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Challenges and opportunities in sustainable management of microplastics and nanoplastics in the environment

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Highlights

- Micro- and nanoplastics (MNP) pose adverse effects on ecosystem and human health.
- MNP management is vital for achieving United Nations Sustainable Development Goals.
- Innovative approaches of plastic management cut environmental MNP burden.
- Plastic wastes need conversion to value-added products upholding circular economy.
- An environmentally safe limit of MNP need to be established and implemented.

Abstract

The accumulation of microplastics (MPs) and nanoplastics (NPs) in terrestrial and aquatic ecosystems has raised concerns because of their adverse effects on ecosystem functions and human health. Plastic waste management has become a universal problem in recent years. Hence, sustainable plastic waste management techniques are vital for achieving the United Nations Sustainable Development Goals. Although many reviews have focused on the occurrence and impact of micro- and nanoplastics (MNPs), there has been limited focus on the management of MNPs. This review summarizes the ecotoxicological impacts of plastic waste sources and issues related to the sustainable management of MNPs in the environment. Moreover, this review critically evaluates possible approaches for incorporating plastics into the circular economy in order to cope with the problem of plastics. Pollution associated with MNPs can be tackled through source reduction, incorporation of plastics into the circular economy, and suitable waste management. Appropriate infrastructure development, waste valorization, and economically sound plastic waste management techniques and viable alternatives are essential for reducing MNPs in the environment. Policymakers must pay more attention to this critical issue and implement appropriate environmental regulations to achieve environmental sustainability.

Keywords: Micro- and nanoplastics (MNPs); Circular economy; Ecotoxicological effects; Plastic pollution; Sustainable waste management

1. Introduction

Global waste plastic pollution has led to an alarming situation owing to the adverse impacts of widely distributed MNPs in ecosystems (da Costa et al., 2016; Kumar et al., 2020b). The ultra-fine fraction of plastics is categorized as NPs (< 0.2 mm), whereas micro- and macroplastics

are commonly distinguished in size ranges of < 5 mm and > 5 mm, respectively (Blair et al., 2017). Based on the source, MNPs are separated into two main groups: primary and secondary. Primary MNPs, such as microfibers/beads and plastic pellets, evolve owing to direct contamination from the discharge of personal care products, synthetic fabrics, and various industries (Rillig, 2012). These anthropogenic MNPs are commonly found in sludge from wastewater treatment plants (WWTPs) and households and industrial sewage sludge (Mahon et al., 2017). Secondary MNPs result from the degradation of larger plastic particles under natural weathering caused by UV radiation and mechanical forces (Cole et al., 2011; Massos and Turner, 2017). Poor segregation of municipal solid waste (MSW) in the disposal process increases the contamination of soil with plastic waste in landfills and open dumpsites (He et al., 2019). The breakdown of macroplastics can result in the accumulation of MNPs in soil and aquatic environments. MNPs are more challenging to trace, control, and quantify in marine and terrestrial environments than at primary MNP sources (Blair et al., 2017). Chemical additives and MNP-bound organic and inorganic contaminants can potentially accumulate throughout the food chain and transport contaminants acting as vectors (Bradney et al., 2019). Organic and inorganic contaminants, such as persistent organic pollutants and potentially toxic elements adsorbed on MNPs, along with a variety of leached chemical additives from plastics cause significant ecotoxicological issues (Alam et al., 2019; Brennecke et al., 2016; Groh et al., 2019). Polychlorinated biphenyls (PCBs), perfluorinated and polyfluorinated alkyl substances (PFASs), and polycyclic aromatic hydrocarbons (PAHs) are commonly added to plastics during production. Plastics also contain metallic elements such as chromium (Cr), lead (Pb), and arsenic (As) (Alam et al., 2019), which can leach into the environment. Moreover, the small particle size of MNPs enhances their ability to be ingested by aquatic species and soil organisms (Wright et al., 2013b). Filter feeders, fish, and earthworms are physically damaged and face chronic toxic effects because of MNP accumulation along the food chain (Wang et

al., 2019). Consequently, it is important to develop a smart strategy for the sustainable management of plastics.

MPs in marine waters and their adverse effects have been reported since the early 1970s, giving rise to research based on MPs in aqueous environments (Carpenter and Smith, 1972). Some studies have been conducted to investigate MPs in soil; however, there is limited focus on NPs in ecosystems (da Costa et al., 2016). Although management of MNPs is challenging, it is essential to control the utilization of products containing MNPs and their release into ecosystems by introducing new rules and regulations. Beyond the global plastic debris issue, policymakers, such as the United Nations Environment Programme (UNEP), have concentrated on MNPs by introducing a set of international and national regulations and policies (Brennholt and Heß, 2018). It has been realized that the indiscriminate disposal of plastic litter is the primary source of majority of MNPs in the environment. Therefore, both onshore and water-based policies for managing disposable plastics and other plastic products have been advocated through recycling, waste management, and wastewater management practices. Regulations for managing plastics cover various aspects, including guidelines and agreements.

The United Nation Sustainable Development Goals (UN SDGs), goal number 12, "Responsible consumption and production," (United Nations, 2010) includes the following main targets: managing chemicals and all waste categories in their life cycles and reducing waste to minimize the negative impacts on the ecosystem. A significant reduction in plastic waste in the environment can considerably reduce MNPs over time. To achieve SDG 12, cleaning of existing plastic debris will enhance the expected results. It is important to achieve the UN SDGs and progress toward a circular economy (take, make, and recycle/recover) in contrast to the linear (take, make, and waste) economic model (Brennholt and Heß, 2018). This will significantly transform plastic disposal and improve plastic recovery through recycling and

upcycling (Blank et al., 2020). Appropriate management of plastic waste and the reduced release of MNPs will also address SDG 3, “Good health and wellbeing;” SDG 6, “Clean water and sanitation;” SDG 14, “Life below water;” and SDG 15, “Life on land.” Increasing evidence suggests that MNPs may also harm aquatic and terrestrial organisms at all trophic levels, including humans. Therefore, it is necessary to encourage studies that offer solutions for the recovery of plastic waste, leading to sustainable treatment of MNPs in ecosystems.

To date, most research has addressed the distribution, transport, fate, and toxicity of MNPs in aquatic and terrestrial environments in the form of case studies (Kanhai et al., 2017; Nobre et al., 2015). Some data, such as quantity, quality, source identification, and estimates of plastics, are available for the sustainable management of MNPs. However, the problem of MNPs in the environment persists and is increasing gradually. This implies that unidentified gaps exist at the interface of research findings and practical actions, which may be due to global, regional, and local challenges, as well as the lack of infrastructure. Various studies have been conducted on sustainable plastic waste management techniques, such as source reduction, value addition, and beneficial utilization via regulatory and legislative changes. Furthermore, innovative approaches, such as plastic waste conversion to energy, biotechnological upcycling, conversion of plastic waste to value-added materials (e.g., adsorbents and catalysts), and utilization in construction materials, support the sustainable management of plastic particulates. Although many reviews have been published on the occurrence, transformation, and ecotoxicology of MNPs (Guo et al., 2020; Wang et al., 2021), the focus on innovative plastic waste management practices is currently inadequate. Hence, this review aims to provide a platform to assess the issues and prospects for sustainable management of MNPs in the environment. Moreover, this review critically explored the ecotoxicological impacts of MNPs in various environmental compartments, starting with various sources of plastics pollution. Finally, this review provides

158 a brief knowledge of current state of the art of plastic waste management strategies, policies,
159 and their practical implications.
160
161 **2. Sources and volumes of plastic wastes**
162 Plastics have entered every aspect of human life because of their extensive daily uses and
163 societal benefits (Kumar et 2020a; Andrady, 2011). The story of synthetic plastics began in
164 1907 when Leo Hendrik Baekeland discovered the first plastic, which was named Bakelite.
165 Since then, plastic production has increased from 1.5 million metric tons (MT) in 1950 to 360
166 MT in 2018 (Plastics Europe, 2016; Geyer et al., 2017). It has become one of the most abundant
167 manufactured materials (generated from around 4% of the world's fossil resources) worldwide
168 (British Plastics Federation, 2008). A list of the major countries producing and consuming
169 plastics is presented in Table 1. Asia contributes to the maximum share of plastic production
170 (China 31%, Japan 4%, and rest of Asia 17%), followed by the North American Free Trade
171 Agreement (NAFTA; 18%), Europe (17%), Middle East and Africa (7%), Latin America (7%),
172 and Commonwealth of Independent States, (CIS; 3%) (Plastics Europe, 2016). The average
173 consumption of plastics at the current rate is approximately 40 kg per person, which scales up
174 to approximately 100 kg in developed countries (Zalasiewicz et al., 2016).
175 Plastic production is linked to the growth and development of a country. High-income countries
176 (HIC) have contributed to 87% of total plastic exports since 1988. The top 10 countries
177 exporting (except Mexico) and importing (except China 1st and India 9th) plastics are HICs
178 (Brooks et al., 2018). Considering regional trade, the East Asian and Pacific (EAP) countries
179 are the major exporters of plastic waste, with significant exports from Hong Kong to China.
180 However, Europe and Central Asia countries (except for Hong Kong) are the leaders of plastic
181 waste exports, followed by North American countries (the USA and Canada). The maximum
182 plastic waste has been imported by the EAP countries (75%) since 1988 (Brooks et al., 2018).

These countries mostly comprise low- to middle-income countries. In contrast, the Organization for Economic Co-operation and Development nation members mostly comprise HICs (33 out of 35) and are the top exporters (64%). Thus, the plastics trade follows a historical parabola, and low- and middle-income countries are considered a battleground for waste management (Hoornweg and Bhada-Tata, 2012). HICs generally have an efficient and secure plastic waste sorting mechanism. Conversely, in many middle-to-low-income countries across South Asia and Sub-Saharan Africa, approximately 80%–90% of plastic waste is poorly disposed because of the lack of adequate infrastructure (Ritchie and Roser, 2018).

From the estimated total of 8300 MT of virgin plastics produced in 2015, 6300 MT of plastic waste was formed, 79% of which ended up in landfills or spilled into the environment, 12% was incinerated, and only 9% was recycled (Kumar et al., 2020b; Jambeck et al., 2015). The amount of newly manufactured or recycled plastics currently in use accounts for up to 30% (2500 MT) of all manufactured plastics (Geyer et al., 2017). Considering the ongoing production and management pace, approximately 12,000 MT of plastic waste will be delivered to landfills or released to the environment by 2050 (Jambeck et al., 2015).

Since the first buoyant plastics were observed in the ocean in 1972 (Carpenter and Smith, 1972), the final fate of plastics (99%) entering the ocean (4–12 MT) remains uncertain (Geyer et al., 2017; Schweitzer et al., 2018; Ter Halle et al., 2016). The proportion of plastics floating in the oceans is estimated to be 5.25 trillion plastic particles (0.26 MT), mostly comprising MPs (Brennholt and Heß, 2018); these often fall as part of marine snow (Taylor et al., 2016). Industries producing raw materials based on plastics (e.g., pellets and granules) act as a virgin source of leakage into marine (Maharana et al., 2020) and freshwater ecosystems (Lechner et al., 2014). For example, an Austrian plastic manufacturing plant was expected to release 94.5 tons (t) of plastic particles annually into the River Danube at a discharge rate of 100 L s^{-1} (Lechner et al., 2014).

