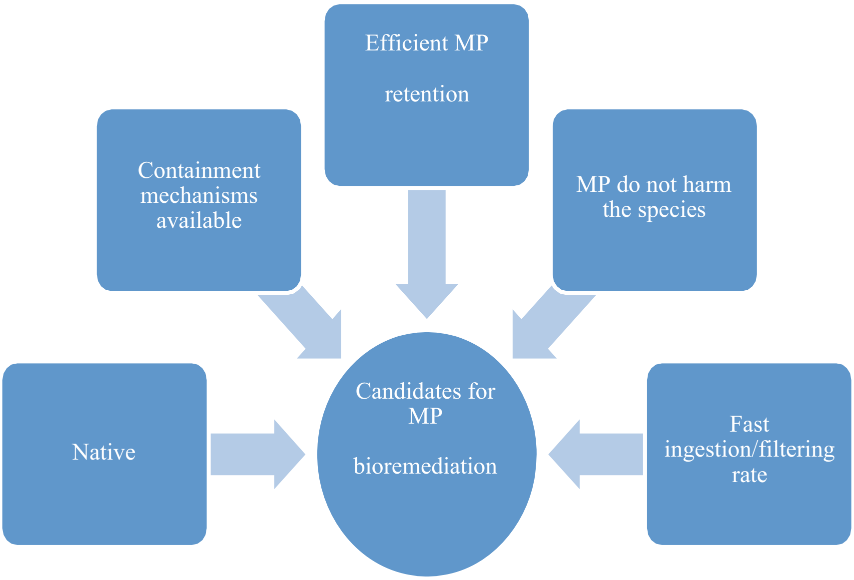

Fig. 7.2 Characteristics features of candidate microbial species for bioremediation of MPs in WWTPs (adapted from Masiá et al. 2020)

bioremediation of MPs are depicted in Fig. 7.2. Pure bacterial cultures have been employed in studies on the breakdown of Microplastics by microbes in the laboratory (Yuan et al. 2020). Enrichment culturing is usually used to isolate microbial cultures from silt, sludge, and wastewater. Pure strains have the benefit of being a simple approach for examining metabolic pathways or assessing the influence of various environmental factors on MP degradation in MP degradation studies.

Furthermore, the whole MP breakdown process, as well as variations in MPs, can be precisely tracked by functional bacteria (Janssen et al. 2002). Auta et al. (2018) identified two pure bacterial cultures from mangrove silt and utilized them to break down PP MPs. The weight loss of PP MPs induced by *Bacillus* sp. strain 27 and *Rhodococcus* sp. strain In MP degradation investigations, metabolic pathways or analysing the effect of various environmental conditions on MP degradation are both important. 36 was 4.0 and 6.4%, respectively, after 40 days of incubation. Following considerable study, it was revealed that the bacterium is responsible for altering the appearance of microplastics and their functional group structures and other features (Auta et al. 2018). To summarize, future studies are required to optimize techniques and enhance bacteria strains to increase their speed arbitrate the degradation process and increase the pace of MP breakdown (Yuan et al. 2020).

7.7.2 Fungal Degradation of Microplastics

Along with bacteria, fungi can get attached with and use Microplastics (Mitik-Dineva et al. 2009). Fungi can make MPs less hydrophobic by increasing the formation of chemical bonds like carboxyl, carbonyl, and ester functional groups. Until recently, however, there was little research on the fungal-arbitrated elimination of MPs in the literature. Using ectopic screening, the problems of obtaining fungal strains with strong MP-degrading activity were demonstrated (Yuan et al. 2020). Research into the breakdown of microplastics by fungi in various environments is still underway, but some progress has been made. *Penicillium simplicissimum* YK was identified by Yamada-Onodera et al. (2001) for application in PE biodegradation. Surprisingly, the aforementioned strain was able to grow better on a solid medium added with 0.5% PE after 500 h of irradiation than on unirradiated media. Dantzler et al. tested two different isolates of *Pestalotiopsis microspora* for their ability to degrade polyurethane (PUR) to determine if the fungus can degrade a range of MPs in another research (Russell et al. 2011). They discovered that a serine hydrolase was revealed to be the reason for the breakdown of PUR, suggesting that fungus-secreted enzymes can help with MP biodegradation. Devi identified two isolated fungus strains capable of degrading HDPE (*Aspergillus tubingensis* VRKPT1 and *Aspergillus flavus* VRKPT2) (Devi et al. 2015). Hydrolyzable polymers can also be degraded by fungi. Deguchi et al. (1997) were the first to report oxidative assault on nylon-6,6 in the white-rot fungus, IZU-154, *Phanerochaete chrysosporium* and *Trametes versicolor*. These findings indicated that these fungal strains have a high ability to break down MPs in vitro. To improve the rate of fungal-mediated MP degradation, future investigations should employ genomes and proteomics approaches.

