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Research paper

Microplastics in multimedia environment: A systematic review on its fate, transport, quantification, health risk, and remedial measures

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HIGHLIGHTS

- Primary and secondary microplastic sources and their impacts on environment.
- Effect of microplastic on soil, sediment and different aquatic ecosystems.
- Diverse sources and routes of micro-plastics destined for human consumption.
- Microplastic recovery from WWTP and removal by conventional and advanced methods.
- Bottleneck, current challenges, and future outlook of microplastic pollution.

GRAPHICAL ABSTRACT

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ABSTRACT

The ever-increasing presence of microplastics in many environmental components has been a cause of worry for humanity due to their small size and potential health risk. Since the last decade, numerous studies have been conducted on the prevalence and dispersion of microplastics. However, at present, there aren't any systematic studies on fate and transport of microplastics that consider multimedia environmental systems and their mitigation measures. Also, there are limited studies on the routes through which humans are exposed to microplastics. In this review, about 380 articles were evaluated to uncover the extent of microplastic fate, transport, and pollution in different environmental components, including soil, freshwater, marine, and atmosphere, as well as its effect on different ecosystems. We gave special attention to understanding many routes and sources of microplastics intended for human consumption and their consequences on human health. Furthermore, we tried to emphasize on the different methods used for sampling, extraction, identification and characterization of microplastics, along with associated benefits and limitations. This study highlighted existing knowledge and gaps in the remediation of microplastics. On this basis, the bottleneck and current challenges have been proposed.

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1. Introduction

Plastics are long-chain polymers composed of carbon, oxygen, hydrogen, silicon, and chloride, and are made from natural gas, oil, and coal (Shah et al., 2008). Currently, plastics have been used as an excellent material in today's day-to-day life. They are used in almost all applications such as packaging, automotive, aquaculture, fisheries, biomedical, shipping, agriculture, building and construction, telecommunications, furniture, transportation, plumbing, personal care products, textiles, clothing, etc. (Ogunola et al., 2018). Plastics have even replaced more conventional materials such as glass and metals because of their lightweight nature, malleability, durability, flexibility, low cost, persistency, thermal and electrical insulation, corrosion resistance, high strength-to-weight ratio, and waterproof properties (Pellini et al., 2018; Zhang et al., 2021a; Ram and Kumar, 2020). The global plastic production reached 359 million tonnes in 2018, an increase of 46.5% compared to 2008 and 3.2% compared to 2017 (Plastics Europe 2019; Mao et al., 2020). Among all the countries, China generates the most (30%), followed by Canada, Mexico, USA (18%), and Europe (17%) (Tiwari et al., 2020). However, plastics are emerging, persistent, and ubiquitous contaminants that could harm the growth and development of organisms, induce oxidative stress, weaken the immune system, reduce lifespan, and impact fertility (Chen et al., 2020a).

'Microplastics' are formed when plastics degrade or break down into smaller fractions under physical, chemical, mechanical, and biological actions (Plastics Europe 2019; Lestari et al., 2020). These plastics are microscopic and pervasive particles and they have been continuously increasing in the environment due to their continuous production, non-biodegradable, persistent, and long-life span in the environment (Chamas et al., 2020). Thus, they have been declared as one of the ten emerging contaminants in the United Nation Environmental Programme (UNEP) Year Book 2014 that could potentially threaten human health and other organisms in all biomes (Constant et al., 2020). Therefore, the accumulation of microplastics in environmental components is gaining attention and becoming a major concern among global researchers and scientists. The abundance of microplastics in lakes, rivers, estuaries, oceans, and beaches worldwide has been documented in highly populated areas or areas with intensive anthropogenic activities (He et al., 2020). Because of the small size, microplastics can enter the human food chain through the consumption of seafood as well as other terrestrial food items, and subsequently can have impact on human health (Bondelind et al., 2020; Rist et al., 2018; Chatterjee and Sharma, 2017; Barboza and Gimenez, 2015). Secondly, plastic waste disposal in municipal waste disposal systems produces poisonous leachate, which can contaminate water and soil (Rajmohan et al., 2019; Kataria et al., 2022). Unprecedented use of plastic products and improper waste management techniques will continue to increase plastic waste (Geyer et al., 2017). The irresponsible behaviour of people regarding the use of plastics, dumping plastic products, improper management systems, and associated harmful impacts have turned the planet into a 'plastic planet' (Chatterjee and Sharma, 2019).

Despite so many studies on different aspects of microplastics pollution, its effect on human health has recently come into the picture. Humans consume seafood in the form of fish and shellfish, marine molluscs, oyesters, shrimp, mussels etc. These marine species are contaminated with microplastics thus, affecting human health. However, there is no direct evidence of human health risk due to ingestion of microplastics (Rahman et al., 2020). Also, there are not many studies on the effects of microplastics on human health, and the knowledge regarding the harmful impacts of microplastics on human health and the routes through which humans are exposed are still in their infancy stages. Moreover, plastics release fatal organic pollutants like dioxins and bisphenol A, which can cause cancer and other neurological damages, including impairment of the reproductive system (Rajmohan et al., 2019; Kataria et al., 2022). Thus, microplastics pollution is a great threat to the environment and all living beings. A proper understanding of this aspect is crucially needed.

Therefore, the present study provides up-to-date and comprehensive information on the prevalence of the microplastics in various environment matrices, including fresh water, atmosphere, soil, marine and food chain. The major emphasis was on various sources of microplastics and their routes to humans along with associated possible health risks. In addition, detailed information on microplastics sampling and quantification techniques as well as updates on various microplastics recovery or removal techniques were covered in order to fully understand the impact of microplastics in soil and aquatic ecosystems. This study was concluded with mitigation strategies, current challenges, and future perspectives of microplastics pollution.

1.1. *Type of plastics*

Different types of commercially available synthetic plastic materials and their uses are shown in Table 1. Polymers such as polyethene (PE), polypropylene (PP), polystyrene (PS), polyvinyl chloride (PVC), polyurethanes, polyethylene terephthalate (PET), polystyrene (PS), and polyester (PES) are examples of "virgin plastics" (Plastics Europe, 2018; Andrady and Neal 2009). They represent 90% of the total plastic produced worldwide (Espinosa et al., 2016). The polymers such as PET, PP, PE, PES, PVC, PS, Polyamide (PA), and nylon are commonly used to manufacture various plastic items and process food products (Sutton and Sedlak, 2017). The most popular plastic polymers used in packaging are PP, PET, low- and high-density polyethene (LDPE and HDPE) (Plastics Europe, 2018). PET is a potential human carcinogen, but it is still used abundantly in producing packaging material, drinking water bottles, pipes, insulation moulding, plastic films, etc. (Karbalaei et al., 2018; Kataria et al., 2022). Polystyrene, which is primarily used in manufacturing of bottles and lids, containers, and protective packaging, has been found to cause chronic toxicity because it translocates in blood and interferes with reproductive disruption processes in marine filter feeders (Sun et al., 2019). Phthalates and Polybrominated Diphenyl ethers used to improve the fire resistance and plasticity of the plastics are also well-known endocrine-disrupting compounds (EDCs). They have also been found in human bodies due to bioaccumulation (Sun et al., 2019).

1.2. *Type of microplastics*

Microplastics are synthetic, long-chain, and organic polymers that can be found in various ecosystems such as soils, subsurface systems, rivers, lakes, wetlands, oceans, and atmosphere (Kumar et al., 2021a). These polymers come in a wide range of particle sizes and densities that can be harmful to aquatic ecosystems, animals, and human beings (Razeghi et al., 2021). Microplastics are typically defined as plastic particles with at least one dimension under 5 mm (Au, 2017; Rillig et al., 2017) or as any polymer with the largest dimension smaller than 5 mm or within a size smaller than 5 mm (Eerkes-Medrano et al., 2015; Anderson et al., 2016; Lestari et al., 2020). Microplastics can be large microplastic particles (L-MPP, 1–5 mm) or small microplastic particles (S-MPP, $<$1 mm), as well as microbeads, fragments, fibres, pellets, flakes, sheets, or foams. Plastics are also classified into different categories based on their size viz. giant ($>$1 m), large ($<$1 m), medium ($<$2.5 cm), micro ($<$5 mm), and nano ($<$0.1 μm) (Elgarahy et al., 2021). They are derived from natural and organic materials such as coal, natural gas, and crude oil by polymerization or polycondensation processes (Phuong et al., 2016).

Microplastics are classified into two types based on their source: primary and secondary. Primary microplastics are small plastic particles released directly into the environment by domestic and industrial effluents, spills, and sewage discharge or indirectly by runoff. For example, scrubbing agents used in cosmetics and biomedical uses, plastic pellets accidently lost during production or handling (OECD, 2022). They are also manufactured as microbeads in industries and used

Table 1
Common thermoplastic resin types, associated resin codes, their predominant uses, and quantity of resin supply/production by weight in Fiscal Year 2020) in north America as reported by the American chemistry council by plastic type.

Plastic Polymer	Abbreviation	Resin Code	Uses	Million Metric Tonnes (MMT)
Polyethylene terephthalate	PET	1	Single-use beverage bottles, food containers, textiles, etc.	Data not available
High-density polyethylene	HDPE	2	Milk bottles, detergent bottles	10.4
Polyvinyl chloride	PVC	3	Window frames, profiles, floor and wall coverings, pipes, cable insulation, garden hoses, inflatable pools, etc	6.9
Low-density polyethylene	LDPE	4	Single-use plastic bags, reusable bags, trays and containers, agricultural film, food packaging film etc.	3.5
Linear low-density polyethylene	LLDPE	4	Single-use plastic bags, reusable bags, trays and containers, agricultural film, food packaging film, etc.	10.4
Polypropylene	PP	5	Food packaging, candy and snack wrappers, microwave containers/ dishware, pipes, automotive parts, non-woven textiles, personal protective equipment/masks, fishing gear and nets, etc.	7.8
Polystyrene	PS	6	Food packaging (e. g., cups, utensils), electrical and electronic equipment, etc.	1.6
Expanded polystyrene	EPS	6	Food packaging (to-go containers, coolers), building insulation, electrical and electronic equipment, inner liner for fridges, etc.	0.4
Thermoplastic polyurethane	TPU	7	Clothing (Spandex), home building, automotive, industrial products	0.1
TOTAL				41.1

Source: (National Academies of Sciences, Engineering, and Medicine, 2022)

in personal care products, sandblasting media, or raw materials for fabricating other products (Andrady, 2017; Schessl et al., 2019). The secondary microplastics are formed in the environment as a result of the breakdown of larger plastic particles via several degradation mechanisms such as chemical and physical ageing, UV radiation (photo-oxidation), mechanical transformation (via waves abrasion), and biological degradation by microorganisms (De Sá et al., 2018; Ogunola et al., 2018). They are further divided into two categories: 1) those formed during the use of products, such as from tyre abrasion and synthetic microfibers from clothing and other textile products; and 2) those formed by the degradation and fragmentation of macroplastics that have been released into the environment (OECD, 2022). One of the primary causes of the global increase in microplastic pollution is the difficulty in removing them from environmental matrices due to their tiny size and low visibility (Auta et al., 2017a).

2. Source of microplastics

Microplastics are heterogeneous substances with varying shapes, sizes, morphology, polymer compositions, and density (Duis and Coors, 2016; Auta et al., 2017a; Wang et al., 2019). Based on origin, microplastics are classified into two classes: primary and secondary.

2.1. Primary microplastics

The primary microplastic source mainly includes industrial production units and domestic activities that release micro-size plastic directly into the environment (Fig. 1) (Auta et al., 2017a). Primary microplastic levels in terrestrial soils and freshwater systems increase as a result of industrial processes such as the production of pharmaceuticals, the blasting of plastics (e.g., thermal cutting), tire wear (e.g., cars, planes), plastic pellets, cosmetics, insect repellent, wastewater treatment (hospital, industries), sewage sludge, etc. (Fendall and Sewell, 2009; Guo and Wang, 2019). However, domestic products or activities, including hand cleaners, nail polish, facial cleansers, shower/bath gels, scrubs, shaving cream, baby products, hair colours, insect repellents and sunscreen, toothpaste, etc., release microplastics into the environment via domestic sewage or wastewater treatment (Auta et al., 2017a). Primary microplastics are also produced during air spray or blasting during painting and polishing. Blasting acrylic, melamine, or polyester microplastic scrubbers and rust or paint remover hulls release a massive amount of microplastics into the environment.