Most marine plastics occur on land (80%) (Andrady, 2011), and are transported via various waterways such as rivers (70%–80%) (Alimi et al., 2018; Castañeda et al., 2014; McCormick et al., 2014). All channels that flow into oceans carry heavy loads of plastic waste with estimates ranging from <<1 kg day^{-1} (Hilo in the USA) to 4.2 t day^{-1} (Danube River in Europe). However, the quantity of waste transferred is characteristic of a given watershed and hence, these results are less relevant globally (Castañeda et al., 2014; Lechner et al., 2014). A large quantity of plastics entering the oceans mainly originates from coastal countries. In 2010, 192 coastal countries (6.4 billion people) generated over 275 MT of plastic waste (11% of the total MSW), of which 4.8 to 12.7 MT (1.7% to 4.6%) entered the oceans (Jambeck et al., 2015). Poorly managed plastic wastes (inadequately disposed or littered) increase the risk of eventually spilling into the sea. For instance, the Great Pacific Garbage Patch contains approximately 79 (45–129) thousand tons of marine plastics (Lebreton et al., 2018), primarily from Asian sources of plastic (Jambeck et al., 2015), the Kuroshio Extension current (Qiu and Chen, 2005), and intense fishing activities (Watson et al., 2013). Approximately 81% of China's coastal regions are contaminated with plastic waste. In 2010, 1.32–3.53 MT of plastic debris was found in marine ecosystems (Jambeck et al., 2015).

Studies have shown that WWTPs can entrap MPs with a retention of 80%–90% in sludge. Even at a reduced rate of 98.41% in the influent, a WWTP released 65 million MPs into the water every day (Murphy et al., 2016). In addition, a significant amount of microbeads (industrially-manufactured primary MPs), 7000 microbeads m^{-3}, enter the ocean through treated sewage that passes through WWTP filters (Rochman et al., 2015). Nevertheless, owing to high effluent discharge, large amounts of plastics still invade the aquatic environment, depending on the type of plastic and sewage treatment method (Mahon et al., 2017; Talvitie et al., 2017). Furthermore, sewage sludge (as fertilizer) was estimated to dump 63,000–43,000 t of MPs per year on

European farmlands, given that European cities produce 1,270–2,130 t of MPs per million inhabitants annually (Nizzetto et al., 2016).

Recycling delays the final disposal of plastics and generates secondary plastics (Mutha et al., 2006). Such secondary plastics have a low economic value and can be released into the environment as MNPs. Recycling has proven to be effective method only when re-use of plastics occurs instead of its primary production (Geyer et al., 2016), which occurs to a lesser extent than expected (Zink et al., 2018). Plastic recycling is considered an economically marginal activity. Thus far, only a small fraction (9%) of plastics have been recycled, whereas the majority (80%) have been landfilled or lost to the environment (Geyer et al., 2017; Schweitzer et al., 2018).

In addition, the recycling rate of other materials, such as metals and papers, exceeds 50% (UNEP, 2013; Van Ewijk et al., 2018). China is the largest importer of plastic waste, with a cumulative import of 45% since 1992. Plastics imported by China in 2016 accounted for 50% of total imports (14.1 MT) from 123 countries (Brooks et al., 2018). However, China's unprecedented waste import ban under the January 2018 "National Sword Program" caused plastic waste to accumulate in Australia and the US (Downes and Dominish, 2018), shifting the waste crisis to Southeast Asia (Parker, 2018). The US, which accounts for only 4% of the world's population, has the highest per capita MSW (12%) compared to the world's most populated nations such as China and India (36%), which generate 27% of waste (Jambeck et al., 2015). Germany is the most efficient country for recycling municipal waste (68%), whereas the US and Australia recycle only 35% and 11.8% of their waste, respectively (O'Farrell, 2019). Since 2006, in Europe, the amount of plastic waste sent for recycling has doubled. Moreover, the recycling rates were the highest in Europe (30%) and China (25%) in 2014, whereas only 9% was recycled in other countries such as the US (Europe, 2019; Hoornweg and

257 Bhada-Tata, 2012). Thus, it is evident that plastic pollution is severe global threat, and
258 innovative and sustainable management of waste is a time-consuming process.

259 **Table 1.** List of major plastic producing and consuming countries

Country	Production (MT)	Per capita consumption (kg capita^{-1})	Source (%)		Input to aquatic ecosystems (MT year^{-1})	References
			Packaging	Construction		
China	59.8	72.6	42.5	21.9	3.53	(EUROMAP, 2016; Jambeck et al., 2015; Ritchie and Roser, 2018)
USA	37.83	100.2	52.0	18.5	0.11	(EUROMAP, 2016; Jambeck et al., 2015; Ritchie and Roser, 2018)

Germany	14.48	98.6	49.5	21.5		(EUROMAP, 2016; Ritchie and Roser, 2018)
Brazil	11.85	33.4	51.0	19.5		(EUROMAP, 2016; Ritchie and Roser, 2018)
Japan	7.99	71.5	46.7	16.9		(EUROMAP, 2016; Ritchie and Roser, 2018)
Pakistan	6.41	8.5	50.3	18.2	0.16	(EUROMAP, 2016; Patel, 2018; Ritchie and Roser, 2018)

Nigeria	5.96	8.0	54.4	16.4	0.34	(EUROMAP, 2016; Jambeck et al., 2015; Ritchie and Roser, 2018)
Russia	5.84	44.9	50.8	21.3		(EUROMAP, 2016; Ritchie and Roser, 2018)
Turkey	5.6	92.1	45.6	19.8	0.19	(EUROMAP, 2016; Jambeck et al., 2015; Ritchie and Roser, 2018)

Egypt	5.46	19.0	43.6	24.6	0.39	(EUROMAP, 2016; Jambeck et al., 2015; Ritchie and Roser, 2018)
Indonesia	5.5	21.6	50.0	16.0	1.29	(EUROMAP, 2016; Jambeck et al., 2015; Ritchie and Roser, 2018)
UK	4.93	50.3	49.5	18.9		(EUROMAP, 2016; Ritchie and Roser, 2018)

Spain	4.71	68.2	52.7	16.1		(EUROMAP, 2016; Ritchie and Roser, 2018)
France	4.56	69.5	48.9	19.6	0.01*	(Advisors, 2019; EUROMAP, 2016; Ritchie and Roser, 2018)
India	4.49	14.2	45.4	21.8	0.24	(EUROMAP, 2016; Jambeck et al., 2015; Ritchie and Roser, 2018)
South Africa	4.47	27.5	55.3	16.2	0.25	(EUROMAP, 2016; Jambeck et

						al., 2015; Ritchie and Roser, 2018)
Iran	3.92	57.3	50.4	21.3		(EUROMAP, 2016; Ritchie and Roser, 2018)
Mexico	3.73	49	50.4	17.1		(EUROMAP, 2016; Ritchie and Roser, 2018)
Thailand	3.53	71.6	46.3	17.9	0.41	(EUROMAP, 2016; Jambeck et al., 2015; Ritchie and Roser, 2018)

Vietnam	3.27	46.6	48.2	18.7	0.73	(EUROMAP, 2016; Jambeck et al., 2015; Ritchie and Roser, 2018)

*11,200 t into the Mediterranean, excluding the Atlantic Ocean, for which data are not available.

3. Ecotoxicological impacts of plastic waste

Particulate plastic, including MNPs, can enter aquatic (freshwater and marine) and terrestrial (such as soil, sediment) ecosystems. The impacts of MNPs on environmental systems constitute a significant concern. Sustainable waste management can be useful for particulate plastic waste to maintain and increase the elasticity of ecosystems by improving soil resilience through enhancement of microbial population and nutrient cycling (Bradney et al., 2019). Particulate plastics pose ecotoxicological threats as their potential toxicity increases with decreasing size, causing harmful physical effects owing to interior abrasions and obstructions (Wright et al., 2013b). Smaller fragments such as NPs and MNPs can easily pass through the protective cell wall and cell membrane and impose adverse impacts on living organisms (Wu et al., 2021). Handling and removal of MNPs from environmental compartments are very difficult in comparison to macro-plastics while the former imposes more adverse impacts on living organisms (Zhou et al., 2021). MNP polymers cannot only release toxic organic chemicals but can accumulate in living tissues and facilitate the transports of inorganic and organic pollutants (Sridharan et al., 2021a, b; Bradney et al., 2019). These processes established MNPs as a substratum to accumulate and as a vector to transport inorganic and organic pollutants (Zhou et al., 2021; Ricardo et al., 2021).

3.1 Effects on terrestrial organisms and nutrient cycling

Previous studies have shown that MPs can accumulate and be stored in soil, altering soil properties and affecting biodiversity (Bläsing and Amelung, 2018; Huerta Lwanga et al., 2017; Maaß et al., 2017; Rillig, 2012; Rillig et al., 2017; Zubris and Richards, 2005). Presence of MNPs can alter soil microbial communities. Polyester microfiber (0.2% of the soil fresh weight) contributed to 20 – 30% increased abundance of arbuscular mycorrhizal fungi hyphae with the alteration of soil structure and water dynamics depending upon MP type, aggregate

size fraction and plant cover (De Souza Machado et al., 2019). Contrarily, polylactic acid (PLA) microparticles resulted in adverse effects on the community composition of arbuscular mycorrhizal fungi (Wang et al., 2020). Similarly, ectomycorriza were found to be sensitive to alteration of soil structure due to MPs (Ritz and Young, 2004). Various compositional elements in MPs may lead to different responses within bacterial communities. Smaller MPs due to their large specific surface area and thereby fast release of compounds via heterotrophic activities could result in a broader shift in bacterial composition than larger MPs (Ren et al., 2020). For example, a polyvinyl chloride (PVC)-MP exposure experiment found that MPs significantly increased the bacterial diversity of a *collembolan* gut, probably because of a shift in feeding behavior after MP exposure (Zhu et al., 2018). Few compostable plastic micro-films (e.g., polycaprolactone (PCL), PLA) were found to be persistent carbon resources to increase the abundance of fungi such as *Aspergillus*, *Fusarium* and *Penicillium (*Accinelli et al., 2020). It was found that MPs addition to the soil at a mass ratio of 28% resulted in 14.6–31.0%, 15.4–54.8%, 39.5–61.0%, 16.9–40.8%, 16.5–57.6%, and 30.6–42.7% reduction of catalase, phenol oxidase, manganese peroxidase, urease, laccase, and β-glucosidase activities, respectively (Yu et al., 2020). Another study found that high level of MPs (28%) increased FDAse and phenol oxidase activity while imparting no such effect at low MP level (7%), indicating that the effect of MPs depends on the dose (Liu et al., 2017). Hence, types, properties and concentrations of MNPs in the soil medium significantly affect the soil microbial population and biological functions.