7.7.3 Microalgal Degradation of Microplastics

The microalgae and plastic interaction waste can substantially alter the properties of these polymers, affecting their fate in aquatic ecosystems (Yokota et al. 2017). Among primary producers, filamentous cyanobacteria of the genus *Phormidium* are known to break down hydrocarbons (Oberbeckmann et al. 2016). It has been well known that species from this genus can be found on plastic surfaces. This raises the intriguing possibility that *Phormidium* is hydrolysing the plastic in the plastisphere actively (Yokota et al. 2017). The advantage of increased sunshine exposure on floating plastic pieces may be the real source of plastic trash enrichment as cyanobacteria are photosynthetic in nature (Roager and Sonnenschein 2019). Microalgae were exposed to high-density polyethylene microplastics at a concentration of 1 g L^{-1} and 400–1000 μm diameter polypropylene (Lagarde et al. 2016); 2 mm polystyrene at 3.96 g L^{-1} (Long et al. 2017), and microbeads from cosmetic products at around 4000 microbeads (Long et al. 2017); and 2 mm polystyrene. Long et al. (2015) explored this interaction in a lab experiment using various aggregates generated from two different algae species (the cryptophyte *Rhodomonas salina*, the diatom *Chaetoceros neogracile*, and a mix of both) and 2 m polystyrene microbeads. The experiment revealed that the microbeads were enriched in all three forms of aggregates. Once absorbed, the microbeads increased aggregate sinking rates to several hundred meters per day, a substantial increase over loose beads' sinking rate (less than 4 mm day^{-1}). These results are evidence to the idea that the aggregates of phytoplankton can act as an MP sink. Furthermore, when an aggregate splits, more surfaces and macropores become available for microbeads to cling to, allowing microbeads to be incorporated not just at the aggregate surface or in macropores, but throughout the aggregate (Long et al. 2015).

7.7.4 Microbial Consortia in Microplastics Degradation

Numerous investigations have revealed that when an axenic bacterium (pure bacterial cultures) biodegrades organic substances, hazardous end products are produced that impede the growth of microbes (Dobretsov et al. 2013). Using a mixture of bacteria to establish an intact community of microbes that have the ability to assist minimization of harmful metabolite effects on microplastics-degrading bacteria when compared to axenic bacterial cultures. In addition to that, a microbe's poisonous metabolites can frequently be used as a growing substrate of different microorganisms. Consortia bacteria have synergistic symbiotic and mutual relationships, which allows them to be more tolerable and active during pollutant treatment (Singh and Wahid 2015). Park and Kim (2019) looked at how a mesophilic mixture of bacteria generated from trash debris broke down PE MPs. Both *Paenibacillus* sp. and *Bacillus* sp. were plentiful in the mixture of bacteria and decreased the dry weight of Microplastics particles (14.7% after 60 days) as well as the mean diameter of Microplastics particle (22.8% after 60 days). Earthworms (*Lumbricus terrestris*) degrade LDPE, according to Huerta Lwanga et al. (2018).

The earthworm's intestinal bacteria absorbed and destroyed LDPE MPs after they were consumed. This shows that MP-degrading microbes are present in the earthworm's digestive system. Lwanga also looked at the role of bacteria in gut of earthworms in the decomposition of microplastics. The most common isolates from the earthworm's gut were members of the genera Firmicutes and Actinobacteria. Researchers have discovered that when isolated strains were tested for microplastics degradation, the particle size of LDPE-MPs was drastically decreased in the presence of bacteria. There are also eicosane, tricosane, and other volatile chemicals present. Only the treatments that included both LDPE-MP and bacteria produced docosane, indicating that these long-chain alkanes were produced as a result of bacterial-mediated LDPE-MPs breakdown. Due to interactions between many microorganisms and various enzymes, the breakdown and usage of microplastics by mixture of bacteria is a baffling process. As a result, it will be critical to in the future to develop in-depth research on the influencing factors (Yuan et al. 2020).