2.2. Secondary microplastics

Secondary microplastics include fibers derived as a result of the deterioration of large plastics on land and in the sea. These materials are disintegrated continually and released as microplastics into the environment by the action of solar radiation (photodegradation), wind waves (mechanical), water waves (hydrolysis), acidic and/or alkali conditions (chemical degradation), and microorganism (biodegradation) (Huang et al., 2020a; Malankowska et al., 2021). Sources of secondary microplastics are abundant and heterogeneous (Horton et al., 2017). Several factors such as climatic conditions (temperature, sunlight, precipitation, etc.), plastic properties (size and density), and chemical structure influence the breakdown process of microplastic debris. Natural weathering is the primary disintegration process, which is accelerated by UV light exposure to plastic. UV radiation exposure promotes polymer structural oxidation, which leads to the breakdown of plastic intermolecular bonds (Andrady, 2011; Wagner et al., 2014). Physical factors such as wind abrasion, fluctuation, and turbulence also contribute to the production of microplastics (Barnes et al., 2009). In addition, parameters such as precipitation chemical composition, soil and water pH, and chemicals produced by microorganisms all have a role in the oxidative disintegration of large plastic particles (Law and Thompson, 2014; Shim and Thompson, 2015). The microorganisms are also significantly responsible for degrading large plastics into small microplastics by releasing some chemicals. In terrestrial ecosystems, these processes contribute to fragmentation, surface abrasion, and oxidative breakdown of microplastics. Therefore, it is not an easy task to identify the exact source of microplastics in different environmental matrices. The microplastics are diverse in the environment, and their degradation is dynamic (Zhou et al., 2020b). Low-density plastic is widely used in mulching and greenhouse to improve compost quality and crop production. These agricultural mulch films and compost formation processes contribute to microplastic accumulation in terrestrial soil (Huang et al., 2020b). Municipal solid waste and its landfill sites also

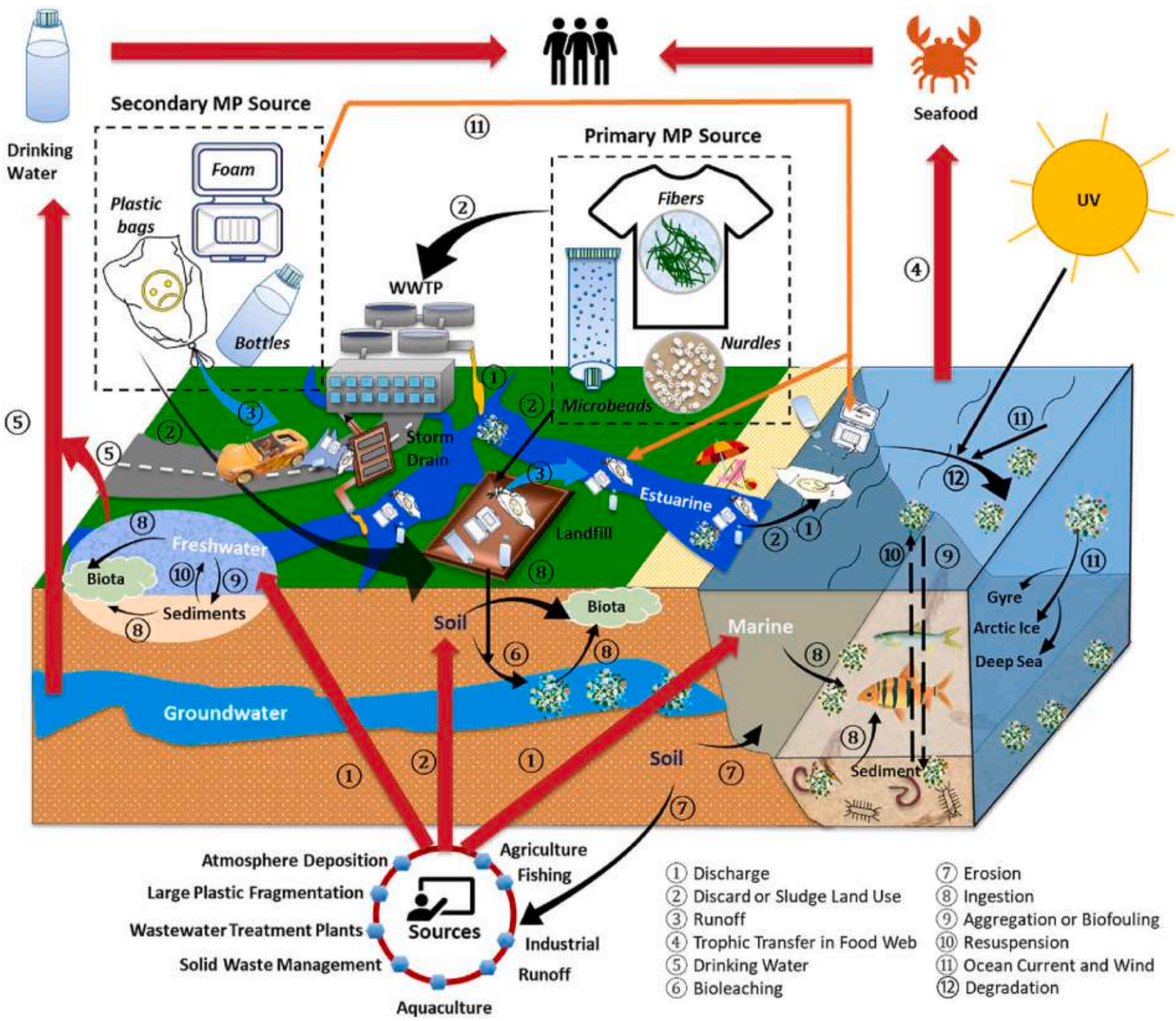

Fig. 1. Sources and transport of microplastics in various environmental systems **(Reproduced with permission from Ref. (**Wu et al., 2019**). Copyright 2019 Elsevier.**

act as secondary microplastic sources contaminating the soil and groundwater system (Zhou et al., 2020b). The presence of microplastics in the aquatic ecosystem cause health problems and toxic impacts on aquatic organisms. Their persistent nature enhances the leaching, accumulation, and adsorption in the environmental matrix (Shim and Thomposon, 2015).

2.3. *Fate and transport of microplastics in the environment*

Microplastics are transported across vast distances due to their shape, size, buoyancy, lightweight, and durability via wind and water. Several studies have been published in which WWTP effluent (wastewater treatment plants) has been identified as a key route for directly transporting microplastics into aquatic and soil environments (Karbalaei et al., 2018). Large-size microplastics and other particles can be retained in the mechanical screen and grit chamber during the early phases of the WWT process. During the primary treatment process, microplastics fibers settle down by gravity via the coagulation/flocculation mechanism (Zhou et al., 2021). In the secondary treatment process, the microplastics encounter microbes, which can affect microbial activity and are retained on activated sludge. Sludge generated by WWTPs could be another possible route for microplastic transport from WWTPs to the soil ecosystem (Eriksen et al., 2014; Auta et al., 2017a).

Plastic debris contributes significantly to secondary microplastics and transports microplastics in soil and marine ecosystems (Obbard et al., 2014). The plastic debris gradually breaks down into microplastic and accumulates on the soil's top surface. Microplastics can enter the soil subsurface via agricultural processes, precipitation, cracks, and soil organism activities (He et al., 2018). Because of these processes, microplastics may leach and percolate in deep soils with water, eventually entering into groundwater (Rillig et al., 2017). Moreover, soil organisms such as earthworms, insects, nematodes, bacteria, fungus, algae, etc., influence microplastics transport and fate via ingestion, redox, and excretion process (Wang et al., 2020c; Guo et al., 2020). Some studies reported that microplastics are also passed from the digestive system (gut) of soil organisms into the faeces and excreta. The research found that microplastics contained in faecal pellets might be an indirect source of microplastics in the marine ecosystem (Duis and Coors 2016; Cole et al., 2014). Surface runoff and wind flow carry microplastic residue from soil into freshwater streams and marine ecosystems (Cole et al., 2011). Microplastics of different morphologies are continuously transported and settle in soil or sediment after being released into the atmosphere, posing a problem to the human respiratory system. Microplastics can be found on beaches, seabed sediments, surface waters, and in a wide range of marine creatures, including sea birds, fish, bivalves, mammals, and crustaceans etc. (Auta et al., 2017a; Hou et al., 2021).

Few studies have reported microplastic presence in remote areas and the continent of the earth where human activity is limited (Fendall and Sewell, 2009; Guo and Wang, 2019). Microplastics can be transported via marine flow and atmospheric fall. Its small size and low density play a significant role in the global distribution of microplastics in every ecosystem. Thus, polar ice and marine sediments have become a global sink for microplastics (Obbard et al., 2014). It has been predicted that among 269 million tonnes of particles found globally, approximately 92% are microplastics. This prediction proved that most microplastics are located in marine sediments as a sink, while the amount of microplastics on the surface is less (Eriksen et al., 2014; Auta et al., 2017a). However, it is uncertain how much microplastics has entered and is being preserved in the marine ecosystem since the beginning of the anthropogenic era.

3. Occurrence of microplastics in different environmental components

The presence of microplastics has become a serious ecological concern in several environmental compartments on the Earth; even karst groundwater cannot avoid microplastic pollution (Panno et al., 2019). Also, scientific studies confirmed the existence of microplastics even in the deepest oceans, tallest mountains, and poles (Walkinshaw et al., 2020). Researchers have emphasized on the detection and distribution of microplastics in diverse ecosystems such as soils/land, freshwater, atmosphere, and seas/oceans in recent years (Fig. 2). In this regard, Table 2 summarizes some notable findings on the presence of microplastics fibre, fragments, and particles in different environmental matrices. Different detection methods such as ATR-FTIR, microscopy, NMR, Fluorescence, SEM, EDX etc. have played an important role in characterizing microplastics in different components of the environment (more details have been explained in section 4).

3.1. Microplastics in freshwater

Freshwater, which is the primary source of drinking water for people, is thought to be the potential source of microplastics exposure to humans (Novotna et al., 2019). Primary microplastics sources in freshwater are those of industrial origin and secondary microplastics, which result from the breakdown of large plastic debris (Eerkes-Medrano et al., 2015; Horton et al., 2017). The microplastics shape can also define the presence in different parts of river; for example, if microplastics are of fragment shape, they easily float and are dominantly found on the water surface, whereas, if microplastics are fibres and pellets, they settle at the bottom and are found in the sediment (Lestari et al., 2020). The Great Pacific garbage patch (also known as the Pacific trash vortex) is a massive accumulation of plastic on the surface waters of the central North Pacific Ocean. Plastic and floating trash come from the Pacific Rim, which includes Asia, North America, and South America. The Great Pacific Garbage Patch contains 79,000 tons of plastic with large particle sizes, from fishing lines, nets, hard plastics, and films, among other things (Rajmohan et al., 2019).

Several factors affect the migration or transport of microplastics in freshwater, including the water body size, wind currents, and particle density (Eriksen et al., 2013; Fischer et al., 2016; Free et al., 2014). Some investigations verified the existence of microplastics even in remote areas such as Antarctica (Cincinelli et al., 2017) and the Arctic (Lusher et al., 2015), central Atlantic Islands (Martins et al., 2020), Arabian Gulf (Abayomi et al., 2017) even in deep-sea Arctic sediments (Kanhai et al., 2019) although the studies are limited.

Several researchers have recently documented the presence of microplastic in freshwater, including lakes, ponds, kart water, rivers, and streams, as well as in sediments (Table 2). In Asia, especially in China and India, microplastics has been reported in surface water. In China, freshwater studies have mainly focused on the eastern regions of China, including the Yangtze estuary (Xu et al., 2018), Minjiang, Jiaojiang, and Oujiang estuary (Zhao et al., 2015), Futuanhe river estuary, and Sha river estuary (Zhou et al., 2018) and the Pearl River estuary (Fok and Cheung, 2015). Yuan et al. (2019) investigated the microplastics in surface water and sediment of Poyang Lake (China), and the average concentration and estimated particles were 0.2034 g/L and 226 particles/L, respectively. Ding et al. (2019) reported that the average quantity of microplastic and estimated particles were 0.92 g/L and 1020 particles/L, respectively, in the Wei River in China, which was much higher than other lakes or rivers in China. Similarly, some other researchers have also detected microplastics in lakes and rivers of China, i.e., 0.57 g/L in the sediment of Tibet plateau (Zhang et al., 2016), 123 particles/L in surface water and sediment of Taihu lake (Su et al., 2016), and 0.47 g/L in the sediment of Pearl river (Fok and Cheung, 2015). According to global modelling studies of world rivers, the Ganges river (India) is the second-largest source of microplastic in coastal seas and oceans (Lebreton et al., 2017). Sarkar et al. (2019) investigated the occurrence of different microplastics in the Ganga river (India)

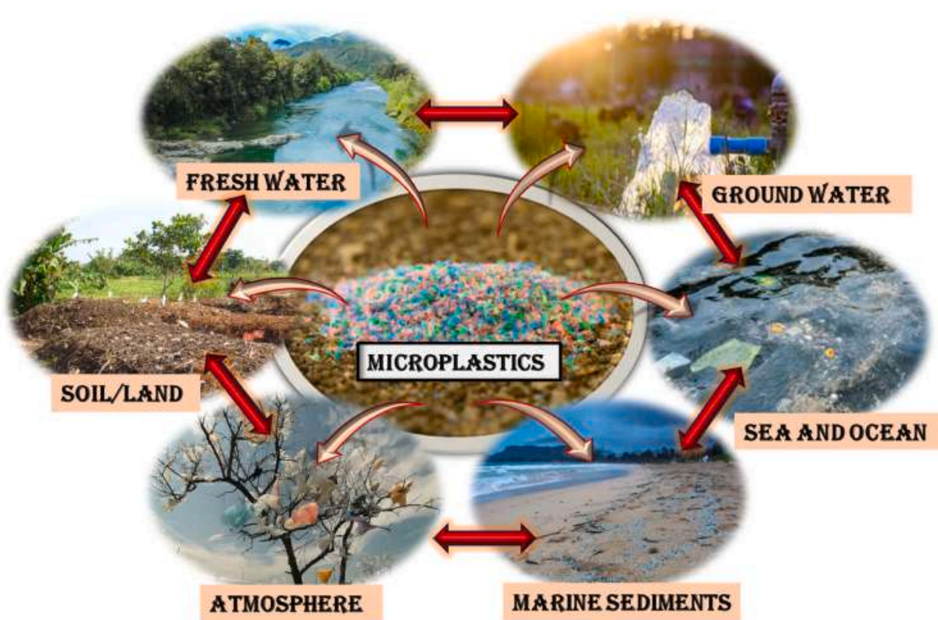

Fig. 2. Graphical representation indicates the transport and occurrence of microplastics in different environmental media.

Table 2
Summary of microplastic existence in different environmental components worldwide.