The interaction of MPs with heavy metals and metalloids (e.g., cadmium [Cd], As) was reposted to drastically reduce the biomass and chlorophyll contents in plants (Wang et al., 2020). The interaction of As with polystyrene MPs (PSMP) and polytetrafluorethylene (PTFE) through bridging with humic acid resulted in a decrease of As bioavailability in the soil, which consequently increased the relative abundance of *γ-proteobacteria*, *α-proteobacteria*, and

313 *Bacteroidia* (Dong et al., 2021). The increase in microbial abundance and some enzymatic
314 activity in the soil was likely due to the decrease of As bioavailability and supply of carbon to
315 microorganisms through PSMP and PTFE (Dong et al., 2021). However, in severely As-
316 contaminated soil, the enzyme activities were reduced even after the addition of PSMP and
317 PTFE (Dong et al., 2021).
318 The combined toxicity of heavy metals and pesticides along with MNPs are very alarming
319 although research on this topic is limited till date. One such study exposed adult zebrafish to
320 100 μg L^{-1} imidacloprid (IMI), 20 μg L^{-1} PSMP, and a combination of PS and IMI (PS + IMI)
321 for 21 days (Luo et al., 2021) (Table 2). The PS and IMI inhibited the growth of zebrafish, and
322 altered the levels of glycolipid metabolism and oxidative stress-related biochemical parameters
323 like super oxide dismutase (SOD) and catalase (CAT) activity. The IMI and PS alone and their
324 combination significantly increased SOD and CAT activities in the liver of zebrafish,
325 indicating an oxidative stress and inflammation in the test organism (Luo et al., 2021).
326 Similarly, the effect of polyester fibres and crumb rubber MPs alone and in the presence of
327 chlorpyrifos was tested on the immune parameters of *Porcellio scaber*, revealing that
328 chlorpyrifos alone significantly affected the haemocyte viability of *P. scaber*, but the presence
329 of both the MPs improved the haemocyte count of *P. scaber* by reducing the chlorpyrifos
330 bioavailability in the system (Dolar et al., 2021). In another study, cationic amine modified PS
331 NPs at 5 mg L^{-1} showed no inhibitory effect on *Microcystis aeruginosa,* a blue green alga,
332 while glyphosate had strong inhibitory effect on the microorganism (Zhang et al., 2018a). The
333 combination of PS and glyphosate had an antagonistic inhibition effect on *M. aeruginosa* due
334 to the adsorption of glyphosate by the PS particles (Zhang et al., 2018a). Similarly, glyphosate
335 alone significantly reduced the relative growth rate, photosynthetic capacity, and root activity
336 of fern *Salvinia cucullata*, but glyphosate along with PSMPs activated the fern's antioxidant
337 defence system by increasing SOD and CAT activities, thereby getting the plant accustomed

with oxidative stress (Yu et al., 2021). In contrast, the presence of PS spheres did not affect deltamethrin (a pyrethroid insecticide) and dimethoate (an organophosphate insecticide) toxicities to *Daphnia magna*, suggesting that PS spheres did not act as a vector for the uptake of these pesticide by *D. magna* (Horton et al., 2018). Therefore, the combined toxicity of MNPs, pesticides and heavy metals may vary depending on MNP types, chemistry of the associated contaminants and defence mechanisms of a concerned organism, which warrant future research involving molecular level ecotoxicology studies.

346 **Table 2:** Selected references on the impact of MNPs and associated organic pollutants/additives on living organisms

MNP type and size	**MNP concentration**	**Organic pollutant/additive**	**Organism**	**Impacts**	**Reference**
PS; 5 μm	100 μg/L	Imidacloprid (IMI); 20 μg/L	Zebrafish	Inhibited growth, altered enzymatic activities	(Luo et al., 2021)
PS; 20, 65 μm	–	Butylated hydroxyanisole (BHA)	Zebrafish	Impaired larval growth	(Zhao et al., 2020)
PS; 50–100 nm	1000 mg kg^{-1}	Tetracycline	*Enchytraeus crypticus*	Increased the number of antibiotic resistant gene	(Ma et al., 2020)
Polyamid; 40–63 μm	–	Phenanthrene	*Gammarusroeseli*	Neurotoxicity and locomotor toxicity	(Bartonitz et al., 2020)
PS; 0.1–5 μm	–	Increased toxicity	*Chlorella pyrenoidosa*	Dibutyl phthalate	(Li et al., 2020)
PS; 100 nm	1–75 mg L^{-1}	Polychlorinated biphenyls (PCBs)	*D. magna*	Increased concentration of MNP increased lethality	(Lin et al., 2019)

PS; 45 μm	10 mg L^{-1}	Polyaromatic hydrocarbons (PAHs)	*D. rerio*	Impaired energy production, decreased development, impaired vascular development	(Trevisan et al., 2019)
PE, polyhydroxybutyrate (PHB); 10–90 μm	–	Fluoranthene	*M. edulis*	Reduced enzymatic activities	(Magara et al., 2019)
PS; 568 nm NPs at environment relevant concentrations	0.4 mg L^{-1}	PAHs	*Perinereis aibuhitensis*	Small concentration of MNPs contributed little to bioaccumulation of PAHs	(Jiang et al., 2019b)
PE, PET; 1–10 μm	–	Glyphosate	*D. magna*	Increased mortality	(Zocchi and Sommaruga,

					2019)
Amino-modified PS; 200 nm	3–20 mg L^{-1}	Glyphosate	*Microcystis aeruginosa*	Inhibited photosynthetic efficiency	Zhang et al. (2018)
PS; 100 nm	2–20 mg L^{-1}	PCBs	*D. magna*	Enhanced the accumulation of PCBs	(Jiang et al., 2018)
PS; 50 nm	10 mg L^{-1}	2,2′,4,4′ - tetrabromodiphenyl ether, triclosan	*Brachionus koreanus*	Reduced the activities of multidrug resistance proteins and P-glycoproteins, produced oxidative stress	(Jeong et al., 2018)
PE; PS 0.5–1.0 μm	–	Organophosphorus flame retardants	*Mus musculus*	Induced oxidative stress, neurotoxicity, and disrupted energy metabolism	(Deng et al., 2018)

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Apart from that, the presence of MNPs can affect nutrient cycling through affecting organic matter (OM) decomposition, nitrification and denitrification, and by changing the microbial communities in soil, sediment and water (Chen et al., 2020b; Seeley et al., 2020; Zhang et al., 2021). Available phosphorus (AP) and available nitrogen (AN) and OM contents were decreased with increasing PSMP and PTFE concentrations in an uncontaminated soil, but AN and AP were increased in an As-contaminated soil where the availability of nutrients was regulated by soil pH changes caused by MPs (Dong et al., 2021). When the initial solution pH exceeded pH_{PZC} of the MPs, the hydroxyl groups on the surface of the MPs consumed the OH^- in the solution through deprotonation, leading to a decrease in the pH of the solution (Calvo et al., 2014), which was proportionately dependent on size and point of zero charge (PZC) of the MPs (PZC_{PSMP} = 3.3-4.0; PZC_{PTFE} = 3.9-4.6) (Dong et al., 2021). In the Brisbane River sediment, the total content of N (TN) and P (TP) was increased with increasing concentration of MPs (>20 mg kg^{-1}), indicating a positive correlation between total carbon (TC) and MPs concentration, and development of a conducive microbial habitat in the sediments by MPs (He et al., 2020). In a fresh water pond system, square-shaped PP MPs accelerated biofilm formation, increased ammonia and nitrite oxidation and denitrification rates, and accumulated P temporarily by enhancing the alkaline phosphatase activity (Chen et al., 2020b). However, disintegration of matured biofilms released N and P into the water at a later stage (Chen et al., 2020b). Similarly, an outdoor mesocosm experiment showed that earthworm *Arenicola marina* produced less casts in sediments containing MPs than uncontaminated control, indicating chemical leaching of vinyl chloride monomers causing toxicity to *A. marina* (Lithner et al., 2011). Metabolic rate of *A. marina* was increased, while microalgal biomass was decreased at high concentration of MPs (~2%), suggesting that small plastic debris physically increased the sediment porosity, altered water movement, and reduced heat transfer in the sediment, which

subsequently altered the algal growth (Ras et al., 2013). The average flux of NH_4^+ was positive in the PVC-amended sediment, but changes in concentration of NO_3^- and NO_2^- in pore water due to sediment PVC contamination was not measurable (Green et al., 2016). Thus, concentration and properties of MNPs have significant impact on terrestrial organisms and maintaining soil/sediment nutrient cycling to sustain the terrestrial ecological balance.

The uptake and transfer of MNPs by crops and the edible parts of plants may seem implausible; MNPs can enter the human food web through alternative pathways. The mobility of MNPs in soil depends on their particle size and texture. For instance, the transportation of MNPs in fine-grained soil is possible but not significant (Bläsing and Amelung, 2018; Zhang et al., 2010). Information on the response of plants to the presence of MNPs in the soil is limited (Table 3). A recent study on *Allium fistulosum* (spring onion) determined that the response to MP exposure depends on the shape and type of the polymer. Exposure to PS and polyester fibers significantly increased root biomass. However, no visible effect was observed with high-density polyethylene (HDPE) particles (De Souza Machado et al., 2019). Despite preliminary research, plant growth retardation has been investigated in soils containing MPs (Qi et al., 2018; Rillig et al., 2019). MPs can affect soil characteristics and their inhabitants; for example, MP treatment disturbed microbial activity, plant performance, soil structure, and physiological adaptation of spring onions (De Souza Machado et al., 2019; Qi et al., 2018; Rillig et al., 2019). MPs can also affect the transport and absorption of nutrients in plants by triggering soil aggregation and clustering at various levels (e.g., loose fragment-nature and tight linear-nature) (Asli and Neumann, 2009; De Souza MacHado et al., 2018; Ma et al., 2010; Zhang and Liu, 2018). In contrast, a study on wheat observed no significant changes in seedling appearance and mass under MP treatment (Judy et al., 2019). In *Vicia faba* plants, PS fluorescent MPs induced oxidative stress and genotoxicity when exposed to 100 nm MPs at 10, 50, and 100 mg

L^{-1} (Jiang et al., 2019a). The study also showed that 5 mm PS MPs could not be transported to plants and were deposited on the exterior of the roots.

Consequently, plant growth was impeded owing to the clogging of root tips in *V. faba*, which hindered the uptake or absorption of water and essential nutrients. In addition, terrestrial snails (*Achatina fulica)*, commonly found in forests, farmlands, and gardens, showed oxidative stress and adverse effects on fitness when exposed to MPs (Song et al., 2019b). However, the impact of particulate plastics indicates that they can pose a significant threat to future ecology, particularly the interaction between soil and plant systems depending upon their type, size and properties. Furthermore, most of the recent studies are conducted for short duration and mainly at laboratory scale, which warrants urgent focus on long term field-based studies to identify the real impacts of MNPs on terrestrial flora and fauna.