7.7.5 Microbial Biofilm in Microplastics Degradation

MPs are exposed to inorganic particles, organic matter, and microbes in aquatic settings (Parrish and Fahrenfeld 2019). Microorganisms of many forms and sizes, including bacteria, algae, viruses, protists, and fungi can cling at the surfaces of Microplastics as a result (Oberbeckmann et al. 2016). Biofilms, which are complex ecosystems made up primarily of microbes, organic and inorganic particles, cell secretions, and other materials, can form as a result of the colonization of these bacteria (Flemming 1998). Microplastics with rough or smooth surfaces, low or high densities, and a wide range of chemical compositions can be used as a biofilm development substrate. Biofilms change and degrade MPs' physical qualities, according to growing research (Rummel et al. 2017). Lobelle and Cunliffe (2011) studied the development of early biofilms on PE MPs surfaces for three weeks. One week later, biofilms were easily visible on PE surfaces, and they remained intact to proliferate for the next 14 days. The total number of heterotrophic microorganisms on the lowered MPs increased as well, Week three saw an increase in cell count from 1.4×10^4 cells cm^{-2} to 1.2×10^5 cells cm^{-2}. Over the course of 3 weeks, As the contact between seawater and air became more hydrophilic, the MPs began to sink. Under the influence of biofilms, there was some damage and certain changes that occurred to PE MPs. Miao et al. (2019) investigated the microbial community in various substrates using high-throughput sequencing and generated community metrics such as evenness, diversity, and species richness. Natural materials have less bacteria connected to plastic degradation than MPs, according to the researchers. MPs were shown to have higher levels of Phycisphaerales, Pirellulaceae, Roseococcus, and Cyclobacteriaceae than originally occurring substrates, demonstrating that microplastics are microbial specific. Biofilms utilizes microplastics as carbon sources, energy sources, and adhesion media in conjunction with microbes and enzymes, attacking and degrading them. Biofilms' destruction of MPs, on the other hand, is extremely difficult and has yet to be thoroughly researched. MP breakdown products, for example, have not been effectively collected and studied. Furthermore, routine data analysis for weight loss and cosmetic alterations has aided in the present understanding of biofilms' impacts on MPs. More controlled study and product tracking will be carried out in the future to better understand these relationships and degradation behaviour.

7.7.6 Bioreactor Systems for Microplastic Removal from Wastewater

The majority of microplastics were eliminated from the bioreactor system by bacterial consumption and the formation of sludge aggregates. Domesticated activated sludges, in particular, have been reported to increase microplastic build-up in WWTPs. During the subsequent secondary settling operation, microplastic-containing sludge was eliminated (Jeong et al. 2016). In WWTPs, the A2O bioreactor system is the most extensively utilized (Liu et al. 2021). Due to the sludge return, however, it has a low microplastics removal effectiveness. Microplastics that had been absorbed by the sludge would return to the aqueous phase in a small percentage (20%). Furthermore, the breakdown of microplastics in A2O is relatively sluggish (Liu et al. 2021). As a result, the typical activated sludge method of removing microplastics from WWTPs is inefficient.

Membrane bioreactor or MBR technology has lately gained popularity WWTPs treatment method. Because of the high concentration of mixed-liqueur suspended particles, it has an exceptional performance in removing microplastics (removal efficiency of 99.9%) (range from 6000 mg L^{-1} to 10,000 mg L^{-1}) (Dvořák et al. 2013; Talvitie et al. 2017). Membrane separation and the classic activated sludge technique were combined in MBR technology. The bulk of microplastics were maintained on the MBR system's biofilm carrier side. This revealed that the adsorption effect is one of the most important factors in microplastic elimination in the MBR system. The pore size of the membrane employed in MBR systems is typically 0.1 m (Atasoy et al. 2007). Biofilter technology is employed as a major technology following the bioreactor system. MPs with the small size of particles and lesser density are flooded into the biofilter treatment unit. The removal of MPs became more challenging as a result of these factors. Biofilter technology, on the other hand, has the best microplastic removal performance (Lei et al. 2018). The major strategies for removing microplastics are biofilm filtration and adsorption, and biofilter technology combined physical and biological purification processes (Liu et al. 2021). Because microplastics are considered microbe transporters, their presence will have an impact on the microbial activity and community. According to Li et al. (2020), the number of functioning units of taxonomy dropped from 1665 to 1533 after the addition of PVC. As a result, there was an increase in the number of functional units of taxonomy to 1735. As a result, the presence of microplastics PVC did not result in a significant decrease in operational taxonomic units and had no effect on microbial community structure. It is also reassuring to learn that virgin microplastics had no effect on phosphorus-accumulating organisms, nitrite-oxidizing bacteria, or ammonia-oxidizing bacteria (Liu et al. 2019). As a result, the microplastics effect on the functioning of bioreactor system must not be overestimated. On the other hand, the additives toxicity in MPs to microorganisms was unknown. The influence of microbe-containing MPs on traditional pollution remediation should be studied in the future.