Sr No.	Location	Environmental components	Average Concentration/ abundance	Size/Range	Detection methods	References
Microplastics in water						
1	Yongding River, Tianjin, China	water Pipe scale	13.2–134.8 n/L 570 to 752 n/kg	>200 μm 50–100 μm	Stereomicroscope and FTIR imaging	Chu et al. (2022)
2	Alluvial aquifer, Shiraz, Iran	Ground water	0.1 to 1.3 MP/L	≤500 μm length	Binocular microscope	Esfandiari et al. (2022)
3	Alluvial sedimentary aquifer, Victoria, Australia	Ground water	38 MP/L	18–491 μm	Laser Direct Infrared (LDIR)	Samandra et al. (2022)
4	Waco Creek, Wilson's Creek and Proctor Springs	Water	0.98–3.38 particles per 800 mL water	53 μm	GC-MS	Stovall and Bratton (2022)
5	North-western part of Germany	Drinking water	0 to 7 MP/L	>20 μm	FTIR imaging	Mintenig et al. (2019)
6	Urban area, Czech Republic	Raw drinking Water	1473- 3605 particles/L	<10 μm	SEM and FTIR	Pivokonsky et al. (2018)
7	Eastern Indian Ocean (EIO) and Bay of Bengal (BoB), India	Sea water	0.34 ± 0.8 item/m^2 (EIO); 2.04 ± 2.3 items/m^2 (BoB)	<2 mm (EIO); <1 mm (BoB)	Microscope and μ-FTIR	Li et al. (2021)
8	Southeast Arabian Sea and southwest coast of Kerala (India)	Surface water	1.25 particles/m^3	>1 mm	FTIR and Raman spectroscopy	James et al. (2020)
9	Southern coast, Kerala (India)	coastal water	1.25 ± 0.8 particles/m^3	0.3–4.8 mm	Microscope and ATR-FTIR	Robin et al. (2020)
10	Tuticorin, Tamil Nadu (India)	Sea water Bore-well water	35 to 72 items/kg 2 to 29 items/kg	100–500 μm	SEM-EDAX, microscopic and ATR-FTIR	Sathish et al. (2020)
11	Red Hills Lake, Tamil Nadu (India)	Water	5.9 particles/L	<5 mm	FTIR and EDX	Gopinath et al. (2020)
12	Yarra River (Australia)	Surface water	2.58 g/L	<2 mm	Microscope	Kowalczyk et al. (2017)
13	Greater Paris	Wastewater	260-320 particles/100 m^3	100–500 μm	Microscope	Dris et al. (2015)
14	Lake Hovsgol (Mangolia)	Surface water	1.2 × 10^4 particles/km^3	<5 mm	Microscope	Free et al. (2014)
Microplastics in Soil and sediments						
1	Brahmaputra River and Indus River (India)	sediments	531-3485 MP/kg and 525-1752 MP/kg	20-150 μm	FTIR microscope	Tsering et al. (2021)
2	Brahmaputra River and Indus River (India)	Sediments	20-240 MP/kg and 60-340 MP/kg	<5 mm	FTIR microscope	Tsering et al. (2021)
3	Arabian Sea and coast of Kerala (India)	Sediments	10–70%	1–5 mm	FTIR and Raman spectroscopy	James et al. (2020)
4	Melbourne and Western Port (Australia)	Sediments	0.06–2.5 items/L and 0.9–298.1 items/kg	<1 mm and >40%	FTIR	Su et al. (2020)
5	Southern coast, Kerala (India)	Sediments	40.7 ± 33.2 particles/m^2	0.3–4.8 mm	Microscope and ATR-FTIR	Robin et al. (2020)
6	Nattika beach, Kerala (India)	Sediment	70-121 items/L	<5 mm	Microscope and ATR-FTIR spectroscopy	Ashwaini and Varghese (2020)
7	Red Hills Lake, Tamil Nadu (India)	Sediments	27 particles/kg	<5 mm	FTIR and EDX	Gopinath et al. (2020)
8	Mellipilla county, Chile	Agricultural soil	1.1–3.5 items/g dry soil	0.16–10 mm	Stereomicroscope	Corradini et al. (2019)
9	Rice-fish culture stations in Shanghai, China	Paddy soils	10.3 ± 2.2 items/kg	<1 mm	μ-FTIR	Lv et al. (2019a)
10	Suburbs of Shanghai, China	Farmland Soil	Shallow-78 items/kg Deep Soil-62.5	<1 mm	Stereomicroscope and FTIR	Liu et al. (2018)
11	Switzerland	Floodplain soils	<593 items/kg	0.125–5 mm	μ-FTIR	Scheurer, and Bigalke (2018)
12	Gulf of Mannar (India)	Sediments	408 particles/kg	<5 mm	Microscope and FTIR	Vidyasakar et al. (2018)
13	Bloukrans River (Australia)	River sediments	0.216 g/L	500 μm	Sieved and Microscope	Nel et al. (2018)
14	Changsha, (China)	Sediments of urban water	270-866 particles/kg	<1 mm	FTIR and Visual	Wen et al. (2018)
15	Southeast Mexico	Home garden soils	0.87 ± 1.9 particles/g	1–10 μm	Microscope	Huerta Lwanga et al. (2017)
16	Sydney, Australia	Industrial soils	300–67500 mg/kg	__	Microscope and FTIR spectroscopy	Fuller & Gautam (2016)
17	Heungnam beach (South Korea)	Sediments	0.3285 g/L	38 μm – 1 mm	Sieves and Visual	Heo et al. (2013)
Microplastics in aquatic organisms						
1	Tropical Eastern Pacific and Galápagos archipelago	Seawater marine organisms	500 μm-plankton net	150–500 μm	Olympus microscope	Alfaro-Núñez et al. (2021)
2	Southern coast, Kerala (India)	Fishes	40.7 ± 33.2 particles/m^2	0.2–0.5 mm	Microscope and ATR-FTIR	Robin et al. (2020)
4	Jurujuba Cove, Niterói, RJ (Brazil)	Mussels' organism Surface water	0.10 g/L	<5 mm	FTIR	Castro et al. (2016)
5	Arabian Sea and coast of Kerala (India)	Fishes gut	–	0.3–3.2 mm	FTIR and Raman spectroscopy	James et al. (2020)
6	Cochin, (India)	Shrimps of coastal water	0.4 particles/shrimps	05-1 mm (72%)	FTIR	Daniel et al. (2020)
7	KwaZulu-Natal, South Africa	Juvenile fish	0.8–1.0 particles/fish	0.1–4.8 mm	FTIR	Naidoo and Glassom (2019)
8		Zooplankton	Abundance in 90%	<5 mm	Microscope and FTIR spectroscopy	

(*continued on next page*)

Table 2 (*continued*)

Sr No.	Location	Environmental components	Average Concentration/ abundance	Size/Range	Detection methods	References
	Port Blair Bay, (Andaman Island)					Goswami et al. (2020)
Microplastics in air and roadside dust						
1	Ahvaz City, Iran	Atmospheric particles	<0.017 m^3	10 μm	High Volume Air Sampler (HVAS) and Binocular microscopy	Abbasi et al. (2023)
2	Kuwait's Baseline	Indoor Aerosol	3.2–27 particles/m^3	≤2.5 μm	fluorescence stereomicroscope, SEM and micro-Raman spectroscopy	Uddin et al., 2022
3	Guangzho, China	Atmospheric deposition	114 ± 40 particles/m^2/day	<5 mm	Microscope and FTIR	Huang et al. (2021)
4	Chennai metropolitan city, (India)	Road/Street dust	227.9 ± 91.4 particles/100 g	<5 mm	Raman spectroscopy SEM-EDS	Patchaiyappan et al., 2021
5	Urban and Rural area Nagpur, India	Roadside Suspended Dust	50-120 particles/day	$PM_{2.5}$ PM_{10}	FTIR	Narmadha et al. (2020)
6	East Indian Ocean	Atmospheric particles	0.4 particles/100m^3	643.1 μm	Microscope, Spectroscopic and FTIR	Wang et al. (2020d)
7	Pearl River South China Sea (China)	Atmospheric particles	4.2 particles/100 m^3 0.8 particles/100 m^3	917.4 μm 953 μm	Microscope, Spectroscopic, and FTIR	Wang et al. (2020d)
8	Metropolitan area, Hamburg (Germany)	Atmospheric deposition	275 particles/m^2/day	<63 μm (fragment); 5-0.3 μm (fibres)	Raman spectroscopy; fluorescence microscope	Klein and Fischer (2019)
9	Remote mountain region (French Pyrenees)	Atmospheric deposition	365 particles/m^2/day	<5 mm	Microscope and μ-FTIR	Allen et al. (2019)
10	Dongguan city, (China)	Atmospheric fallout	175-313 particles/m^2/day	<5 mm	SEM, Visual observation and μ-FTIR analysis	Cai et al. (2017)
11	Greater Paris	Atmospheric deposits	118 particles/m^2/day	500–1000 μm	Microscope	Dris et al. (2015)
12	Urban area (Paris)	Atmospheric deposition	2-355 particles/m^2/day 3–10 ton/year	<5 mm	Stereomicroscope and ATR-FTIR	Dris et al. (2016)

sediment. Their abundance in river sediment was found in the range of 107–410 particles/kg. Sruthy and Ramasamy (2017) reported approximately 0.027 g/L of small size (0.2–1.0 mm) microplastics in sediments of Vembanad lake (India).

In North American regions, microplastics have been found in the freshwater sources and sediments of the USA and Canada. Anderson et al. (2017) reported that the surface water of Winnipeg lake (Canada) was contaminated with 1.74 g/L of microplastics and the estimated microplastics was 1933 particle/L. The Canadian lakes and rivers were also contaminated with 0.50 g/L of microplastics, as reported by Anderson et al. (2016). Similarly, Zbyszewski and Corcoran (2011) also detected the microplastics in the sediments of Lake Huron in Canada and the USA, and their approximate concentration was 3500 particles/L. Eriksen et al. (2013) detected microplastics in the surface water (0.02 particles/L) of the Great lake of the USA. Some researches in European nations revealed that microplastics infiltrate rivers and lakes. According to Dris et al. (2015), the French river Seine was polluted with microplastic contamination. Similarly, microplastics have been found in the surface water and sediments of rivers and lakes in other countries, including Lake Chiusi and Lake Bolsena in Italy (Fischer et al., 2016), surface water of Flemish rivers in Belgium (Slootmaekers et al., 2019), and sediments of the Kelvin river in the United Kingdom (Blair et al., 2019).

The concentration of microplastics varied within different locations on the continent. Furthermore, since microplastics settle in freshwater systems, sediments are the principal sink of microplastics in rivers. Therefore, their concentration in sediments are substantially greater than in water (Wang et al., 2017). In aquatic environment, microplastics are not biochemically inert and leach chemical additives called plasticizers. These plasticizers have been introduced during manufacturing to impart various properties to the product like heat stabilization, acid scavenging, slip compounds, flame retardants, etc. (Lechner, 2020).

Although microplastics pollution on land is many times more than in the oceans, the aquatic systems have received much scientific attention compared to their terrestrial counterparts (de Souza Machado et al., 2017). Also, microplastics act as a vector or carrier of toxic sponges for transporting chemicals or pollutants by absorbing or adsorbing them (Amrutha and Warrier, 2020). Microplastics can absorb and concentrate hydrophobic persistent organic pollutants (POPs) in water, such as polychlorinated biphenyls (PCBs) and polycyclic aromatic hydrocarbons (PAHs), as well as inorganic pollutants such as heavy metals, due to their chemical structure (Talvitie et al., 2017). Metals and microorganisms have also been reported to attach to microplastics (Bondelind et al., 2020).

3.2. *Microplastics in atmosphere*

The main sources of microplastics in the air are synthetic textiles, erosion of synthetic rubber tires, and urban dust (Prata, 2018). Other sources may include building materials, industrial emissions, plastic fragments from house furniture, particle resuspension, landfills, traffic particles, waste incineration, tumble dryer exhaust, synthetic particles used in horticultural soils, and sewage sludge as fertilizer (Dris et al., 2016, 2017; Liebezeit and Liebezeit, 2014). After being released into the atmosphere, the diverse forms of microplastics are continually transported and settled in soil or sediment. Then, there is a possibility of their translocation, bioaccumulation, and accumulation in trophic levels (Klein and Fischer (2019). Total microplastics in the environment are airborne microplastics in wet and dry atmospheric deposition. Dry deposition is described as gravitational deposits of microplastics floating in the air and gases and being settled by gravitational force. They could be deposited on any ground surface or forest canopy that has been washed away by rains (Schlesinger and Bernhardt, 2013). Wet deposition includes precipitation, suspended, particulate, and dissolved particles separated by rainout and washout and dispersed in an environmental matrix.

Zhang et al. (2020d) observed high concentration of microplastics in indoor environments (1–60 particles/m^3 and 1600–11,000 particles/m^2/day) compared to the outdoors. Dris et al. (2017) reported that microplastics concentrations in a Paris apartment were between 1 and 60 fibres/m^3 higher than outdoors (0.3–1.5 fibres/m^3). The high detected concentrations of microplastics in indoor environments might be attributed to the higher flux of indoor microplastics sources and fewer particles being removed by dispersion processes than in outside environments. Vianello et al. (2019) collected the indoor airborne microplastics samples from three apartments in Aarhus, Denmark, and reported that the most abundant types were polyester (59–92%), followed by polyethene (5–28%), nylon (0–13%), and polypropylene (0.4–10%).

Another thing is that the local wind-blown debris was majorly involved in adding microplastics to the atmospheric deposits. The predominant microplastics present in this atmospheric deposits were fragments, fibres, and particles (Dris et al., 2016; Cai et al., 2017). The average concentration of microplastics in outdoor air varied within different regions of the world. For example, the deposition rate of atmospheric microplastics ranged from 175 to 313 particles/m^2/day in Dongguan, China (Cai et al., 2017). However, atmospheric transport may also carry microplastics to isolated mountains and marine regions (Bourzac, 2020). A recent study (Allen et al., 2019) on remote and pristine mountain regions (French Pyrenees) confirmed the possibility of atmospheric microplastics being transported from urban regions to other remote areas where there is no anthropogenic activity. Approximately 366 particles/m^2/day of microplastics in the atmospheric deposition were observed in remote mountainous regions.