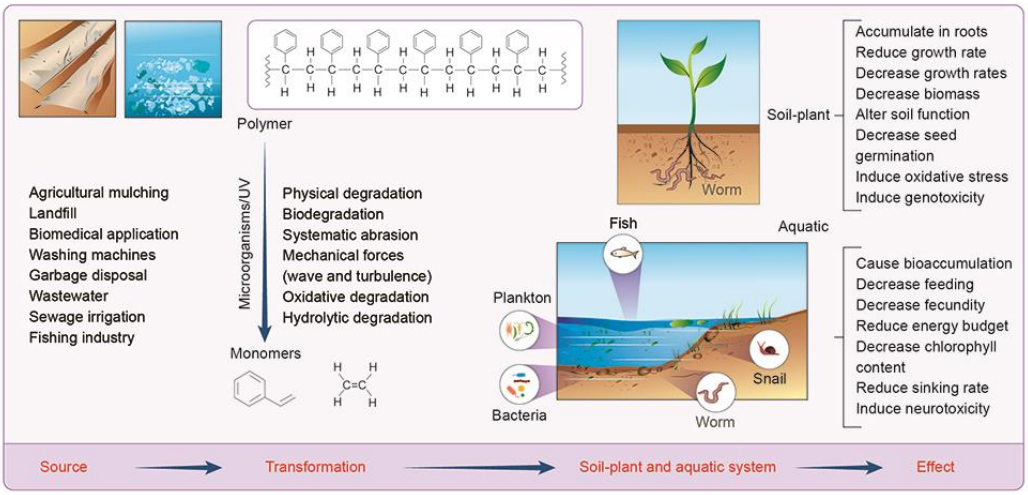

Fig. 1. Schematic diagram showing sources, transformation, and the fate of plastic contaminants in soil-plant and aquatic environments.

3.2 Effects on aquatic and marine organisms

Plastic waste can be fragmented into particulate plastics in aquatic systems (i.e., oceans, rivers, and lakes) owing to many influences such as waves, systematic abrasion, and oxidative (e.g.,

photo and thermal) processes (Andrady, 2011). Worldwide dispersion and accumulation of MPs in the environment can negatively affect marine and other aquatic systems (Table 3). MPs are mainly observed at high concentrations in populated coastal areas and adjacent urban areas (Ivar Do Sul and Costa, 2014). A wide range of organisms consume or ingest MPs when the size of MPs matches the size of their prey, leading to enhanced bioavailability of MPs in organisms (Galloway and Cole, 2018; Wang et al., 2016). Environmental MPs have been observed in the food web, including organisms from primary to higher trophic levels, which could affect the entire food web (Oliveira et al., 2012; Wright et al., 2013b).

PSMPs significantly affect the fitness, feeding activity (7.4%), and bioaccumulation in *Arenicola marina* (lugworm), even at low exposure levels (Besseling et al., 2013). The ingestion of MPs is a threat to the fish community and increases premature mortality. In a study on *Artemia* sp. nauplii and *Danio rerio* (zebrafish), *Artemia* sp. nauplii were exposed to MPs of various sizes (1–20 μm); MPs were then observed in zebrafish fed with these nauplii (Batel et al., 2016). This study demonstrated that MPs can move along the food web at various trophic stages. When MPs are digested, minute pieces of MPs can move to the muscles of fish and other marine animals. Other chemicals associated with MPs have been observed worldwide, including in subtropical and coastal areas (Hirai et al., 2011). For example, the marine fish *Oryzias latipes* ingests and loads toxic substances adsorbed on MPs, causing oxidative stress, pathological toxicity, and liver inflammation (Rochman et al., 2013).

The accumulation of MPs in various fish tissues depends on the size of the MPs and the type of tissue and organ. In zebrafish, 5 μm PSMPs accumulated in the liver, gills, and intestines, whereas 20 μm PSMPs only accumulated in the gills and gut. PSMPs also induce liver inflammation, enhance catalase and superoxide dismutase, and modify lipid, energy, and metabolic profiles in zebrafish (Lu et al., 2016). The neurotoxic effects of MPs have been found in living organisms, but little is known regarding their toxicity mechanisms. PEMPs adversely

affected neural function of the common goby *Pomatoschistus microps* (Luís et al., 2015), decreased acetylcholinesterase activity, and suppressed neural activity in the marine mussel *Mytilus galloprovincialis* (Avio et al., 2015).

In *Perca fluviatilis* (European perch), the accumulation and ingestion of particulate plastic (90 μm sized PSMPs) impeded growth and hatchability, modulated inborn behavior and feeding choices, altered olfactory aptitudes, and increased the likelihood of killing by feeders (Lönnstedt, and Eklöv, 2016). In summary, fish and other marine organisms prefer to consume particulate plastics over typical food, and this has an immediate effect on their digestion.

The problem of particulate plastic ingestion is not limited to fish. Other marine organisms, including zooplankton (e.g., copepods), algae, sea turtles, and sea cucumbers, are also vulnerable to particulate plastics (Auta et al., 2017). Filter feeders, such as copepods, are a vital component of the marine food web. The decline of filter feeders in aquatic settings can jeopardize various trophic stages. In the treatment of MPs on *Calanus helgolandicus* (marine copepods), PS plastic adversely affected fertility, movement, and marine copepod behavior (Cole et al., 2016). Decreased feeding rates as an effect of MPs have been reported in *Carcinus maenas* (shore crab) (Watts et al., 2015) and *Nephrops norvegicus* (Welden and Cowie, 2016). Increased immunological responses have been observed in mussels (Von Moos et al., 2012); however, marine copepods showed decreased fertility, survival, and feeding rates owing to MP ingestion (Cole et al., 2015; Lee et al., 2013).

Ingestion of PS enhanced mRNA expression, caused cell stress and inhibited acetylcholinesterase activity in *Artemia franciscana* (brine shrimp) (Eom et al., 2020). Recent studies have shown that *Ostrea edulis* (European flat oyster) was affected by 2 months of exposure to PLA MPs at a concentration of 80 μg L^{-1}, suggesting that *O. edulis* was under stress (Green et al., 2016). In the same study, the biomass and abundance of benthic macrofauna, including *Idotea balthica* (isopod) and shell clams (*Scrobicularia plana* and

Scrobicularia plana) decreased because of reduced mortality, feeding, and fertility after MP ingestion (Auta et al., 2017; Green et al., 2016).

Exposure to particulate plastics causes significant growth inhibition in microalgae, and this inhibition increases with increasing exposure concentrations. However, the adverse effects of MPs can decrease with increasing particle size (Besseling et al., 2014; Mao et al., 2018; Sjollema et al., 2016; Zhang et al., 2017). In *Skeletonema costatum*, MPs induced a significant decrease in photosynthetic competence and chlorophyll levels (Zhang et al., 2017). Several studies have found that MP exposure causes significant oxidative stress in intracellular organs and various physical reactions in freshwater microalgae and affects gene expression in specific metabolic processes (Mao et al., 2018). It is very important to identify the combined effects of MPs and other pollutants, as organisms are typically simultaneously exposed to a complex mixture of MPs and adjacent substances in nature.

Combined with toxic chemicals, MPs can cause deleterious effects, such as mortality, metabolic anomalies (Oliveira et al., 2013; Rist et al., 2016), and pathological issues in aquatic animals (Rochman et al., 2013). However, the levels of MP ingestion and pathways for enhancing the movement of pollutants in aquatic animals are controversial (Koelmans et al., 2016). In many countries surrounded by coastal regions, particulate plastic pollution, economic loss, and ecological and toxicological impacts on human health and biota are severe (Sridharan et al., 2021a). Therefore, a synergistic understanding on MP toxicity mechanisms with sorbed chemicals and additives (Table 2) is essential for the sustainable management of MPs.

487 **Table 3.** Ecotoxicological effects of particulate plastics on plant-soil systems and aquatic organisms

Species	Microplastic size	Concentration	Exposure time	Main finding(s)	Reference
Vicia faba	5 μm and 100 nm PS	10–100 mg L^{-1}	2 d	MPs accumulation in roots, a decrease in biomass, a decrease in growth; induction of oxidative stress, and genotoxicity	(Jiang et al., 2019a)
Allium fistulosum	8 μm PS	2% (wt/wt)	60 d	Changes in soil structure, alteration of water dynamics and microbial activity in soils, and adverse effects on seed germination and root length	(De Souza Machado et al., 2019)
Lepidium sativum	0.05 μm PS	10^3 to 10^7 particles L^{-1}	3 d	A decrease in seed germination and root growth	(Bosker et al., 2019)
Achatina fulica	76.3 μm PET	0.01–0.71 g kg^{-1}	28 d	Oxidative stress and adverse effect on fitness	(Song et al., 2019b)
Arenicola marina	25–150 mm UPVC	5% (wt/wt)	10 d	A decrease in energy budget and feeding	(Wright et al., 2013a)

Paracentrotus lividus	50 nm PS	3 and 25 mg L^{-1}	2 d	Developmental defects	(Della Torre et al., 2014)
Arenicola marina	400–1300 μm PS	0–100 g L^{-1}	28 d	Reduction in feeding activity and decrease in body weight and energy efficiency	(Besseling et al., 2013)
Daphnia magna	0.05–10 μm PS	5 mg L^{-1}	14 d	Increased bioaccumulation	(Ma et al., 2016)
Skeletonema costatum	1 μm PVC	0–50 mg L^{-1}	4 d	Inhibition of algae growth and reduction in chlorophyll content and photosynthesis	(Deng et al., 2017)
Scenedesmus obliquus	~ 0.07 μm PS	44–1100 mg L^{-1}	3 d	Inhibition of algae growth and reduction in Chlorophyll-a content	(Besseling et al., 2014)
Calanus helgolandicus	20 μm PS	1000 particles L^{-1}	1 d	Reduction in sinking rate and densities	(Cole et al., 2016)
Centropages typicus	7.3 μm PS	4000–25000 beads L^{-1}	1 d	Significant reduction in algal ingestion	(Cole et al., 2013)
Artemia franciscana	1–10 μm PS	1–1000 particles L^{-1}	30 d	Increased mRNA expression, enzymatic activities, and mortality, inhibition of acetylcholinesterase activities	(Eom et al., 2020)

Sparus aurata and *Dicentrarchus labrax*	40–150 μm PVC and PE	1–100 mg L^{-1}	1 d	No effects on cell viability, oxidative stress, immunity-related gene expression, and cell death	(Espinosa et al., 2018)

PS, polystyrene; UPVC, unplasticized polyvinyl chloride; PVC, polyvinyl chloride; PE, polyethylene; PET, polyethylene terephthalate

4. Plastic waste management considerations

The potential for downstream plastic waste mitigation is extremely low owing to the low value of most post-consumer plastic products and low recovery incentives. Appropriate segregation, identification, and quantification are required to control the entry of plastics into the environment, which is currently lacking in many countries. Precise annual statistics on plastic waste generation and information on its categories are urgently required to determine regional best management practices and set up treatment facilities (Singh and Sharma, 2016). Implementing stringent laws and strengthening legal measures can control the mishandling and leakage of parent plastic materials in the supply-usage chain (Kish, 2018). Certain random efforts include an upfront tax to fund the mitigation of plastic waste and increase public awareness. However, the realization of such strategies becomes impractical because of the dispersion and difficulty in collecting small MNPs.