7.8 Challenges and Future Perspectives

Multiple knowledge gaps and areas of disagreement must be addressed before the number of MPs in WWTPs can be estimated. To have a better picture of annual MP deposits into WWTPs, temporal and regional trends in MPs must be investigated. An estimate of the annual variance of MPs in wastewater and the ability of WWTPs to handle such flows have yet to be explored, but it is crucial for gaining a better grasp of worldwide MPs wastewater trends. Furthermore, nothing is known about MPs' ability to store and transfer chemical and microbiological contaminants across the landscape (including diseases). Treatment plants contain a wide range of dangerous pollutants and pathogens, but little is known about MPs' ability to adsorb them at all phases of the treatment process and thereafter. Existing studies of microplastics in WWTPs have some flaws that need to be addressed in future research. The establishment of standardized sampling and analysis methodologies to understand the fate of microplastics better in WWTPs or any other media of environment should be the centre of attention of future research. Standardization of reporting units for MP concentration is required. Furthermore, because size of MP particle ranges from 100 nm to 5 mm, mass units are the most accurate depiction of MP contamination within a given sample, allowing for more efficient comparisons between sampling locations. Simultaneously, additional study into specific microplastics should be prioritized, particularly in industrial zones. Hydraulic retention time, salinity, and dissolved organic matter, all of which influence the treatment processes' ability to eliminate microplastics in WWTPs, deserve further investigation. Furthermore, the microplastic removal effect of reaction intermediates, removal of contaminants and their toxicity created by the current treatment technique was unknown. Several MPs-degrading functional microbes have been identified, and several methods for characterizing them have been established. To garner a decrease in MPs, functional microbial agents must be explored and even genetically engineered. The investigation of MP degradation mediated by microbes is a significant undertaking. Greater formation of microbial potential and their use in MP treatment will be required in the future to reduce MP pollution.

On average, a total of 70% of MPs were eliminated during primary treatment. The role of dissolved air flotation (DAF) in the removal of MPs was the most evident at this stage (Bui et al. 2020). The membrane bioreactor (MBR) system is currently the most outstanding treatment technique for secondary treatment, with a success rate of over 99% (Lares et al. 2018). This study indicates that integrating primary treatment units with MBR procedures improves MP removal from wastewaters. One of these technologies' disadvantages is that there are still few studies testing MBRs' efficacy on MPs; the influence of MPs on the cost–benefit analysis and membrane fouling, have gotten little attention. Furthermore, the operating parameters and environmental elements of MBR had a considerable impact on MP removal efficiency. And the research has not focused on these difficulties so far. As a result, it is critical to pay more attention to future investigations on Microplastics removal in various bioreactors, particularly MBRs.

It is also worth noting that the lack of consistency in analytical investigations leads to a wide range of outcomes. Ignorance of small MPs (20 m) and lack of MPs mass estimation are just a few of the significant issues with the analysis approach. Despite purification and clean treatment, it is impossible to eliminate contaminants from MPs samples, resulting in an unspecific spectrum that is difficult to differentiate from the library's standard spectrum. Long-term data is required for the accurate assessment of MPs concentrations in WWTPs throughout the year.

7.9 Conclusion and Recommendations

Despite the fact that WWTPs are not specifically designed to remove MPs, millions of MPs are discharged into the environment every day, both from treated water outflow and sewage sludge used for soil augmentation. As a result, these facilities are seen as a potential source of MPs entering aquatic environments. However, all MP study results are based on laboratory circumstances that can never match the actual environment, hence absolute choices based on laboratory research on how MPs function in an organism's natural habitat are unreliable, and therefore, extensive future research is required. This chapter's conclusions and recommendations are as follows:

a. WWTPs should be prioritized as hotspots for preventing microplastics from entering the environment.
b. More emphasis on improving and implementing advanced tertiary treatment techniques to remove more MPs from treated water is needed.
c. Depending on the species, bioremediation could be a viable option for degrading or accumulating microplastics in wastewater treatment.
d. Research into new methods and biotechnologies for removing MPs from sludges with high efficiency is required.
e. Evaluation of candidate species' ability to retain MPs in realistic environmental concentrations should be carried out.

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