3.3. Microplastics in soil

Soil is the second largest depository of microplastics and transport routes through which microplastics pollution occurs in various surface and subsurface environmental matrices (Horton et al., 2017; Zhang and Liu, 2018; Rillig, 2012). A range of sources, including industrial sources, residential plastic waste, sewage sludge, air depositions, and wastewater irrigation, contaminate the soil ecosystem directly and indirectly (Zhou et al., 2020a; Blasing and Amelung, 2018). Once accumulated in soil, microplastics may naturally breakdown and bioaccumulate in plants, soil organisms, and biodiversity (Chae and An, 2018; de Souza Machado et al., 2018). Microplastics also act as a carrier and transfer various other soil pollutants to soil biota, marine systems, and other toxins to living organisms (He et al., 2018). Exposed agricultural soil surfaces could be significant contributors of microplastics to the atmosphere or rivers (through runoff). Microplastics in floodplains are likely to enter aquatic systems via heavy rains and floods (O'Connor et al., 2019; Gao et al., 2021).

Soil microplastics pollution in China requires special attention since large amount of plastics are produced, consumed, and discharged in China every year (Zhu et al., 2019a). Microplastics are fragmented in soil depending on type of land application (Wang et al., 2020c). In recent years, low-density plastic has been used for mulching purpose in China's farmland, which could be a key source of microplastics in agricultural soil (Sarker et al., 2020). Many of these mulches contain plastic waste residues that release harmful additives, such as 50–120 mg phthalates/kg, leading to 74–208% higher phthalate concentrations than in non-mulched soils in China (Kong et al., 2012). Recently, Zhou et al. (2020a) also quantified microplastics in different agricultural sites in the vicinity of Hangzhou Bay, China, and their average concentration was 310 items/kg. Mulched soils contain an average of 571 pieces/kg, higher than non-mulched soils, i.e., 263 pieces/kg, and particle size varied from 1 to 3 mm in soil. Liu et al. (2018) quantified both microplastics and mesoplastics in the deep and shallow soils of farmlands around Shanghai, China. The average microplastics concentration in deep and shallow soil was 62.5 items/kg and 78.0 items/kg, respectively. Mesoplastics were also prevalent, with concentrations of 6.75 items/kg and 3.25 items/kg in shallow and deep soils, respectively.

According to published research, 63 to 430 thousand tons of microplastics are transported annually from farmland to ocean or surface water in Europe and 44–300 thousand tonnes in North America (Guo et al., 2020). Many studies have shown that microplastics in soil from municipal landfills are due to the progressive disintegration of plastic waste (Dris et al., 2016; Guo et al., 2020; Rochman, 2018). Recent studies also evidenced microplastics occurrence in terrestrial ecosystems (Rillig et al., 2017; Horton et al., 2017). Fuller and Gautam (2016) reported that soils around industrial locations in Australia were polluted with 0.03–6.7% microplastics, with quantities ranging from 300 to 67,500 mg/kg. Microplastics have also been detected in home garden soils in Campeche, Mexico, where the mean concentration was 0.87 $\pm$ 1.9 particles/g (Lwanga et al., 2017). Furthermore, Scheurer and Bigalke (2018) reported microplastics in nearly 90% of Swiss floodplain soils at depths ranging from 0 to 5 cm. They observed that the average microplastics content was 5 mg/kg, with the highest value of 55.5 mg/kg. From the metropolitan area of Chennai (India), it was reported that the street dust samples had an average microplastics abundance of 227.94 $\pm$ 91.37 per hundred grams (Patchaiyappan et al., 2021).

3.4. Microplastics in marine environment

Microplastics are ubiquitous emerging marine pollutants that pose a global environmental threat (Shim and Thomposon 2015). According to global model calculations, 1.15–2.41 million tons of plastic garbage reach the ocean through rivers each year, with the top 20 polluting rivers primarily located in Asia, accounting for 67% of the global total (Lebreton et al., 2017). Cosmetics, pellets, and air-blasting media containing microplastics may infiltrate rivers through domestic and industrial drainage systems (Sharma and Chatterjee, 2017). Wastewater treatment plants (WWTPs) also contribute to the quantity of plastics in the ocean by discharging wastewater directly into the oceans or rivers, which then carry them to the sea (Sun et al., 2019).

Human activities such as tourism, recreational and commercial fishing, shipping, and the marine industry release enormous quantities of plastics/microplastics into the ocean in coastal regions (Cole et al., 2011). Plastics cannot biodegrade easily and may remain in the marine environment for long periods. From the Arctic to the Antarctic, an estimated 5 trillion pieces of plastics are floating in the world's oceans (Isobe et al., 2015; Matsuguma et al., 2017). Their distribution and fate in the ocean still need to be explored since microplastics are challenging to separate mechanically after they reach the environment. Some studies confirmed the presence of microplastics in surface waters to deep-sea (Desforges et al., 2013), sediments (Matsuguma et al., 2017), and freshwater systems (Sarkar et al. (2019).

In offshore Pacific waters, microplastics concentration ranged from 8 to 9200 particles/m^3. It has been increasing continuously in other regions, including west coast Vancouver Island, Straits of Georgia, and Queen Charlotte Sound in British Columbia, Canada (Desforges et al., 2013). Fauziah et al. (2015) reported the occurrence of microplastics debris in sand beaches in Peninsular Malaysia. A total of 2542 pieces of microplastics debris were found on all the studied beaches. Bagaev et al. (2021) recently made several investigations on the presence of microplastics in Russian seas. Seven of these investigations found that microplastics levels in water ranged from 0.6 to 336,000 items/m^3 and from 1.3 to 10,179 items/kg in sediments. Other studies have been published on the distribution of microplastics fragments (diameters of $<$5 mm) in open seas including Arctic polar waters (Lusher et al., 2015), marginal seas (Isobe et al., 2015), and coastal waters (Isobe et al., 2017). Lusher et al. (2015) reported microplastics presence in surface (top 16 cm) and sub-surface (6 m depth) samples of Arctic waters south and southwest of Svalbard, Norway. La Daana et al. (2018) reported the presence of microplastics in ice cores from remote areas of the Arctic Ocean. Isobe et al. (2015) investigated microplastics concentrations in the East Asian Seas around Japan, finding a total particle count of about 1.72 million pieces km^2 (10 times higher than in the North Pacific and 27 times higher than in the global oceans). Nel and Froneman (2015) investigated the presence of microplastics in South African beach sediment and seawater, where the microplastics concentrations varied from 340.7 to 4757 particles/m^2 and 204.5 to 1491.7 particles/m^3, respectively.

Microplastics abundance in sediments was reported to be 37.1- 42.7 items/kg in the Yellow Sea, China (Zhu et al., 2018), 250–300 items/kg in Edgbaston Pool, UK (Vaughan et al., 2017), and 45,76,115 items/km^2 in the Balearic Islands, Spain (Ruiz-Orejón et al., 2018). Zhang et al. (2020a) quantified the amount of microplastics in deep-sea (ranging from 4601 m to 5732 m) sediments in the Western Pacific Ocean, where the microplastics abundance averaged 240 items/kg (dry weight). The microplastics were mainly fibrous in shape (52.5%), blue (45.0%), and less than 1 mm in size (90.0%). Matsuguma et al. (2017) studied microplastics (<5 mm) in surface sediments from Japan, Thailand, and Malaysia, where the abundance of microplastics ranged from 100 pieces/kg (Gulf of Thailand) to 1900 pieces/kg (a canal in Tokyo Bay).

Microplastics suspended in the atmosphere are distributed in marine air and ocean surface. Wind movement and air dynamics could carry microplastics pollution from the terrestrial ecosystem to the ocean and marine atmosphere. Liu et al. (2019b) investigated the presence of atmospheric microplastics (SAMPs) in the western Pacific Ocean for the first time in 2019. The abundance of SAMPs was found to be 0.13 ± 0.24 n/m^3 in the coastal region and 0.01 ± 0.01 n/m^3 in the pelagic area. SAMP abundance was 0.46 n/m^3 during the day and 0.22 0.19 n/m^3 at night. About 90% of the microplastics were present as fibres and fragments. Microplastics abundance was also found in remote areas (Northwestern Pacific), i.e., 1.0 × 10^4 items/km^2 (Pan et al., 2019).

3.5. Microplastics in the food chain

According to the findings, a wide range of aquatic organisms can consume and accumulate microplastics (Alfaro-Núñez et al., 2021). Numerous marine organisms, including estuarine crustaceans, fish, intertidal shellfish, mussels, barnacles, lugworms, sea cucumbers, amphipods, and sea birds have been shown to consume microplastics (Digka et al., 2018; Provencher et al., 2018; Covernton et al., 2019; Iannilli et al., 2019; Mohsen et al., 2019; Xu et al., 2020). After accidental or intentional consumption, microplastics are transported through the epithelium of the gastrointestinal tract before being retained or egested through faeces. Microplastics can be fragmented in marine animals into even nanoplastics (Niederholtmeyer et al., 2018). Microplastics in the food chain seriously threaten marine ecosystems and human health. Microplastics contaminated food chains have harmed around 690 marine species (Carbery et al., 2018). Rothstein (1973) first reported microplastics pollution in the marine environment and its presence in the marine food chain. After that, numerous researches on microplastics pollution and microplastics fragment ingestion in marine organisms such as whales, turtles, fish, snails, and seabirds, among others, were published (Matsuguma et al., 2017).

Retention of microplastics in the gastrointestinal tract can severely affect the organism's health by causing physical abrasions and/or perforations, decreasing nutrient uptake, and reducing feeding activity because of the feeling of false satiety (Walkinshaw et al., 2020). Various studies confirmed that microplastics could also enter the food chain of aquatic and terrestrial ecosystems. Most microplastics researches have focused on microplastics ingestion and thier analysis in the stomachs of marine organisms (Rochman et al., 2013). Lwanga et al. (2018) quantified the microplastics in the terrestrial organisms waste such as earthworms cast and chicken faeces. Microplastics in earthworms casts and chicken faeces were 14.8 ± 28.8 particles/g and 129.8 ± 82.3 particles/g, respectively. The floating particles and microplastics fragments are readily swallowed and absorbed by small marine organisms, which are subsequently directly fed by large organisms. Microplastics may therefore, have an impact on the whole food web (Choy et al., 2019; Nelms et al., 2019). Panebianco et al. (2019) found that microplastics were present in more than 50% of the snails (i.e. a total of 425 specimens), with an average of 0.92 ± 1.2 particles/snail. The feeding habit of snails and their presence near agricultural fields have enhanced the exposure of microplastics contamination in the snails. Snails are a part of the human food chain also. The reported studies on the microplastics existence in the food chain has become a major human health concern globally.

According to some studies, the chlorinated Polychlorinated Biphenyls (PCBs) in tissue was high in birds with microplastics in their stomachs (Yamashita et al., 2011). The larger microplastics can cause internal abrasion, clogging of the digestive system, and intestinal lesions (Cole et al., 2013; Wright et al., 2013; Ahrendt et al., 2020). Microplastics also exists in the soft tissues of marine organisms. Van Cauwenberghe and Janssen (2014) quantified microplastic in soft tissues of commercially grown bivalves: *Mytilus edulis* and *Crassostrea gigas*. During human consumption, the average quantity of microplastics in *M. edulis* and *C. gigas* was 0.36 ± 0.07 particles/g (wet weight) and 0.47 ± 0.16 particles/g, respectively.

4. Sampling, identification and quantification of microplastics

4.1. Sampling of microplastics

One of the crucial procedures in the analysis of microplastics is sampling. Depending on the goals of the research, different sampling techniques can be used for microplastics characterization. Because microplastics are so tiny, accurate sampling methods are essential for producing high-quality results. As the microplastics distribution is heterogeneous, large samples are more representative than small samples and can be further made smaller by homogenization. Random sampling is a sampling strategy that might be used to identify contamination in a site that is likely to be homogenous. Systematic grid sampling, which divides up sample locations in a regular pattern, may be utilized to verify the extent and hotspots of contamination. The first point is selected randomly, and the rest points are arranged in a well-ordered pattern. Transect sampling is also a one-dimensional systematic sampling that may be used to identify and verify the extensiveness of contamination. It is used along with linear features such as roads. Unaligned grid sampling is another approach that integrates the utility of both random and systematic grid sampling. A stratified sampling approach may be used to identify the contamination in a delineated sub-area of the entire sampling region (Moller et al., 2020). In riverine environments, microplastics can be sampled by a dynamic or stationary sampling method. In dynamic sampling, trawls are pulled by boats. The stationary sampling technique collects microplastics samples from small rivers (Campanale et al., 2020). For microplastics sampling in sediments in freshwater bodies, sampling is performed manually, demonstrating an area and a depth. Now-a-days instead of manual sampling, a corer with a specific diameter, box corer, and Ekman or Van veen grab corer can be referred in a known area for a required volume of material (Campanale et al., 2020). Microplastics of different colours, sizes, and polymers are separated and extracted after sample collection using a variety of techniques, including physical separation, density separation, filtration, magnetic separation, electrocoagulation, etc. Matrix removal techniques are utilized to take the organic matter out of the microplastics samples in order to get precise findings and cut down on processing time (Hanvey et al., 2017). Enzymatic digestion, H_2O_2 oxidation, and acid or alkaline digestion are a few typical processes used for organic matter removal from microplastics samples.