Cleaning up the final sink, such as the soil and ocean, is not an economically or logistically feasible option. Hence, reasonable four-level upstream mitigation strategies, such as source reduction and a zero waste strategy, infrastructural development for managing and mitigating point and nonpoint sources of pollution, waste resource reuse and recovery, and value addition, have often been proposed in the literature (Sarkar et al., 2020). Source reduction is the best long-term solution for MP pollution. Regulating primary MPs such as microbeads has been possible via source control (Welden and Cowie, 2016). However, plastic wastes continuously undergo fragmentation and weathering via physical, chemical, and biological processes (Ragaert et al., 2017), making the management of MNPs tremendously challenging. Manufacturers need to innovate sustainable solutions for benign alternatives using eco-friendly and biodegradable raw materials that reduce the amount of MNPs (Engler, 2012; Peng et al., 2017). However, degradable plastics cannot be considered harmless because they can be easily disintegrated into MNPs. Improving the infrastructure of developing countries, such as

trapping plastic wastes in stormwater drainage systems and collecting waste in managed rubbish bins, can mitigate plastic pollution.

Removing MNPs using membrane filtration in WWTPs is another option (Talvitie et al., 2017). In the membrane filtration method, biological catalysts either potential microorganisms or robust enzymes, or both are linked with a partition method and run by a film-derived system, such as microfiltration and ultrafiltration (Zhou et al., 2021). Membrane filtration method exhibited a substantial removal efficiency (99%) of MPs, improved final effluent quality, and minimized treatment steps. However, this technology requires high cost and energy, and may not be suitable for large-scale implementation. Finally, upcycling plastic waste can create new products and encourage waste recycling. As novel, environmentally friendly, and economically attractive valorization pathways are urgently needed to mitigate plastic pollution in the environment, a few recent innovations in this direction are discussed in the following sections.

5. Innovations in particulate plastic waste management

Concerted efforts have been made to reduce, reuse, and recycle waste to minimize plastic waste disposal (Awoyera et al., 2020). As shown in Fig. 2, the recycling of plastic waste reduces the environmental burden and generates paths for further applications, such as the fabrication of value-added carbon materials (Chang et al., 2018; Chen et al., 2020a; Cho et al., 2019; Dankwah et al., 2018; Wen et al., 2019), fuels, aromatic chemicals (Miandad et al., 2018; Omol et al., 2020; Thahir et al., 2019), construction/cementitious materials (Awoyera and Adesina, 2020; Lahtela et al., 2019), polymers, and fibers (Choi et al., 2019; Hu et al., 2019).

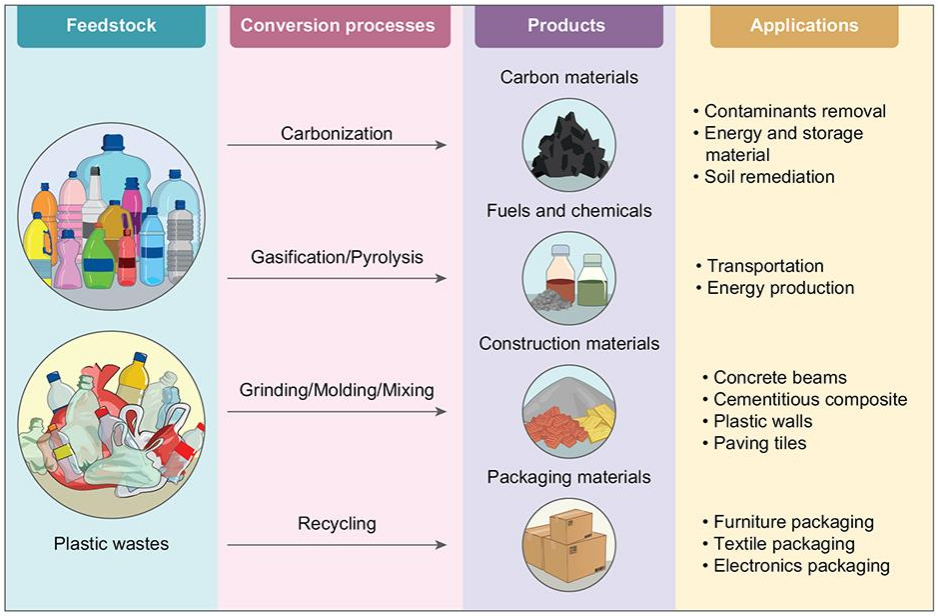

Fig. 2. Paths to the sustainable management of plastic waste for beneficial reuse.

5.1 Value-added carbon materials

Considering the abundance of carbon atoms in plastic waste, the fabrication of carbonaceous materials via the carbonization of plastic waste is an emerging recycling approach (Min et al., 2019). To date, several types of plastic polymers, such as polypropylene (PP), PE, PS, PVC, and polyethylene terephthalate (PET), have been effectively converted to carbon nanotubes (CNTs), carbon hollow spheres (HCSs), porous carbon sheets (PCSs), and biochar composites (Chen et al., 2020a; Choi et al., 2019; Gong et al., 2019; Ma et al., 2018; Song et al., 2019a). Min et al. (2019) synthesized economical porous carbon flakes (PCFs) via PS waste pyrolysis. Hybrid PCF-MnO_2 was also fabricated using the selective deposition of MnO_2 nanosheets on the surface of the PCFs. The synthesized materials showed outstanding cyclic stability and capacitance retention in the supercapacitor devices (Table 4). Ma et al. (2018) synthesized PCSs using MgO from a mixture of thermoplastic wastes (Table 4). This modest approach converts plastic waste into valuable products according to the imperative need for waste management and environmental sustainability.

555 Zhang et al. (2019) fabricated hierarchically porous carbon by carbonizing low-cost PET waste
556 using $ZnCl_2$/NaCl eutectic salts (Table 4). The fabricated material displayed a high specific
557 surface area, low thermal conductivity, rapid water transportation, and improved solar
558 absorption efficiency. Choi and Lee, (2019) adopted an electron beam treatment to synthesize
559 carbon nanosheets (CNSs). This method provides a new avenue for producing PE-derived
560 CNSs that can be successfully applied as transparent conducting electrodes. Activated carbon
561 (AC) was also synthesized (de Paula et al., 2018) from waste PS foam (PSF) with outstanding
562 textural characteristics, without any prior treatment through a modest and conventional two-
563 step process (i.e., carbonization and chemical activation). AC displayed improved capacitance
564 and excellent dye adsorption efficiency (> 1 g g^{-1}, Table 4). Oh and Seo (2019) produced a
565 PET biochar composite through the co-pyrolysis of PET waste and rice straw. This composite
566 can be used to adsorb various pollutants such as 2,4-dinitrotoluene (DNT), 2,4-dichlorophenol
567 (DCP), Pb, selenate (SeO_4^{2-}), and chromate (CrO_4^{2-}). These promising results suggest that the
568 co-pyrolysis process can enhance the pollutant removal efficiency of biochar-mineral
569 composites. Zhang et al. (2018b) also incorporated HDPE into rice husk biochar to fabricate
570 biochar/plastic composites (BPC) using the extrusion method. The bending and tensile
571 strengths of BPC reached 53.7% and 20 MPa, respectively, which far exceeded those of wood-
572 plastic composites.
573

Table 4. Synthesis of value-added carbon materials from plastic waste.

Plastic polymers	Process	Product	Product characteristics	Applications	Reference
PS-waste	Template method; direct pyrolysis of PS-waste at 700 °C for 5 h and selective deposition of MnO_2	Porous carbon flakes (PCFs)	Specific surface area = 1087 m^2 g^{-1}; specific capacitance = 308 F g^{-1} at 1 mV s^{-1} and 247 F g^{-1} at 1 A g^{-1}; cycle stability = 93.4% over 10,000 cycles	Electrode material for capacitor	(Min et al., 2019)
PS	Template method; carbonization at 700 °C for 1 h and activation with KOH	Hierarchical porous carbon sheets (PCSs)	Specific surface area = 2650 m^2 g^{-1}; pore volume = 2.43 cm^3 g^{-1}; specific capacitance = 323 F g^{-1} at 0.5 A g^{-1} and 222 F g^{-1} at 20 A g^{-1}; cycle stability = 92.6% over 10,000 cycles; energy density = 44.1 Wh kg^{-1}; power density of 757.1Wkg^{-1}	Electrode material for capacitor	(Ma et al., 2019)

Mixed thermoplastics (PE, PP, PS, PVC, PET)	Pyrolysis at 500 °C with MgO	PCSs	Specific surface area = 713 $m^2 g^{-1}$; pore volume = 5.27 $cm^3 g^{-1}$	Not specified	(Ma et al., 2018)
PET	Controlled carbonization at 500 °C with $ZnCl_2$/NaCl eutectic salts	Hierarchically porous carbon	Specific surface area = 645.6 $m^2 g^{-1}$; pore volume = 0.606 $cm^3 g^{-1}$; energy conversion efficiency = 97%	Energy storage and conversion, environmental remediation (metallic ion removal efficiency = 99.9%; dye removal efficiency > 99.9%)	(Zhang et al., 2019)

Waste PSF	Pyrolysis at 530 °C for 5 h, activation with KOH	Activated carbon (AC)	Specific surface area = 2712 $m^2 g^{-1}$; pore volume = 1.2 $cm^3 g^{-1}$;	Energy storage and conversion, environmental remediation (methylene blue (MB) removal = 1.04 g g^{-1})	(de Paula et al., 2018)
Low-density polyethylene (LDPE)	Carbonization at 530 °C for 2 h, doping with metals (Fe, Co, Ni)	Graphitic and magnetic carbon spheres nanocomposites	Magnetism; LDPE-Fe = 6.00 emu g^{-1}, LDPE-Co = 12.15 emu g^{-1}, LDPE- Ni = 2.58 emu g^{-1}; high degree of graphitization, high conductivity, and high thermal stability	Energy storage and conversion, environmental remediation	(Castelo-Quibén et al., 2019)
Mixed thermoplastics (PE,	Catalytic carbonization at 700 °C modified organically	Hierarchically PCSs	Specific surface area = 2198 $m^2 g^{-1}$; pore volume = 3.026 $cm^3 g^{-1}$; specific capacitance = 207 F g^{-1} (aqueous	Electrode material for capacitor	(Wen et al., 2019)

PP, PS, PVC, PET)	montmorillonite, KOH activation		electrolytes) and 120 F g^{-1} (organic electrolytes) at 0.2 A g^{-1}		
Mixed plastic wastes	Not specified	Reductants	Reduction ranged 68.7- 91.24%	Reductants for metal oxides reduction	(Dankwah et al., 2018)
PE-wastes	Carbonization at 1200 °C under N_2 condition	Transparent conducting film (TCF)/ carbon nanosheet (CNS)	Open-circuit voltage (V_{OC}) = 0.60 V, short-circuit current (J_{SC}) = 6.34 mA cm^{-2}, fill factor (FF) = 52.2%, power conversion efficiency (PCE) = 1.98%	Energy storage and conversion	(Choi et al., 2018)
PVC plastic wrap	KOH-assisted room temperature dehalogenation and annealing at 600 °C for 2 h	Carbonaceous materials	Specific surface area = 641 m^2 g^{-1}; specific capacitance = 399 F g^{-1} (KOH electrolytes) and 363 F g^{-1} (H_2SO_4 electrolytes) at 0.2 A g^{-1}	Energy storage and conversion	(Chang et al., 2018)

PET waste	Carbonization at 500 °C AC for 2 h ($ZnCl_2$ activated) and 800 °C for 2 h (K_2CO_3 activated)	Specific surface area = 700 m^2 g^{-1} ($ZnCl_2$ activated) and 1400 m^2 g^{-1} (K_2CO_3 activated)	Environmental remediation (MB removal = 625 mg g^{-1} and victoria blue B = 323 mg g^{-1})	(De Castro et al., 2018)

5.2 Fuels and chemicals: waste to energy

The utilization of plastic waste as feedstock for the production of fuels and chemicals is an evolving technology that might help achieve sustainability in the near future (Miandad et al., 2019; Song et al., 2019a). Thermochemical conversion technologies, such as gasification and pyrolysis, are considered a promising approach for converting plastic waste into bio-oil, biochar, and syngas (Fivga and Dimitriou, 2018; Lopez et al., 2018). The gasification of plastic waste primarily produces a mixture of gaseous fuel comprising H_2, CO, CO_2, CH_4, and N_2 (Lopez et al., 2018). In general, gasification offers better flexibility in converting plastic waste or mixed waste with/without other feedstocks to valuable products (Lopez et al., 2018). The main drawback associated with plastic waste gasification is the high tar content in the gaseous products (Pinto et al., 2016).

At present, pyrolysis, catalytic steam reforming (Miandad et al., 2019; Omol et al., 2020), and hydrothermal technologies are emerging applications (Nabgan et al., 2017). The pyrolysis-reforming method yields high H_2, generally more than 30 g per 100 g of plastic waste (Barbarias et al., 2016). The gaseous product is free of tar and nullifies the gasification problem (Lopez et al., 2018). The major products of pyrolysis include bio-oil as a heavy auxiliary fuel, and biochar and syngas are value-added co-products, making the overall process cost-effective (Fivga and Dimitriou, 2018). Various catalysts can be applied in the pyrolysis process to enhance the hydrocarbon yield and ensure fuel properties similar to those of fossil fuels (Sun et al., 2018; Zhang et al., 2019). Fuel yields from the catalytic pyrolysis of various plastics vary in the range of 84.5–89.2 wt.% for PE, 93–96 wt.% for PS, and 84 wt.% for PP (Fivga and Dimitriou, 2018).

Miandad et al. (2019) applied modified zeolite as a catalyst for converting plastic waste (PS, PE, PP, and PET both individually and in mixed ratios) to liquid fuel at the bench scale. This process yielded 70% higher oil than with PP (54%) and PE (42%), and the product displayed

high heating values in the range of 41.7–44.2 MJ kg^{-1}, which is comparable to the petroleum diesel fuel (Miandad et al., 2019). Budsaereechai et al. (2019) used low-cost binder-free pelletized bentonite clay as a catalyst for bio-oil production from plastic wastes (PS, PP, low-density polyethylene (LDPE)), and HDPE) by catalytic pyrolysis. The bio-oil thus obtained displayed better engine performance and lower emissions of carbon monoxide (CO) and carbon dioxide (CO_2) than uncatalyzed bio-oil and fossil fuels (Budsaereechai et al., 2019). Thahir et al. (2019) utilized PP plastic waste as the feedstock at a pyrolytic temperature range of 500–650 °C in a fixed-bed reactor under vacuum and obtained 88 wt.% mixed oil, 6–67 wt.% gasoline, and 64–83 wt.% diesel oil. Shah et al. (2010) reported the pyrolysis of PE waste and the yields of various end products such as pyrolytic oil (48.6%), pyrogas (40.1%), wax (10.1%), and char (0.6 wt%).

Al-Salem et al. (2020) utilized mixed plastic waste collected from a landfill site and demonstrated the production and yields of pyrolysis oil (5.5 wt.%), light wax (23.8 wt.%), heavy wax (69.4 wt.%), and gaseous constituents (1.3 wt.%). Production cost is a crucial factor for the application of thermochemical technology in waste management. A pyrolysis plant with a utilization capacity of 100 kg h^{-1} plastic waste was designed to study the cost-effectiveness of the technology using Aspen HYSYS software (Fivga and Dimitriou, 2018). At the proposed rate of 1000 kg h^{-1}, the fuel production cost was £0.87 kg^{-1}, which was 58% higher than the current market scenario. However, a large-scale production approach may lower the cost by 2–18.9 times and signify potential cost-effectiveness (Fivga and Dimitriou, 2018). This type of techno-economic analysis needs to be further emphasized in the near future to make plastic waste conversion technology more economically competitive in the market.

5.3 Construction materials

Plastic waste can be utilized as construction materials in various forms, such as lightweight aggregates, asphalt mixtures, fillers, and insulation materials (Awoyera et al., 2019; Hashem et al., 2019; Kamaruddin et al., 2017). Dhawan et al. (2019) recycled non-degradable plastic waste bags as floor and wall tiles. The products exhibited lower flammability and improved tensile strength. Khalid et al. (2018) demonstrated the application of plastic waste to fiber-reinforced concrete beams. The mixing of plastic waste improves the mechanical properties, such as the first crack load and strength of the beams. Owing to the lightweight nature of plastic waste, they can be utilized in the fabrication of lightweight cementitious composites (Hama and Hilal, 2017). In addition, plastic waste can be used to fabricate asphalt mixtures (Awoyera and Adesina, 2020). In particular, the application of LDPE and HDPE plastic waste is suitable for producing asphalt mixtures (Awoyera et al., 2020; Kumi-Larbi et al., 2018). Plastic waste is also used as substitutes for traditional blocks, bricks, and wood walls (Awoyera and Adesina, 2020). Recycled plastic waste were incorporated into heat molds and compressed to produce bricks. Nevertheless, construction materials containing plastic waste cannot be used for load-bearing purposes and are more appropriate for constructing partition walls. Kumi-Larbi et al. (2018) prepared sand blocks using plastic waste and displayed strong and durable mechanical properties.

Akinwumi et al. (2019) examined the utilization of shredded plastic waste to synthesize stabilized earth blocks. The application of a 1 wt.% shredded plastic waste of sizes smaller than 6.3 μm was effective in block production. Aciu et al, (2018) used recycled plastic waste to construct eco-friendly mortars, in which 75 wt.% of PVC waste was used as a partial substitute for sand. The obtained mortar showed properties similar to those of the masonry mortar grade M20. Introducing plastic wastes into concrete as a partial replacement for sand in a fine aggregate mixture can improve the load resistance effect of the material by 39% when using

20 wt.% plastic wastes. Plastic waste-containing concrete exhibits high-energy absorption properties (Mustafa et al., 2019).

5.4 Polymer, waxes, and packaging

In 2015, approximately 20 MT of plastic packaging was used in Europe (Plastic Europe, 2016), generating approximately 30 kg of packaging waste per inhabitant per year (Exlaind, n.d.). In 2018, the EU adopted a plastic recycling strategy to achieve a circular economy. The main goal of this strategy is to recycle all plastic packaging waste by 2030 (Solis and Silveira, 2020). It is estimated that in 2017, approximately 42% of packaging waste was recycled in the EU, and 14%–18% was recycled worldwide (European Commission, 2018; Exlaind, n.d.). Currently, only a small fraction of segregated plastic waste is mechanically recycled, and the synthesis of low-quality recycled polymers is often more expensive than that of virgin plastics (Solis and Silveira, 2020).

Labor-intensive segregation is essential before mechanical recycling of plastic waste. This can damage polymers over their lifetimes (Ragaert et al., 2017). According to Förpacknings-och tidningsinsamlingen (FTI, 2019), plastic packaging recycling can be performed up to seven times before the plastic polymers deteriorate and become unusable (Solis and Silveira, 2020). Chemical recycling is considered promising for producing high-quality recycled polymers and supports achieving a circular economy (Solis and Silveira, 2020; Thunman et al., 2019).

Recycling PET polymers is cumbersome because of their relatively high inertness, high temperature resistance, and the underdeveloped food contact grade PET recycling technology (Welle, 2011). Recycling and further use of polyolefin packaging polymers require improved cleaning and assessment procedures. Polyolefin packaging polymers show higher diffusion and contaminant sorption rates than PET (Palkopoulou et al., 2016). Therefore, the acceptable limits of contaminants in recycled polymers are lower in polyolefins than in PET. The

European Food Safety Authority has implemented and issued 79 scientific statements evaluating over 130 procedures for reusing recycled plastics in food contact materials. Among these, 95% were related to PET recycling, and the remaining 5% discussed the recycling of polyolefins.

In addition, the US Food and Drug Administration (FDA) released 206 promising ideas as no-objection letters regarding the use of post-consumer recycled plastics in food containers. Ideas 153, 22, and 21 deal with the recycling procedures for PET, PS, and polyolefin, respectively. Six protocols deal with issues and practical barriers associated with PET recycling. Four are related to recycling plastic shopping bags, PS and PP, polycarbonates, and PE-phthalate resins.

5.5 Adsorbents of contaminants

There is a huge demand to use carbon-based materials for adsorbing contaminants, especially those derived from cheap sources such as waste materials and by-products of industries to boost the circular economy (Zhang et al., 2020). Plastic waste is converted to AC through a variety of physical and chemical treatments with numerous applications, including the adsorption of heavy metals, gases, and other harmful environmental contaminants (Adibfar et al., 2014; Al Lafi et al., 2017; Yu et al., 2019) (Table 5). The different adsorption capacities of AC are defined by surface area, pore volume, pore size, and the presence of functional groups on the pore surfaces (Mendoza-Carrasco et al., 2016).

An oxygen-enriched porous carbon adsorbent obtained by carbonizing PET waste effectively adsorbed CO_2 (1.31 mmol g^{-1}) at a CO_2 concentration of 12.5% (Kaur et al., 2019). It has also been confirmed that chemically-AC nanoporous materials derived from a suitable precursor PET tend to adsorb N_2, methane (CH_4), CO_2, and natural gas (Adibfar et al., 2014; Kaur et al., 2019; Sanal., et al 2017). PET from plastic bottles has been utilized as a carbon precursor to develop new CF_4 adsorbents through carbonization and activation with potassium hydroxide

(KOH) (Yuan et al., 2020). CNTs synthesized from PE bottles helped remove diuron pesticides from water (Deokar et al., 2017).

Toxic heavy metals, such as Cd(II), Pb(II), and mercury (Hg(II)), are potentially adsorbed from aqueous solutions by PS and PE after chemical treatment via nitration and sulfonation in the former and potassium permanganate ($KMnO_4$) in the latter (Al Lafi et al., 2017; Bekri-Abbes et al., 2006; Donbebe and Dixon, 2014; Mahmoud et al., 2016). A hydrogel composite synthesized from linear low-density polyethylene-g-poly (acrylic acid)-co-starch/organo-montmorillonite can adsorb Pb(II) (430 mg g^{-1}) from aqueous solutions (Irani et al., 2015). Radiation-induced grafting of polymer-based adsorbents from PP effectively adsorbed copper (Cu^{2+}) from aqueous solutions with a maximum adsorption capacity of 208.3 mg g^{-1} (Hassan et al., 2017). AC prepared from PET after chemical activation with KOH succeeded in adsorbing p-nitrophenol and iron (Fe(III)) with adsorption capacities of 659 and 74 m^2 g^{-1}, respectively (Mendoza-Carrasco et al., 2016).