4.2. Identification and quantification of microplastic

Quantification is the counting and categorizing microplastics based on their size, colour, and type of polymers (Moller et al., 2020). Optical counting of microplastics is usually performed with a microscope, but it has some limitations concerning accuracy. Visual counting may result in misreading microplastics and the risk of identifying non-plastic particles as plastic. Due to the high diversification of polymers, specification of the chemical composition of polymers is essential to assure accuracy. Generally, the detection of microplastics can be categorized into two steps, the first one is physical (i.e., colour, size), and the second is chemical (i.e., composition, structure) identification (Sun et al., 2019). A combination of microscopy and spectroscopy may improve the accuracy of the optical counting. Some research proposes the 'hot needle test' to reduce the risk of accurate plastic counting. Zhang et al. (2018) suggested a method for identifying soil polymers. After density separation with water, the residues present in the supernatant are examined by comparing prior microscopic imaging. After heating the sample at 130 °C for 3–5 s, liquefied plastics are recognized as thermoplastics. However, specific natural polymers melt at particular temperatures, which reduces the feasibility of determining the exact polymer time.

The visual identification has been validated in numerous studies (Moller et al., 2020). However, it does not differentiate the type of plastic and is less suitable for particles with a diameter <50 μm (Zhang et al., 2018). The different extraction methods are integrated into a chromatographic unit for quantitative and qualitative identification of plastic polymers. Pyrolysis-gas chromatography-mass spectrometry and thermal extraction desorption-gas chromatography-mass spectrometry are mass-based techniques, while RAMAN and Fourier Transform infrared spectroscopy (FTIR) are particle-based techniques used for the identification of microplastics. Duumichen et al. (2017) introduced Thermal Extraction Desorption Gas Chromatography mass spectrometry (TED GC-MS). However, TED GC-MS is suitable for quick analysis of samples but is a destructive method. This method does not require any pretreatment and the time required for the complete measurement of a sample is about 2 h, a short time considering the sample mass and the depth of information. Gel permeation chromatography (GPC) is another method of size-exclusion chromatography that uses organic solvents to separate analytes based on their sizes. It is frequently used for the examination of polymers. This technique is based on differences in molecular mass, in which large molecules are excluded from the pores of the gels and are eluted first. In addition to providing a complete assessment of the particles size, shape, and polymer distribution, high temperature GPC also provides a qualitative study of stabilisers without the need for any visual sorting (Hintersteiner et al., 2015).

Vibrational spectroscopy like RAMAN or FTIR is usually used for microplastics analysis as it allows error-free identification of plastic polymers. Their spectra are used to identify several types of plastics by comparison with a spectral library (Corradini et al., 2019). Raman microspectroscopy identifies microplastics down to 500 nm pixel resolution. It can be increased to 100 nm using silver colloid for surface-enhanced Raman spectroscopy (Lv et al., 2020), while micro-FTIR spectroscopy identifies particles from 10 to 500 μm (Moller et al., 2020). According to most studies, FTIR is an excellent approach for analyzing sediment samples. To increase the precision of the data, FTIR can be combined with an optical microscope. RAMAN and FTIR are both non-destructive techniques. ATR-FTIR spectroscopy (Attenuated total reflection Fourier transform infrared spectroscopy) is also a fast and efficient approach for identifying polymers of plastic marine debris in marine water and biota (Jung et al., 2018).

Scanning electron microscopy (SEM) is also used to determine the polymer's size, shape, or morphology. To overcome the constraints of a stereomicroscope, SEM is installed for the physical analysis of microplastics. SEM images of the external surface of microplastics makes it easy to differentiate between synthetic microplastics and many organic materials that can be found with microplastics (Cooper and Corcoran 2010).

Energy-dispersive X-ray spectroscopy (EDS) is used for elementary analysis to determine the chemical compositions of plastic particles. However, it would not be easy to access SEM-EDX regularly, as it is an expensive detection method and requires more time and effort to prepare the sample. Furthermore, SEM-EDX cannot distinguish coloured microplastics. Advanced and updated microscopic analysis such as PLM (polarized light microscopy) can also be used to analyze PET, PP and PE based microplastics. A new method for analyzing size-independent microplastics is proton nuclear magnetic resonance spectroscopy (Peez et al., 2019). This method is capable of quantitative and qualitative analysis of samples containing polyethene, polystyrene, and polyethene terephthalate. However, this method is not cost-effective. As a result, this method is inadequate for soil sample analysis. Microplastics can also be identified via thermogravimetric analysis. It can be combined with differential scanning calorimetry or mass spectrometry for better results.

Out of the above-mentioned methods, FTIR microscopy is the most common approach found in microplastic research due to its exceptional reliability and easy application. To analyze the microplastics size, shape, colour and morphology, new methods such as SEM, EDX, TED GC-MS, GPC etc. can be used alone or in combinations.

5. Effect of microplastics

The worldwide presence of microplastics in the environment is regarded as an ecological hazard and a significant concern by scientists, governments, and policymakers (Li et al., 2016; Vaughan et al., 2017; Bergmann et al., 2019; Chen et al., 2020a). The existence of microplastics in living organisms is influenced by the interaction of biological and non-biological factors and ecological security, although the exact mechanism is unclear. Our understanding of the fate and impact of microplastics on the biosphere is critically needed.

5.1. Effect on soil ecosystem

Because of the persistent nature in the terrestrial environment, microplastics may interact with flora and soil organisms. According to Kleunen et al. (2020), microplastics may harm plants and their growth with concentrations of EPDM (Ethylene Propylene Diene Monomer) rubber exceeding 8% (v/v), can cause 50% mortality (LC_{50}) at concentration 13% (v/v). The growth of two species, *Leucanthemum ircutianum* and *Prunella vulgaris*, had also been suppressed during interaction with 5% (v/v) of EPDM microparticles in soil. Microplastics could modify plant structure in various ways, including changes in size, shoot, and root length, number of leaves, and colour (Kleunen et al., 2020; Khalid et al., 2020). Fig. 3 illustrates microplastics exposure in the soil system and their transport and accumulation in plants and microorganisms.

Similarly, Pflugmacher et al. (2020) showed that 3 mm polycarbonate microparticles with 59 items/kg concentration reduced *Lepidium sativum* shoots and roots length. The negative impact of microplastics on the growth of *Vigna radiata* was reported by Chae and An (2020). The seed germination and seedling are more sensitive to the microplastics. Bosker et al. (2019) studied the impact of various microplastics concentrations (107 items/mL and 104 items/mL) on seed germination of *Lepidium sativum* and found that seed germination was significantly affected during the first 8 h of microplastics solution exposure. Small microplastics exposure can significantly alter the natural ecosystem (Lozano and Rillig, 2020). Boots et al. (2019) reported a remarkable reduction in root growth of *Lolium perenne* with low-density microplastics. The plant roots directly encounter microplastics when sewage sludge as fertilizer and organic manures are applied (Watteau et al., 2018). Microplastics also affects the soil quality and nutrient cycle in the soil system (Zhou et al., 2020b). Qi et al. (2020) reported alteration in soil chemical characteristics, including pH, conductivity, and C:N ratio with LDPE and biodegradable microplastics. Yuanqiao et al. (2020) observed a decrease in the water and nitrate holding capacity due to a high dose of microplastics (360 kg/h.m^2).

Microorganisms such as mycorrhizal fungi, phosphate-reducing bacteria, mineral-reducing bacteria, and nitrogen-fixing bacteria in soil are essential for plant growth. In some studies, it is observed that microbial activity has been reduced during microplastics exposure, which directly affects plant growth and crop production (Powell and Rillig, 2018). Microplastics could alter mycorrhizal growth of fungi and comparative richness in the plant root system (Wang et al., 2020c). Similarly, Chen et al. (2020b) found that Polylactic acid (PLA) affected the interaction between microbial community and soil particles. It could also affect the mineralization and nitrogen fixation, directly reducing root expansion and plant growth. In addition, it diminishes enzyme production such as urease, glucosidase, and phosphatase. Based on the

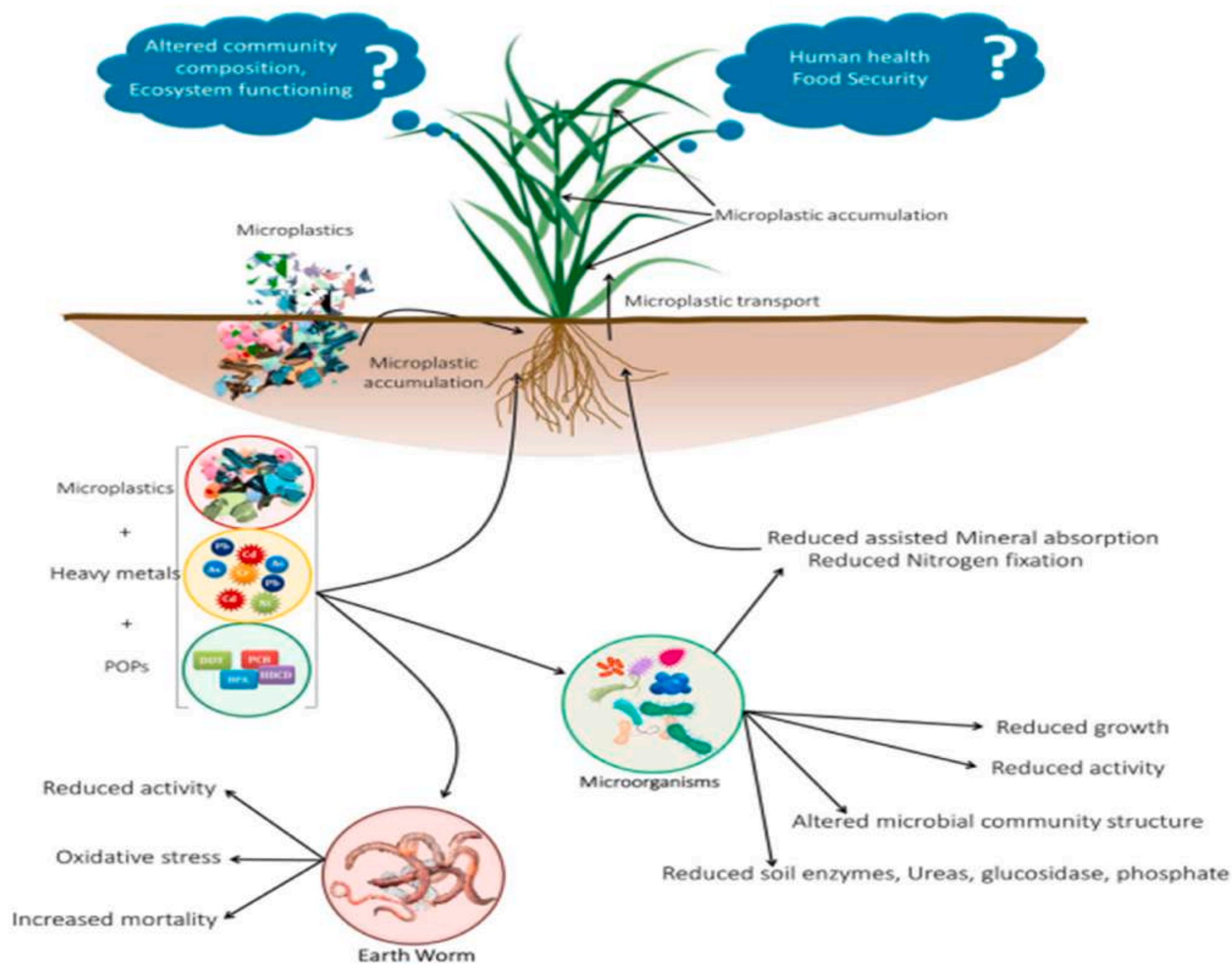

Fig. 3. Microplastic in the soil affect plant growth directly or indirectly by impacting the growth of soil-dwelling organisms **(Reproduced with permission from Ref. (**Khalid et al., 2020**). Copyright 2020 Elsevier.**

available literature, it can be assumed that exposure to microplastics, directly or indirectly, alters the natural ecosystem of soil, including plants, soil microorganisms, and crops (Huang et al., 2020b). The impact of microplastics exposure and its mechanism is still mysterious due to limited literature. Notably, plant feeder species or herbivores directly ingest atmospheric microplastics from the surface of plant leaves, thus leading to a more prominent route to enter the food chain and reaching higher trophic levels (Dovidat et al., 2020).

5.2. *Effect on aquatic ecosystem*

Microplastics and pharmaceuticals are classified as emerging contaminants that threaten aquatic ecosystems. In the last few years, many pharmaceutical products have been discharged into wastewater streams to enter the natural ecosystem. Microplastics associated with pharmaceuticals are more toxic to aquatic organisms (Li et al., 2018a). Microplastics ingestion, adsorption, and interaction behaviour differ with aquatic organisms such as submerged plants, phytoplanktons, fishes, and other top carnivores in the natural ecosystem (Allen et al., 2017). As microplastics reach the aquatic system, microorganisms such as microbial biofilm, algae, fungus, and bacteria fragment them by establishing colonies (Hoellein et al., 2014; Wang et al., 2020b). The accumulated microplastics then colonize in biofilms, which serve as a food source for aquatic organisms and eventually contaminate the food chain. Microplastics properties such as quantity, particle size, shape, origin, source, and chemical composition are important in interaction and accumulation in the natural ecosystem (Yuan et al., 2019; Gutow et al., 2016). Exposure of zooplanktons to microplastics leads to a decreased growth cycle, enhanced mortality, ingestion capacity, and even disturbing the coming generations (Besseling et al., 2014). One of the studies confirmed that exposure to small-sized microplastics caused a more significant toxic effect on various microalgae than large-sized microplastics (Anbumani and Kakkar, 2018; Huang et al., 2020a). Notably, larger-sized microplastics can reduce the photosynthesis mechanism by blocking the sunlight, whereas small-sized microplastics affect the cell wall of the algae and destroy their internal structure (Huang et al., 2020a).