Carbonization of mixed waste plastics (PP, PE, and PS) on organically-modified montmorillonite followed by KOH activation resulted in the formation of porous carbon nanosheets (PCNS), which effectively adsorbed methylene blue (MB) with a 95% adsorption capacity (Gong et al., 2015). Radiation-induced grafting formed covalent bonds of polymer chains of polyacrylic acid to polyurethane (PU) foam, resulting in the adsorption of monovalent azo dyes (basic red, BR29) (Goel et al., 2013). Glass fiber-reinforced plastic (GFRP)/clay ceramic, PET converted to graphene, and pyrolyzed PP effectively adsorb MB and acid blue 25 (AB25) (El Essawy et al., 2017; Ma et al., 2020a; Yasui et al., 2019).

Conversion of PS plastic waste char to carbon-metal double-layer oxide (C/MnCuAl-LDOs) nano-adsorbents with high surface area (60.43 m^2 g^{-1}) adsorbed Congo red (CR) dye with an adsorption capacity of 317.2 mg g^{-1} (Miandad et al., 2019). The preparation of porous superhydrophobic foams from PS using a high internal phase Pickering emulsion (HIPPE) for

oil spill cleanup exhibited sufficient adsorption capacity (20.4–58.1 g g^{-1}) and was easy to prepare and recycle (Yu et al., 2019). Furthermore, porous granules modified with cross-linked polyethylenimine (PEI) also adsorbed humic acid (Wang et al., 2008).

The various mechanisms involved in the adsorption of hydrophobic organic pollutants and metals to plastics include hydrophobic interactions, hydrogen bonds, electrostatic interactions, π-π-, van der Waals interactions, the formation of various surface functional groups, and the attrition and adsorption-precipitation of different charged minerals and organic matter (Holmes et al., 2014; Hüffer and Hofmann, 2016; Xu et al., 2018). In contrast, in the adsorption of hydrophilic organic compounds, such as PFASs, the dominant interaction is partitioning rather than hydrophobicity (Wang et al., 2015). Surface morphology, particle size, functional groups, and pH play essential roles in the adsorption capacity of plastics (Kedzierski et al., 2018; Li et al., 2018; Llorca et al., 2018). Factors such as temperature, particle size, and reaction time that affect the surface tension and solubility can also significantly affect adsorption capacity (Zhan et al., 2016). The adsorption behavior of plastics remains ambiguous because of the use of virgin plastics in experimental studies that do not represent the actual adsorption mechanism in real MPs found in nature (Ateia et al., 2020).

166 **Table 5.** Summary of selected literature showing the removal of contaminants (organic and inorganic) using plastic-derived products

Plastic polymer	**Modification process**	**Product name and characteristics**	**Contaminant name, removal capacity, and mechanisms**	**Reference**
PET	Physical activation with steam and chemical activation with KOH	Activated carbon; BET surface area = 1235 m^2/ g with steam and 1002 m^2/g with KOH; pore volumes 0.24–0.27 cm^3 g^{-1}	*p*-nitrophenol and/or Fe(III); adsorption capacity: PNP = 659 mg g^{-1} and Fe(III) = 74 mg g^{-1}; electrostatic and π–π interactions; pseudo-first and pseudo-second order kinetic models	(Mendoza-Carrasco et al., 2016)
PS	High internal phase Pickering emulsion (HIPPE)	SiO2@PS foam; density = 0.036–0.182 g cm^{-3}; porosity = 83.6–96.9%	Oil spill; adsorption capacity = 20.4–58.1 g g^{-1}; capillary force and van der Waals forces; n.d.	(Yu et al., 2019)
PE	Chemical activation of PE by $KMnO_4$	PE/MnO_2 complex; n.d.	Pb^{2+}; adsorption capacity = 50.5 mg/g; intra-particle diffusion	(Al Lafi et al., 2017)

			and film diffusion; Langmuir and Freundlich model	
LDPE	Combining acrylic acid, starch, organo-montmorillonite, and LDPE	Linear low-density polyethylene-g-poly (acrylic 2 acid)-co-starch/organo-montmorillonite hydrogel composite (LLDPE-g-PAA-co-starch/OMMT)	Pb(II); removal capacity = 430 mg/g; interactions between the OH groups of clay (OMMT) and the functional groups (OH, COOH and –COO−) of LLDPE-g-PAA-co-starch; Freundlich and Langmuir model	(Irani et al., 2015)
PVC	Unmodified	n.d.	Pb^{2+}; adsorption capacity = 1.0 mol g^{-1}; electrostatic attraction; Freundlich model;	(Donbebe and Dixon, 2014)
PP	Pyrolysis of PP along with SS (information not provided about SS)	Activated carbon; BET surface area = 1916.1 m^2 g^{-1}; total pore volume = 1.12 cm^3 g^{-1}; Iodine number = 1185–1460 mg g^{-1}	Methylene blue (MB); adsorption capacity = 476.88 mg/g; Freundlich, Langmuir, Redlich–Peterson, and Dubinin–	(Ma et al., 2020a)

			Radushkevich, pseudo-I and II-order	
PET	Carbonization and chemical activation by KOH	PET-K (2)700; BET surface area = 459–736 m^2 g^{-1}; total pore volume = 0.295 cm^3/g; micropore volume = 0.267 cm^3 g^{-1}	CF_4; adsorption capacity = 2.43 mmol g^{-1}; Physisorption and intra-particle diffusion; Langmuir isotherm model and pseudo-second-order kinetic model	(Yuan et al., 2020)
PET	2 g of raw PET waste heated to 800°C in an electric furnace at a rate of 8°C min^{-1} to form graphene	Graphene; BET surface area = 721.7 m^2 g^{-1}	Methylene blue (MB) and acid blue 25 (AB25); Adsorption capacity: MB = 481 mg $g^{-1}$and AB25 = 460 mg g^{-1}; Pseudo-second-order model and Langmuir and Freundlich isotherm models	(El Essawy et al., 2017)

Polyamide (PA)	Glass fiber-reinforced plastic (GFRP)/clay ceramics, was produced by mixing crushed GFRP with clay and firing the resulting mixture at 1073K	GFRP/clay ceramics; Apparent porosity = 38.2–66.2%, specific surface area = 2.83–14.9 $m^2 g^{-1}$, carbon content = 0.24–1.12%;	MB; reduction rate of MB > 80%; n.d.	(Yasui et al., 2019)
PS	Co-precipitation. Pyrolysis of PS followed by chemical activation (nitration and sulfonation) and immersion in solution $MnSO_4.4H_2O$, 0.078 g $CuSO_4$ $5H_2O$ and 2.6 g $Al(NO_3).9H_2O$	Carbon-metal double layered oxides (C/MnCuAl-LDOs); BET surface area = 60.43 $m^2 g^{-1}$; pore volume = 0.002 7 cm^3/g; pore size = 99.85 Å	Congo red; adsorption capacity = 317.2 mg g^{-1}; hydrogen bonding and hydrophobic, electrostatic interactions; Langmuir and Freundlich model	(Miandad et al., 2018)
PU	Covalently linked polymer chains of poly(acrylic acid) to PU foam using ^{60}Co-gamma radiation	Poly (acrylic acid) grafted polyurethane (PAA-g-PU)	Monovalent azo dye (basic red, BR29); Uptake capacity = 220mg/g	(Goel et al., 2013)
PS	Sulfonation by 96% H_2SO_4. Sulfonated under moderate agitation at 60 °C for 1.5 h and a dimension of particles of (0.5–1 mm^2).	resin; Cation-exchange capacity = 0.8 meq $100g^{-1}$of clay	Pb(II) and Cd(II); Sorption capacity of Pb = 0.29 mmol g^{-1} and Cd = 0.6 mmol g^{-1}; removal	(Bekri-Abbes et al., 2006)

			percentage = > 70%; Diffusion and exchange reaction; Langmuir and Freundlich models	
PP, PE, and PS	Carbonization of PP, PE, and PS with organically-modified montmorillonite (OMMT) formed carbon nanosheets (CNS). Chemically activated (KOH) CNS at 850 °C for 1.5 h under Ar atmosphere formed porous carbon nanosheets (PCNS)	PCNS; BET surface area = 2315 $m^2 g^{-1}$; pore volume = 3.319 $cm^3 g^{-1}$; purity = > 99.6%	Methylene blue; Adsorption capacity = 769.2 mg g^{-1}; Pore filling, hydrogen bonding, and π–π and electrostatic interactions; linear regression equation	(Gong et al., 2015)
Styrofoam (PS)	Nitration and sulfonation with nitric and sulfuric acid	Nitrated Styrofoam (Nit-Sfm) and sulfonated Styrofoam (Sulf-Sfm); CEC of Nit-Sfm = 8.2 and CEC of Sulf-Sfm = 10.6 mmol g^{-1}	Cd(II), Pb(II), and Hg(II); Removal capacity Hg(II) = 1950−2450, Pb (II) = 750−1100 and Cd (II) = 450−600 μm g^{-1}; Ion-pair interaction and cation-exchange mechanism	(Mahmoud et al., 2016)

PP	Bulk grafting of acrylonitrile (AN) onto PP waste using gamma radiation forming PPw-g-AN. Reflux with hydroxylamine hydrochloride changing cyano groups to amidoxime groups.	Amidoximated adsorbent; n.d.	Cu^{+2}; adsorption capacity = 208.8mg g^{-1}; chemisorption; pseudo-first-order and pseudo-second-order, Langmuir, and Freundlich models	(Hassan et al., 2017)
PS	Prepared via a dissolution/precipitation and etching/oxidation method; surface modification with cross-linked polyethylenimine (PEI).	Porous PS granules; Size = 1.3–2.4 mm	Humic acid; adsorption capacity = 4.5 mg g^{-1}; diffusion; Freundlich isotherm	(Wang et al., 2008)
PET	Chemical activation of carbon treated with KOH and $ZnCl_2$	Activated carbon; surface area = 682–1338 m^2 g^{-1}; total pore volume = 0.47–0.79 mL g^{-1}; micropore volume = 0.34–0.61 mL g^{-1}; mesopore volume = 0.71–0.81 mL g^{-1}; Iodine number = 821–984 mg I_2 /g C	N_2, CH_4, and CO_2, highest yield percentage = 31%; Freundlich and Langmuir models	(Adibfar et al., 2014)

PET	Carbonization at 500–800 °C and treatment with KOH	O-enriched porous carbon; BET surface area = 1690 m^2 g^{-1}; micropore volume = 0.78 cm^3 g^{-1}	CO_2, 1.31 mmol g^{-1} at 30 °C; Physical adsorption; Langmuir, Temkin, and Freundlich models	(Kaur et al., 2019)
PE	Synthesis of carbon nanotubes using Ni/Mo/MgO catalyst by combustion in a muffle furnace at 800 °C for 10 min.	Carbon nanotube (CNT); BET Surface area = 217 m^2 g^{-1}, mesopore volume = 0.358 cm^3 g^{-1}, and micropore volume = 0.064 cm^3 g^{-1}	Diuron, 40.37 mg g^{-1} at 303 K and bulk diffusion, film diffusion, intra-particle diffusion, and adsorption. Pseudo-second-order model	(Deokar et al., 2017)

5.6 Biotechnological upcycling

The persistence of plastics in the environment owing to the absence or low activity of catabolic enzymes has caused havoc in marine and terrestrial life (Chae and An, 2018; Thiel et al., 2018). At the same time, to meets its carbon demand, the chemical industry depletes crude oil and fossil fuel resources to produce 90% of its products. Addressing these challenges demands a transition from conventional to non-conventional carbon and energy sources, where plastics provide significant scope and input to the circular economy. Non-conventional raw materials such as plastics have high theoretical values of Cmol (mole of carbon) metabolite per Cmol raw material, and their mixtures show even more pronounced results (Blank et al., 2020; Liu et al., 2020).