When aquatic organisms consume microplastics laced with allied chemical contaminants, these allied toxins are released into their tissues, posing potential health risks (Campanale et al., 2020). Microplastics can easily capture industrial toxins and soil pollutants, carry them to long distances, and disperse them in aquatic and marine ecosystems (Li et al., 2018a). The mercury can be carried by microplastics to *Artemia nauplii* (also known as "sea monkeys" are small seawater crustaceans belonging to the Artemiidae family) and then enter the food chain at a higher trophic level (Tang et al., 2019). Toxic metal ions can also be transferred to other aquatic organisms, including snails, fish, corals, and amphibians, via the food chain (Batel et al., 2020; Carbery et al., 2018). In addition to the toxicity and impacts of microplastics, their transport mechanism may affect the bioaccumulation of associated chemical contaminants in the aquatic organisms. Researchers have begun to understand the transport behaviour of plastic debris in aquatic ecosystems due to increased microplastic pollution (Teuten et al., 2009). Recent studies on plastic debris showed that the microplastic additives gradually leaches from waste and contaminates the aquatic system (Paluselli and Kim, 2020). It could also create several health problems such as toxicity, endocrine- disruption, and mutations in the aquatic organisms (Capolupo et al., 2020).

5.3. Effect on humans

The exposure of microplastics to human health has become a serious global concern. Microplastics are ubiquitous in the environment and enters the human body predominantly via two pathways, i.e., ingestion and inhalation. Its exposure can potentially cause adverse health problems in human beings (Liu et al., 2019a; Wang et al., 2020a). Humans are being exposed to microplastics pollution due to their packet food habits. The exposure to microplastics and its health impacts on humans are currently unclear. Aquatic food products have been identified as the primary source of microplastics to human exposure (Huang et al., 2020a). The amount of microplastics particles in the food source and their transport from the food source to humans must be determined to assess the health risk of microplastics exposure. Some studies reported the presence of microplastics in different food products and resources, including table salt (Karami et al., 2017a), beer (Kosuth et al., 2018), wine (Prata et al., 2020), sugar or honey (Gerd and Elisabeth, 2015), plastic tea bags (Hernandez et al., 2019a) and water bottles (Mason et al., 2018). Hernandez et al. (2019a) reported that a single tea bag releases about 11.6 billion microplastics and 3.1 billion nanoplastics in a cup of tea.

Based on recent studies, microplastics have been observed in commercial salts that are available in more than 120 brands worldwide (Zhang et al., 2020c; Kim et al., 2018). Microplastics were also detected in drinking water sources such as tap and bottled waters (Koelmans et al., 2019). Recently, Schwabl et al. (2019) reported nine types of microplastics in human faeces, and their mean abundance was 2 particles/g in a size range of 50–500 μm. The presence of Polyethylene-terephthalate (PET) and Polypropylene (PP) indicated the ingestion of microplastics from diverse food sources (Walkinshaw et al., 2020; Bouwmeester et al., 2015). Airborne microplastics exposure causes respiratory and lung problems in humans. Vianello et al. (2019) reported that humans could inhale approximately 272 particles/day from indoor air. The inhalation of microplastics depends upon particle size; generally, <2.5 μm size particles are easily transported to the lung via respiratory tract (Wang et al., 2020d). Plastics smaller than micro size are more toxic to neurons, lungs, and respiratory system (Jeong and Choi, 2019). Wang et al. (2020e) reported the synergistic toxicity of microplastics and its associated bisphenol-A on intestinal epithelial cells. Microplastics exposure causes cell toxicity by cellular oxidative stress. Microplastics exposure reduces lipid digestion due to the formation of microplastics oil droplets and inhibits enzyme activity during the digestion process (Tan et al., 2020). Human tissue can also uptake microplastics via endocytosis (airways surface and gastrointestinal tract) (Wright and Kelly, 2017). Microplastic fibres can cause occupational health problems among workers. Studies among occupational nylon flock workers indicate that most workers faced several health risks such as increased lung cancer, respiratory irritation, occupational asthma, coughing, and lung capacity (Warheit et al., 2001). About 4% of people who work in the nylon industry in the US and Canada have these health problems (Boag et al., 1999; Wright and Kelly, 2017).

5.4. Transportation of microplastics to human

The presence of microplastics in various environmental components has been observed worldwide, such as surface water (marine water, freshwater), seabed sediments, beaches, wastewater effluents, ice, aquatic organisms and their predators, food products such as salt, honey, sugar, bottled water, plastic containers, indoor and outdoor air, etc. (Table 2) (Rahman et al., 2020; Petersen and Hubbart, 2020). However, their occurrence is not restricted to the source of their availability as microplastics can migrate over long distances via wind and water currents which mark their presence even in remote areas such as polar ice caps (Arctic and Antarctic) and mid-oceanic islands (Walkinshaw et al., 2020). The direct or indirect sources for microplastics transportation are destined for humans through ingestion and inhalation (Fig. 4).

5.4.1. Ingestion

The ingestion of food and water polluted with microplastics is the primary route for human exposure (Galloway, 2015), and marine organisms top the list among the food. Microplastics can be ingested by various marine life via different processes (Barboza et al., 2018). Microplastics in marine organisms destined for human consumption have been widely reported (Wang et al., 2020b; Zantis et al., 2020; Walkinshaw et al., 2020; Huang et al., 2020a; Yao et al., 2021). Marine organisms ingest microplastics in two ways: directly from their natural surroundings or indirectly via trophic transfer from prey and consuming contaminated feedstock (Barboza et al., 2018). Direct ingestion of microplastics is often a consequence of feeding strategy. Indirect ingestion or "trophic transfer" occurs when microplastics are confounded with prey (Barboza et al., 2018; Walkinshaw et al., 2020), e.g., widely reported microplastics presence in the stomachs of blackmouth catshark (*Galeus melastomus*) is attributed to bioaccumulation from their microplastics laden prey (Alomar and Deudero, 2017). The northern fulmars (*Fulmarus glacialis)* contain plastic debris in their stomachs and are used as a bio-indicator for ocean microplastics pollution (Terepocki et al., 2017). Numerous planktons, crustaceans, molluscs, and echinoderms consume microplastics from their surroundings during feeding, resulting in microplastics bioaccumulation in fish, shrimp, crabs, and other seafood consumed by humans (ShiChun et al., 2019; Huang et al., 2020a; Daniel et al., 2021; Yao et al., 2021). The highest concentration of microplastics (0–10.5 microplastics/g) is found in mollusks, followed by crustaceans (0.1–8.6 microplastics/g) and fish (0–2.9 microplastics/g) (Karami et al., 2017b). Numerous studies have found microplastics in a variety of marine life, including molluscs, crustaceans, and finfish, destined for human consumption (Jabeen et al., 2017; Walkinshaw et al., 2020; Wang et al., 2020b; Zantis et al., 2020), and these are among the most commonly caught marine species and farmed aquaculture species, according to Food and Agriculture Organization (FAO, 2020). These data highlighted low biomagnification in the marine food chain and a higher risk to the members of lower trophic levels compared to the higher trophic level (Walkinshaw et al., 2020). The large specific surface area and stability makes microplastics a suitable adsorbent for hazardous substances and pathogenic microorganisms. Their fine size could allow translocation to other body systems. The dispersion throughout the whole body along with hazardous substances is a matter of great concern for human health (Wright and Kelly, 2017; Pandey et al., 2020; Caruso, 2019). Polyethene, polypropylene, polyester, and polystyrene are the topmost generated polymers worldwide (Plastics Europe, 2020). Fibre and fragments of these microplastics are commonly detected in the digestive tract of fish (Wang et al., 2017; Zantis et al., 2020). However, reports regarding microplastics outside the digestive tract are currently scanty (Walkinshaw et al., 2020; Wang et al., 2020b). Recently, Ragusa et al. (2021) observed a diverse range of microplastics in the human placenta. Unfortunately, they could not explain the mechanism of microplastics translocation to the bloodstream, such as respiratory or the gastrointestinal system. The transfer of microplastics from a lower trophic level to a higher trophic level was reported through the aquatic food chain (Huang et al., 2020a), such as waterbirds (Brookson et al., 2019), penguins (Le Guen et al., 2020), seals (Hernandez et al., 2019b), humpbacked dolphins (Zhu et al., 2019b), beluga whales (Moore et al., 2020), sharks (Maes et al., 2020), and even humans (Schwabl et al., 2019). The

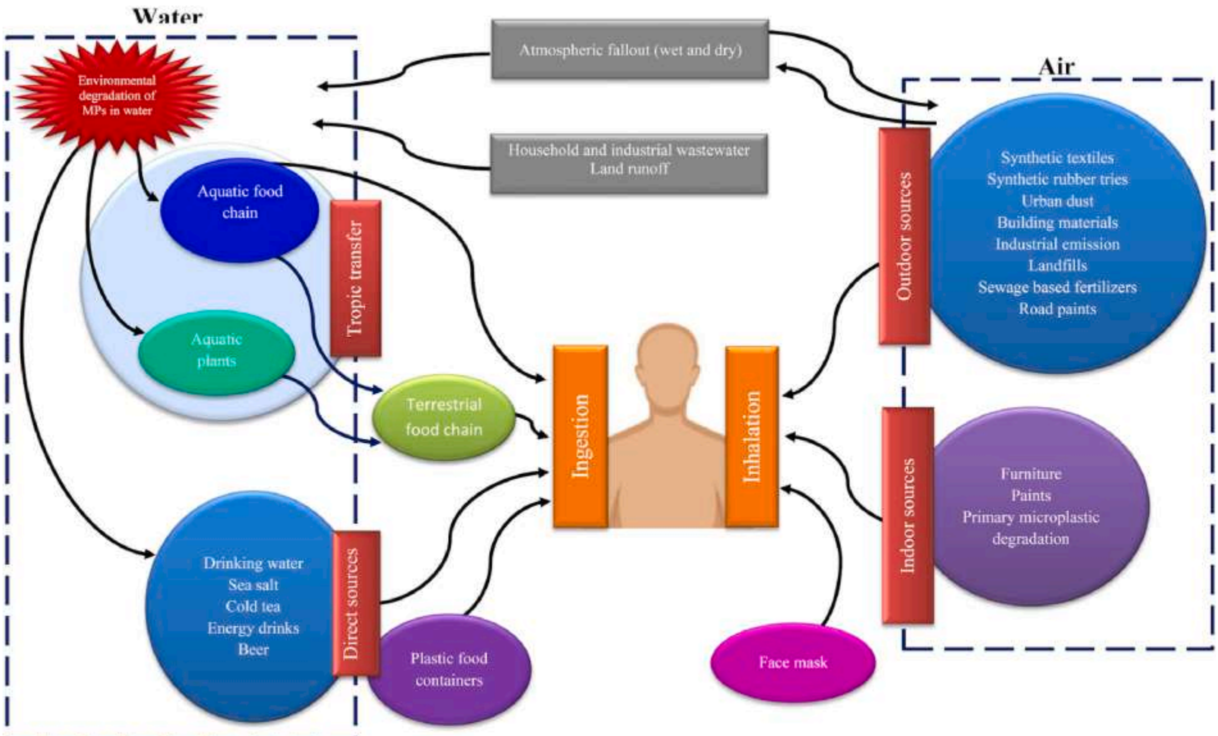

Fig. 4. Major sources and routes of microplastic to humans.

presence of microplastics in freshwater birds such as Geese, Duck, and loons was also confirmed (Holland et al., 2016; Reynolds and Ryan, 2018). Another way through which microplastics are destined for humans is sea salt (Kim and Song, 2021), drinking water (Shen et al., 2021c), cold tea, energy drinks, beer (Shruti et al., 2020a), and food containers (Du et al., 2020; Fadare et al., 2020). Around 28 sea salt brands from 16 countries on six continents showed microplastics presence ranging from 0 to 1674 particles/kg, and sea salt is more contaminated with microplastics than rock salts and lake salts (Kim et al., 2018). Karami et al. (2017a) observed 1 to 10 microplastics/kg of salt of 17 brands from 8 different countries, and among the 72 extracted particles, polymers share 41.6%, followed by pigments (23.6%) and amorphous carbon (5.50%), and 29.1% remained unidentified. Recently, Vidyasakar et al. (2021) compared microplastic concentrations in two major salt-producing states in India. The research found that salt of Gujarat origin contained higher microplastics (46–115 particles/200 g) than Tamil Nadu salt (23–101 particles/200 g), which were polyethene, polyester, and polyvinyl chloride.

The microplastics presence in human drinking water, such as raw water (Koelmans et al., 2019), and bottled water (Makhdoumi et al., 2021), is now an emerging issue. The highest concentration of microplastics was recorded in beer (28 particles/L), followed by energy drink (7 particles/L) and cold tea (6 particles/L), and the predominant polymer types were polyamide and poly (ester-amide) (Shruti et al., 2020a). After direct and hot water flushing, Du et al. (2020) observed that microplastics varied from 3 to 29 pieces per take-out food container of different polymer materials. Microplastics were prevalent in containers with rough surfaces. They estimated that those who use take-out food containers 4–7 times each week might consume 12–203 microplastics particles. Duckweed (Dovidat et al., 2020), seagrass (Goss et al., 2018), and mangrove (Li et al., 2018a) have all shown the potential to trap microplastics via various mechanisms. They provide another route for microplastics to higher trophic levels through the terrestrial food chain. Cigarette butts comprise over 15,000 detachable strands of plastic fibres. Their disposal in the open leads to an estimated 0.3 million tonnes of waste entering the oceans yearly (Belzagui et al., 2021; Shen et al., 2021b). Recently, Conti et al. (2020) revealed the presence of nanoplastics and microplastics in edible fruits and vegetables purchased from markets in Catania and first to evaluate the estimated daily ingestion by adults and children.

5.4.2. *Inhalation*

More than 50% of secondary microplastics derived from land-based anthropogenic activity are retained in the terrestrial environment (Bullard et al., 2021). Atmospheric fallout not only acts as the source of microplastics for water bodies (ocean and inland) and land, but also as a direct source for humans by inhalation. Microplastics deposition rate (average) were measured in atmospheric fallout in different megacities such as Paris, France (110 ± 96/m^2/day; Dris et al., 2016), Dongguan City, China (36 ± 7/m^2/day; Cai et al., 2017), London, UK (575–1008/m^2/day; Wright et al., 2020), Hamburg, Germany (136.5–512.0/m^2/day; Klein and Fischer, 2019), Tehran, Iran (88–605 items/30 g dry dust; Dehghani et al., 2017) and remote areas of the Pyrenees Mountains (365/m^2/day; Allen et al., 2019). This indicates that microplastics pollution has become a global issue (Mishra et al., 2021). Stanton et al. (2019) observed 2.90–128.42 fibres/m^2/day in the dust sample collected from the roofs of university campus in Nottingham, UK, and concluded that fibres of atmospheric fallout were of natural origin. Fibrous microplastics were the most prevalent shape observed in atmospheric fallout, and they were preferentially carried over a longer distance (more than 95 km) than microbeads, and they were easily inhaled by humans (Bullard et al., 2021). Fibres were dominant in microplastics in indoor (88.0%) and outdoor (73.7%) dusts from China (Liu et al., 2019a). However, the physical characteristics of microplastics, such as size, shape, density, etc., along with different meteorological conditions, determined the travelling length of microplastics in the air. Synthetic textiles, synthetic rubber tires, and urban dust are the main source of microplastics in the air (Chen et al., 2020a; Wang et al., 2020d). Other outdoor sources include construction and building materials, industrial emissions, waste management practices such as incineration, landfills, and sewage sludge, and transportation (which is attributed to particles such as rubber tyres and road paints).

The indoor sources include particle fragments from house furniture, paints, and domestic use plastic-based articles such as plastic containers (Wang et al., 2020d). The low dispersal mechanism in the indoor environment causes a higher concentration of microplastics (1600–11,000 particles/m^2/day) than outdoors (Zhang et al., 2020d). Therefore, children are more vulnerable as a result of direct inhalation and ingestion through the mouth via dirty toys, and fingers contaminated with settled microplastics and dust-carrying microplastics (Dehghani et al., 2017). According to Liu et al. (2019a), children in major Chinese cities inhaled an average (geometric mean) of 17,300 ng/kg-BW (average body weight) of microplastics derived from polyethene terephthalate daily. Zhang et al. (2020b) collected 286 indoor dust samples from 12 different countries and observed polyethene terephthalate-based microplastics ranging from 38 to 120,000 μg/g, whereas polycarbonate-based microplastics ranged from <0.11–1700 μg/g. The median daily intake of polyethene terephthalate-based microplastics by infants was in the range of 4000–150,000 ng/kg-BW/day. However, the atmospheric transport mechanism of microplastics is still unclear, and inhalation of microplastics is the least explored field (Petersen and Hubbart, 2020; Rahman et al., 2020; Can-Güven, 2021). Wet deposition (rainfall and snowfall) is a major event associated with atmospheric microplastic accumulation and removal. Microplastics were elevated with high rainfall, i.e. 2–34 particles/m/day with 0–0.2 mm of rainfall and 11–355 particles/m/day with 2–5 mm of rainfall (Dris et al., 2016). Xia et al. (2020) observed a positive correlation ($R^2 = 0.94$) between rainfall and microplastics concentration in Lake Donghu, China, and reported that microplastics ranged from 7.4 to 29.6 items/L with a dominance of size <2 mm, fibre shape, and transparent colour. Snowfall is responsible for microplastics deposition in urban and remote (e.g. the Arctic) terrestrial land and the ocean (Zhang et al., 2020e). Bergmann et al. (2019) recently observed microplastics deposition ranging from 190–154 × 10^3 particles/L and 0–14.4 × 10^3 particles/L in melted snow collected from Europe and the Arctic, respectively. Abbasi and Turner (2021) recorded >16,000 microplastics retrieved from filtered washes of hand and face skin, head hair, and saliva of humans (n = 2000, exposure time of 24 h). The maximum microplastics were from head hair (>7000, or, on an average, >3.5 microplastics/individual/day) and minimum from saliva (about 650, or on average 0.33 microplastics/individual/day). Males had almost twice the amount of microplastics as compared to females. A high concentration of microplastics on head hair can be correlated with atmospheric fallout. Recently, COVID-19 attributed to the use of facemask which increased their consumption and production across the world and have introduced new risks to human health and environmental challenges by adding vast amounts of polymers such as polypropylene, polyurethane, polyacrylonitrile, polystyrene, polycarbonate, polyethene, or polyester in the environment (Fadare and Okoffo, 2020; Aragaw, 2020). The WHO estimates that approximately 89 million medical masks are required monthly to respond to COVID-19 (WHO, 2020; Fadare and Okoffo, 2020). Microplastics are now regarded as an emerging component of air pollution due to their inhalation and interaction with other pollutants such as heavy metals (mercury, lead), PAHs, pesticides (DDT, hexachlorobenzene), pharmaceuticals product, etc. (Zhang et al., 2020e; Puckowski et al., 2021).

6. Removal/recovery of microplastics

When compared to the rural environment, the urban environment contains a large amount of microplastics, and common practices such as domestic waste disposal, street washing, and rain runoff, transport microplastics into the sewer system, where they end up in municipal wastewater treatment plants (WWTPs) (Bilgin et al., 2020). The removal and/or recovery of microplastics from WWTPs can significantly reduce their amount to be discharged into the natural environment such as water bodies (usually the final disposal site), further reducing their availability for bioaccumulation and transfer to higher trophic levels such as a humans.

6.1. Microplastics recovery from WWTP

The recovery rate of microplastics in conventional WWTPs, including preliminary, primary, and secondary stages, was 88% and can be enhanced to 99.9% by adding tertiary stage in WWTPs (Sun et al., 2019; Iyare et al., 2020). The removal of microplastics is dependent on the type of unit operation and unit process. The majority of microplastics in WWTPs are removed through screening, grit removal, grease removal, skimming, sedimentation (preliminary and primary treatment), activated sludge (secondary treatment), and membrane bioreactor (MBR), rapid sand filter (RSF), disc filter (DF), dissolved air flotation (DAF), ultrafiltration (UF), reverse osmosis (RO), gravity filter, etc. (tertiary treatment) (Fig. 5). Overall, the preliminary treatment for microplastics removal is a function of the nature and functioning of unit operations (Liu et al., 2021). The microplastics removal efficiency of any unit operation is strongly influenced by the microplastics concentration and nature, such as shape, size, and density. The removal efficiency of microplastics is low in a single unit operation; however, the combination may improve removal efficiency. Generally, grit and grease treatment exhibits poor removal efficiency for microplastics (Liu et al., 2021), but a recent study revealed 69–79% removal efficiency by screening and grit treatment (Ziajahromi et al., 2021). The grease skimming process performs better for low-density and relatively large microplastics that easily float during flotation (Sun et al., 2019; Bilgin et al., 2020). Aerated grit chamber with primary settling tank exhibited 40.7% removal efficiency (Liu et al., 2019c). The combination of screening, grit and grease processes, skimming, and settling removed a considerable portion of microplastics load ranging between 32% to 98% (Talvitie et al., 2015; Hidayaturrahman and Lee, 2019; Sun et al., 2019). Flotation and sedimentation are two popular unit operations used to remove microplastics, and their efficacy depends on the microplastics shape, size, and density. The flotation exhibited 59% removal efficiency for microplastics with low material density (<1 g/cm^3), relatively large size (1–5 mm), and flat shapes (e.g., films). Conversely, sedimentation rapidly removed the microplastics (91% removal efficiency) with high material density (>1.1 g/cm^3) and relatively large structures (e.g., fragments). Moreover, neither flotation nor sedimentation was the primary mechanism for the removal of microplastics with small sizes (<1 mm) and densities near water (1 g/cm^3) (Bilgin et al., 2020). The conventional secondary treatment, which includes biological reactors (aeration tanks, trickling filters, etc.) and settling tanks, could not reduce the significant microplastic load as they were not developed primarily for microplastics removal (Iyare et al., 2020; Zhang and Chen, 2020). Iyare et al. (2020) observed that biofiltration was more effective than trickling filters and solids contact tanks. The primary sedimentation tank and aeration tank with clarifier showed only 33.75% and 20.07% removal efficiency, respectively (Murphy et al., 2016). Individual aeration tanks had a removal efficiency of 79% (Gundogdu et al., 2018), which increased to 95.6% when the aeration tank was used as a secondary unit operation in the activated sludge process (Michielssen et al., 2016). The full-scale conventional activated sludge process removed 86% of microplastics (Pittura et al., 2021). Modern operation units in WWTPs such as sequential batch reactor (SBR), aerobic membrane bioreactor (MBR), anaerobic membrane bioreactor, and disc filter (DF, pore size 20 μm) exhibited 99.2% (Lee and Kim, 2018), 99.9% (Talvitie et al., 2017), 99.4% (Michielssen et al., 2016), 98.5% (Talvitie et al., 2017) removal efficiency for microplastics, respectively. The A^2O (aerobic-anoxic-aerobic method) was unsuitable for microplastics removal due to its poor removal efficiency (16.9%) and significant sludge return (Jiang et al., 2020). The tertiary treatment technologies are specific, but membrane-based technologies for microplastics removal exhibit the best performance. Some common tertiary techniques which reduce significant microplastics load are rapid sand filter (97–98.9%) (Talvitie et al., 2017; Hidayaturrahman and Lee, 2019), Granular sand filtration

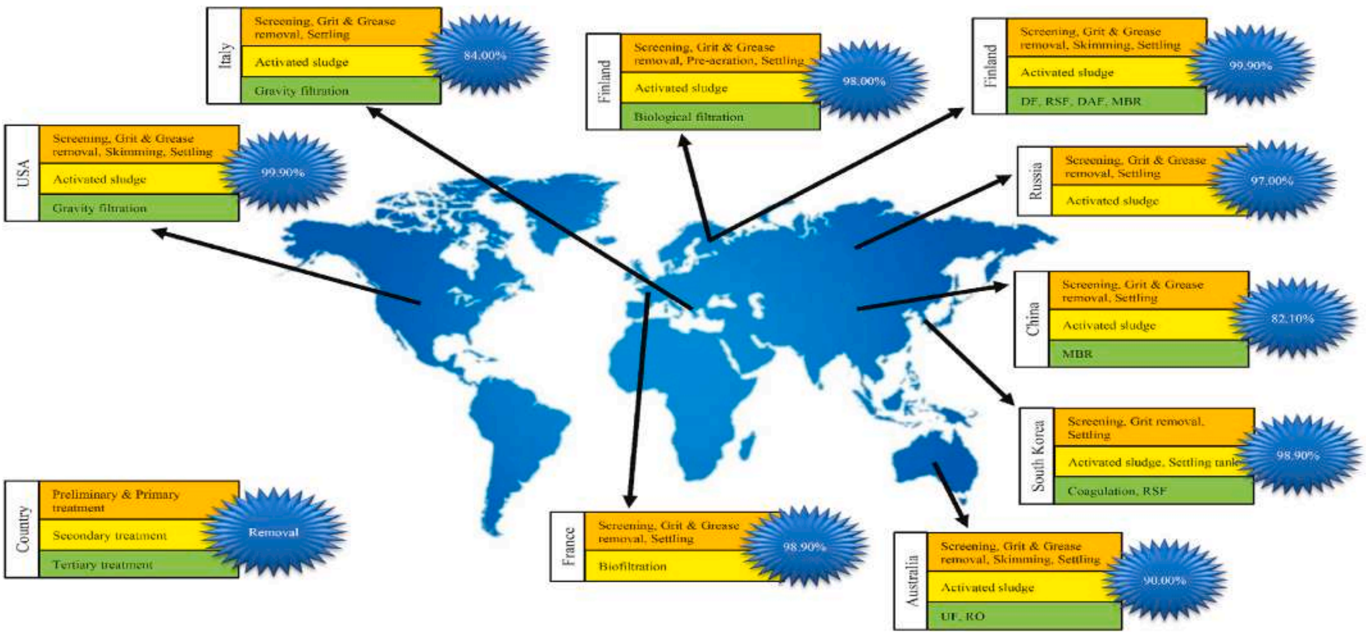

Fig. 5. Wastewater treatment process and overall removal efficiency for microplastics. USA (Carr et al., 2016), Italy (Magni et al., 2019), Finland (Talvitie et al., 2015, 2017), Russia (Talvitie and Heinonen, 2014), China (Lv et al., 2019b), South Korea (Hidayaturrahman and Lee, 2019), Australia (Ziajahromi et al., 2017), France (Dris et al., 2015).

(97.2%) (Michielssen et al., 2016), ozone (99.2%) (Hidayaturrahman and Lee, 2019), and membrane disc filter (99.1%) (Hidayaturrahman and Lee, 2019). Rapid sand filter technology can fragment microplastics into smaller particles (Sol et al., 2020). Ziajahromi et al. (2017) demonstrated poor microplastics removal efficiency of ultrafiltration (41.6%) and reverse osmosis (25%). Shen et al. (2021a) demonstrated significant removal efficiency of >96% for granular polyethene microplastics (10 μm) and fibrous polyamide microplastics (100 μm) by aluminosilicate filter media modified by cationic surfactant. The results of a full-scale wastewater treatment plant in eastern China using MBR and an oxidation ditch in a parallel system showed that membrane filtration is more efficient in microplastics removal. MBR removed 99.5% of microplastics from influent, whereas oxidation ditch removed 97% (Lv et al., 2019b). MBR effectively removes small microplastics (<100 m) and nanoplastics. However, the smaller size of the microplastics makes the filtration process more complex and expensive due to membrane scaling and fouling (Malankowska et al., 2021), which demands improvements in this aspect. Although DAF had a higher removal efficiency for low-density microplastics, (Sol et al., 2020) the overall removal efficiency was insignificant even when combined with flocculants and surface modifiers (Sturm et al., 2021). The microplastics removed by DAF varied from 43.8% to 68.9% (Wang et al., 2021). Sarkar et al. (2021) used a pulse clarifier to remove more than 85% of microplastics from drinking water treatment plants.