Because of the single-use of plastics and the inefficiency of recycling methods, 3.5 billion tons of carbon in 80% of plastics produced globally remains unused (Geyer et al., 2017). Exceptionally high-performance fiber-reinforced plastic derived from the combination of deconstructed PET (reclaimed PET) with renewable and available monomers provides the option of upcycling PETs, which are the largest produced polyesters (Rorrer et al., 2019). Similarly, the pyrolysis products of PS, PET, and PE act as selective feedstocks for the bacterial production of biodegradable polyhydroxyalkanoate (PHA) (Goff et al., 2007; Guzik et al., 2014; Kenny et al., 2012). For instance, the conversion of PS to PHA using styrene oil, an intermediate pyrolysis product, can be achieved by *Pseudomonas putida* CA-3 (Ward et al., 2006). Therefore, coupling thermochemical degradation with biological upcycling processes presents a method of utilizing carbon sources in plastic wastes for a sustainable circular economy (Wierckx et al., 2019).

Enzyme-mediated hydrolysis of plastics plays an important role in biodegradation, depending on the chemical bonds present in the polymer. Microbial enzymes such as laccases, manganese peroxidase, lignin peroxidase, hydroquinone peroxidase, urease, esterase, protease,

carboxylesterase, lipase, and cutinase modify or even completely degrade PE, PS, PUR, and PET (Wei and Zimmermann, 2017). Mixed microbial cultures have shown more promising results than monocultures for degrading PS and PE (Mukherjee et al., 2016). Aromatic plastics, such as PET or polybutylene terephthalate (PBT), have relatively excellent properties but are more resistant to microbial degradation (Müller et al., 2001). Several putative xenobiotic biodegrading genes, including homogentisate 1,2-dioxygenase, N-ethylmaleimide reductase, cytochrome P450, and 2,4-dichlorophenol 6-monooxygenase, are abundant in plastic-attached microbes in the North Pacific Gyre, and are likely involved in plastic degradation (Bryant et al., 2016).

A novel bacterium, *Ideonella sakaiensis* 201-F6, was found to hydrolyze PET into environmentally safe monomers, terephthalic acid and ethylene glycol, using two enzymes, PETase and MHETase (Yoshida et al., 2016). The limited capacity of this bacteria to degrade PET owing to the requirement of high incubation temperatures for enzymatic hydrolysis can be overcome by rational protein engineering (Song et al., 2019b). Advanced understanding of microbial metabolic pathways and genome editing techniques, such as clustered regularly interspaced short palindromic repeats (CRISPR)-associated protein Cas9 systems, can help overcome major challenges related to biotechnological upcycling of plastic wastes (Blank et al., 2020).

6. Socioeconomic considerations

Economically sound, socially acceptable, and environmentally sustainable plastic waste management and recovery are key challenges to achieving the UN SDGs by 2030 (Fig. 3). The major constraints on plastic waste management and recycling are (i) lack of legislation on plastic collection and recycling, (ii) adverse effects of plastics at societal levels, (iii) lack of environmentally suitable design of plastic dumping, (iv) high input and labor costs, (v) burden

from importing plastic waste to low-infrastructure countries, (vi) demand for quality recycled materials, and (vii) requirement of technology upgradation for quality control (Mwanza and Mbohwa, 2017).

Life cycle assessment is a suitable technique for investigating environmentally sustainable waste management solutions (Aryan et al., 2019; Goulart Coelho and Lange, 2018). Studies on the recycling of two types of plastic wastes, namely, PET and PE in the city of Dhanbad, India, showed that the replacement of virgin PET and PE flakes by recycled PET and PE flakes and the reduced emissions during the recycling process of these two plastic wastes resulted in a low impact on the environment (Aryan et al., 2019). This indicates a potential policy plan for improved management of plastic waste. In addition, mechanical separation of plastics from a mixed fraction of waste prior to incineration yielded the maximum amount of specific polymers for recycling, suggesting the most impactful scenario for environmentally sound management of plastic waste (Rigamonti et al., 2014).

Although there are socio-environmental benefits of plastic reuse, global dependence on single-use consumer products has led to environmental risks because of the accumulation of large quantities of plastics in the oceans and rivers. Jambeck et al. (2015) estimated that without effective intervention, the ocean plastic mass will likely exceed fish mass by 2050. Therefore, significant changes in socially acceptable policy prescriptions must be designed by policymakers to reduce plastic consumption and production, which is affected by the perception of stakeholders (McNicholas and Cotton, 2019). The study suggested that public awareness and education, more clearly defined waste responsibilities, and changes in the consumer behavior environment must be considered when formulating government policies.

To ensure economically sound plastic waste management practices, it is necessary to focus on recycling used plastic materials through innovative recycling technologies and green infrastructure to expanding the investment in recycling (Jang et al., 2020). The available plastic

waste management options are source reduction, reuse, and recycling, as well as energy recovery from waste-to-energy (incineration) and fuel production. Currently, instead of disposal, attention is focused on converting waste into value-added products such as liquid fuels, biodegradable polymers, and low-investment green polymers. However, the potential of green polymers to replace non-biodegradable plastics is very limited (Banu et al., 2020; Panda et al., 2010). Integrating the present system with the production of petrochemical products, such as oil and gasoline, by thermochemical treatments (catalyst-mediated pyrolysis and hydrocracking), improves the economic condition of waste management systems. This will secure large amounts of non-renewable resources (coal, petroleum, gas), reduce greenhouse gas emissions into the environment, bring high-quality products to the market rather than low-quality materials, and reduce dumping of waste for landfills (Mastellone, 2020). However, when developing a policy for sustainable plastic waste management, the following issues need to be addressed:

(1) Enforcement of robust legislation on plastic reuse and recycling,

(2) Additional investment in plastic waste recycling industries,

(3) Focus on generating value-added products through a waste-to-wealth program,

(4) Establishing environmentally suitable structures for plastic dumping,

(5) Raising public awareness through education, conversation, social campaigns on plastic use, and proper waste disposal (e.g., red containers for non-degradable waste; green containers for degradable waste) to separate non-degradable and biodegradable plastics at the municipal or community levels, and

(6) Producing high-quality materials through recycling and using more biodegradable products (from cloth, leaves, and jute) as a replacement for plastics.

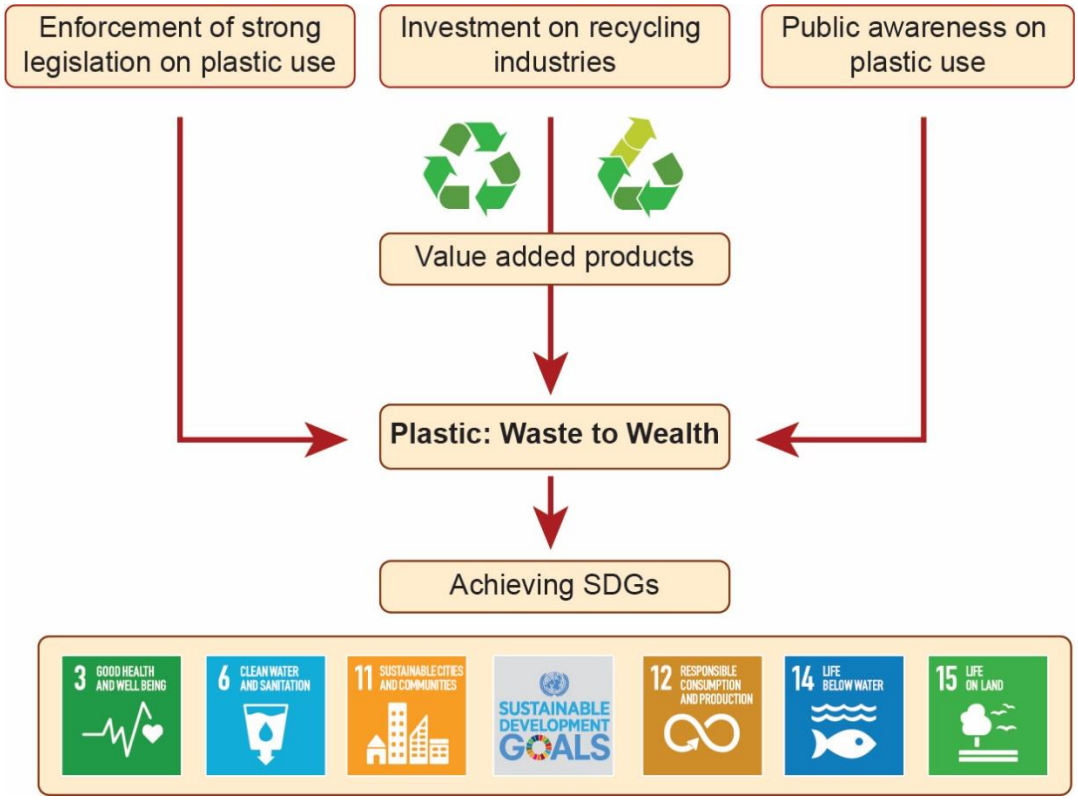

Fig. 3. Schematic diagram showing the achievement of the Sustainable Development Goals by managing plastic wastes in the environment.

7. Conclusions and future research

The accumulation of MNPs in the environment owing to plastic waste mismanagement has become an undeniable problem worldwide. This may result in tremendous adverse effects on environmental sustainability and human health. MNP pollution can be tackled through various means, such as reducing plastic manufacturing and consumption, including plastics in the circular economy, and appropriate waste management. Biodegradable plastics are considered a suitable alternative to reduce plastic consumption. However, biodegradability can vary depending on the conditions. Recycling, reuse, and valorization of waste plastics have attracted significant attention. New approaches such as conversion of waste to fuels, chemicals, value-added carbon materials (adsorbents and solid catalysts), and road building and construction

materials are promising techniques for plastic waste valorization. However, they are still limited by technological advancements for large-scale industrial applications.

Moreover, not all plastic products are suitable for recycling. Therefore, further studies are required to develop cost-effective alternatives for plastics and environmentally friendly and economically sound plastic waste valorization methods. In addition, the hazards and fate of primary MNPs need to be evaluated, and appropriate standards need to be implemented for environmentally safe limits. Although some countries have implemented legislation to reduce plastic consumption and enhance the reuse of plastics, advocating these approaches for all types of plastics is challenging. Moreover, the ability to formulate these policies depends on geographic region and socioeconomic status. Therefore, stringent environmental regulations and the development of appropriate infrastructure and economically sound, environmentally sustainable, and socially acceptable plastic waste management strategies are critical to reduce the threat of MNPs in the environment.

Declaration of Conflicts of Interest

The authors declare that they have no known competing financial interests or personal relationships that could appear to influence the work reported in this manuscript.

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