According to the preceding discussion, most of the microplastics in WWTPs are removed/retained by skimming, sedimentation, and tertiary filtration. However, these technologies are not originally developed for microplastics, allowing a large portion of microplastics to water bodies. Furthermore, sludge retains more microplastics and serves as a source of microplastics by releasing microplastics into the environment during conventional sludge management practices such as landfilling (Miri et al., 2021). The technologies designed explicitly for microplastics removal are still in the preliminary stages of research. Developing new technologies and/or upgrading existing techniques to address microplastics released into the environment from WWTPs might be a viable option. Ma et al. (2019) studied the polyethene-based microplastics removal from drinking water systems using coagulation (Fe-based coagulant) followed by an ultrafiltration process. They found that the individual conventional coagulation process (Fe-based coagulant) had a nonsignificant microplastics removal efficiency of 13%, indicating that the individual coagulation process was insufficient for microplastics removal. However, a combination of Al-based coagulant and Polyacrylamide (PAM) enhanced the coagulation performance. It increased the removal efficiency from 13% to 91% for particles size <0.5 mm and reduced membrane fouling during ultrafiltration. Recently, Zhou et al. (2021) reported that polyaluminium chloride (PAC) was better than ferric chloride ($FeCl_3$) in the removal of polystyrene (PS) and polyethene (PE) microplastics. Ye et al. (2021) fabricated two types of bubble-propelled iron oxides-MnO_2 core-shell micromotors and tested them under the external magnetic field to remove microplastics. The Fe_2O_3–MnO_2 micromotor separates >10% of suspended microplastics from the polluted water in 2 h. Wang et al. (2020f) compared the filtration characteristics of four agricultural waste-based biochar and sand to immobilize uniformly graded microplastics spheres. The findings showed that 10 μm diameter microplastics spheres were immobilized to a greater extent (60–80%) on all four biochars than on similar grain-sized sand filters. After examining SEM images, they proposed three mechanisms of immobilization of microplastics spheres on biochar: 'Stuck', 'Trapped', and 'Entangled'. Because microplastics are hydrophobic, they can be removed using the froth flotation process. Microplastics with higher density, larger size, and lower concentration were removed from the waste stream by froth flotation. Cationic species such as potassium, sodium, and calcium did not affect the removal of microplastics. At an aeration volume of 5.4 mL/min and Al^{3+} concentrations (froth dose) of 28 mg/L, froth flotation removed 100% microplastics (Zhang et al., 2021b).

6.2. *Electrocoagulation*

Currently, electrocoagulation has been used to effectively remove dyes, heavy metals, and clay particles with >80% removal efficiency (Perren et al., 2018). At an initial pH between 3 and 10, about >90% of polyethene microbeads were removed from artificial wastewater, and maximum removal efficiency was observed to be 99.24% at a pH of 7.5 (Perren et al., 2018). Shen et al. (2022) evaluated the efficiency of electrocoagulation for the recovery of four diverse microplastics. They observed a maximum recovery of 93.2% for polyethene, 91.7% for polymethylmethacrylate, 98.2% for cellulose acetate, and 98.4% for polypropylene at pH 7.2. The aluminium anode performed better than the iron anode, with an overall >80% recovery of microplastics in the pH range of 3–10. The electrocoagulation exhibited appreciable removal efficiency for fibre microplastics than granular microplastics. The microplastics removal efficiency positively correlated with electrolyte concentration and applied voltage density. The optimized conditions were the electrolyte concentration of 0.05 M, pH of 7.2, applied voltage density of 10 V, and Al anode. Elkhatib et al. (2021) employed electrocoagulation to remove commercial polyester microplastics from synthetic solutions and wastewater samples. In synthetic solution, the recovery of polyester microplastics was about 99% at pH 4 and 7 and current densities of 2.88 and 8.07 mA/cm^2. In wastewater samples, 96.5% of microplastics, 92.2% of chemical oxygen demand, and 88.8% of thermotolerant coliform colonies were removed. A 30 min of electrocoagulation using aluminium electrodes followed by centrifugation removed >90% of microplastics floc (Kim and Park, 2021). Microplastics characteristics such as size, shape, and density, as well as electrocoagulation process components such as initial pH, coagulant dose, coagulant nature, and flocculant aids, had a significant impact on the electrocoagulation process's efficiency and must be optimized for the practical application of microplastics removal.

6.3. Sol-gel process

The sol-gel process is a chemical method for producing a highly crosslinked solid of an inorganic-organic macromolecule by sequential hydrolysis of the precursor in acidic or basic media, followed by polycondensation of the hydrolyzed products. N-alkyl substituted chlorosilanes are commonly used as precursors due to their high reactivity with water (Hurkes et al., 2014). In 2017, Herbort and Schuhen proposed a host-guest relationship for removing microplastics from water by agglomeration utilizing silicon-based precursors. The aforementioned technique included the fabrication of an inclusion unit (inorganic-organic macromolecules) and then a capture unit, which was then combined to create an inclusion compound and alkoxy silyl presence, which provided the necessary 3-D network. The sol-gel process provides structured composite silica gels, which are used to capture micro beads that can be separated by simple separation techniques such as a sand trap. These capture units can be further utilized for energy generation. The hydrophobic stressors (microplastics) trapped in hybrid silica gel increased their volume, allowing easy filter separation compared to granular activated carbon. Herbort et al. (2018) synthesized diverse bioinspired alkoxy-silyl, functionalized molecules and subsequently generated agglomerate via sol-gel process, which was 666 times more in the volume of the original particles that allowed cost-effective separation. The various alkyl trichlorosilane exhibited different characteristics for localization and fixation of microplastics based on alkyl groups as they significantly influenced the reaction rate and agglomeration behaviour. The intermediate alkyl group between 3 and 5 carbon atoms was best suited for polyethene (PE) and polypropylene (PP) mixture removal; however, long alkyl groups (8 or >8 carbon atoms) were ineffective to localized microplastics, which ultimately caused a reduction in microplastics removal (Sturm et al., 2020). Further, microplastics chemical composition, surface chemistry, and physical interaction with the organosilanes play an essential role in the removal process. The removal efficiency of microplastics was 76.4% and 46.3% for n-butyltrichlorosilane and iso octyltrichlorosilane, respectively; however, large amounts of residues were left dissolved in water. PE-X (abcr eco Wasser 3.0 P E-X) can be employed on a technical scale since no dissolved residue was found, removing the risk of organosilanes entering the environment (Sturm et al., 2021).

6.4. Dynamic membranes and membrane bioreactor

In recent years, dynamic membranes gained significant attention due to low energy consumption and no demand for extra chemical because pollutants in wastewater itself make the filter layer which is easy to clean and exhibits potential for the removal of low-density microplastics (Lu et al., 2016; Li et al., 2018b). Li et al. (2018b) evaluated the practicability of dynamic membranes for microplastics removal and observed that the dynamic membrane was formed in a very short time on the 90 μm supporting mesh and was able to operate in low transmembrane pressure (70 mm–180 mm of water head) and total filtration resistance (2.89×10^{-9}/m to 6.52×10^{-9}/m). Microplastics removal using dynamic membrane technology could be energy-efficient. However, filter blockage, construction, and operational cost should be thoroughly evaluated when utilizing an extra unit for microplastics removal.

Membrane bioreactor (MBR) has shown superiority over conventional activated sludge process (ASP). MBR requires less operation space, minimum sludge generation, significant improvement in the overall efficiency of wastewater treatment, and is easy to combine with the conventional biological treatment process and can be a potential replacement for secondary sedimentation and tertiary filtration. Lares et al. (2018) observed better removal of microplastics by MBR (99.3%) compared to the conventional activated sludge process (98.3%). A combination of upflow granular anaerobic sludge blanket (UASB) and anaerobic membrane bioreactor (AnMBR) removed 94% of microplastics with 87% of fibres and 100% of particles (Pittura et al., 2021). Lv et al. (2019b) compared to the membrane bioreactor efficiency with oxidation ditch in a full-scale WWTP of Eastern China. The influent 40% microplastic were of size >500 μm and 29% between 62.5 and 125 μm. In terms of plastic mass, the membrane bioreactor exhibited 99.5% removal efficiency than the oxidation ditch (97%), while based on the numbers for microplastics, it was 82.1% for the former and 53.6% for the latter. Microfiltration membrane modules in MBR eliminated considerable amounts of microplastics >300 μm in size, which are the most common microplastics in surface water (Auta et al., 2017a). The integration of submerged flat-sheet UF membranes with MBR retained up to 99.4% of influent microplastics (Talvitie et al., 2017; Lares et al., 2018). However, MBR process-based sludge retains a large amount of microplastics that demand further treatment, which increases overall treatment costs. Membrane fouling is another major hurdle that hurts membrane fibres and increases maintenance costs. Maliwan et al. (2021) operated sequencing-batch MBRs for the diverse feed of microplastics for 124 days. They observed that the presence of microplastics decreased floc size in sludge, floc hydrophobicity, and floc negative zeta potential. The decrease in molecular size and increase in the extracellular polymeric substance (EPS) concentration further facilitated the divalent cation (Ca^{2+} and Mg^{2+}) uptake by microplastics. In contrast to the control, a 4-month operation of sequencing-batch MBRs did not experience severe cake fouling.

7. Degradation of microplastics

Microbial degradation of microplastics in different environments is an integrated process with physicochemical factors. Microplastics serve as a supporting material for the growth as well as carbon and energy source for microbes. Microbes in their pure culture (bacteria and fungi), and consortia are commonly studied for microplastics degradation. The methods reported for microplastics degradation assessment can be grouped into three categories: (1) those focus on the elimination of small molecules; (2) those focus on chemical changes (hydrophobicity, functional groups) in the polymer structure; and (3) those focus on physical changes (tensile strength, surface morphology, crystallinity, etc.) in material properties. Mass loss, carbon dioxide evolution and Gel Permeation Chromatography methods assess the degradation based on bond cleavage. Nuclear magnetic resonance (NMR), infrared (IR) spectroscopies, contact angle assess biodegradation by observing changes in chemical functionality. Dynamic Mechanical Analysis (DMA), thermal analysis, surface analysis (scanning electron microscopy (SEM) and atomic force microscopy (AFM)) assess biodegradation based on changes in materials properties (Chamas et al., 2020).

7.1. Degradation by bacteria

Bacteria in pure culture at laboratory condition is commonly explored for microbial-mediated degradation of microplastics due to easily probe metabolic pathways, impacts of environmental variables and changes in microplastics during the degradation process. Auta et al. (2017b) observed the degradation ability of *Bacillus cereus* and *Bacillus gottheilii* for polyethylene (PE), polyethylene terephthalate (PET), polypropylene (PP), and polystyrene (PS) isolated from the mangrove sediments in Peninsular Malaysia. The calculated weight loss percentages of the microplastic particles by *B. cereus* after 40 days were 1.6%, 6.6%, and 7.4% for PE, PET, and PS, respectively. *B. gottheilii* recorded weight loss percentages of 6.2%, 3.0%, 3.6%, and 5.8% for PE, PET, PP, and PS, respectively. The common genus screened for microplastic degradation included *Bacillus, Rhodococcus*, *Bacillus gottheilii*, *Enterobacter asburiae*, *Bacillus subtilis*, Chelatococcus, *Comamonas acidovorans*, *Pseudomonas*, *Paenibacillus amylolyticus*, *Ideonella sakaiensis*, *Stenotrophomonas maltophilia*, *Spingobacterium multivorum*, *Lysinibacillus* etc., however, weight loss rate of microplastics during degradation is only around 15% indicating that microplastics are not very biodegradable by a single pure bacterial strain (Matjašič et al., 2021; Yuan et al., 2020). Species of *Pseudomonas, Bacillus, Brevibacillus*, and *Streptomyces* genus showed high efficiency against different plastic polymers (Matjašič et al., 2021; Ali et al., 2021). The consortiums of bacterial microorganisms increased biodegradation efficiency either by playing a direct role in biodegradation or removal of toxic intermediates produced during degradation. Further, metabolic cross-feeding and the production of metabolites that drive co-metabolic breakdown are two additional ways in which individual members of a microbial consortium might indirectly enhance biodegradation (Yuan et al., 2020). The consortium consisting of species of *Pseudomonas* and *Bacillus* genus have widely been explored for microplastics degradation (Matjašič et al., 2021). Recently, consortia of *Stenotrophomonas* sp. and *Achromobacter* sp. exhibited LDPE beads degradation ability (Dey et al., 2020).

7.2. Degradation by fungi

Fungi (especially white-rot and brown-rot fungi) degrade microplastics in more efficient way than bacteria due the deep penetration of mycelia into the surface of polymeric substances and release high amount of extracellular enzymes (such as lignin peroxidase, manganese peroxidase, versatile peroxidase and multi-copper oxidase laccase) to degrade polymers into their oligomers, dimers, and monomers (Ali et al., 2021; Rose et al., 2022). *Aspergillus* sp. and *Penicillium* sp. are potential strains and *Aspergillus flavus* is the most popular and successful strain for microplastics biodegradation, respectively. Some other species are *Fusarium solani* (Zahra et al., 2010), *Trichoderma viride* (Munir et al., 2018), *Zalerion maritimum* (Paço et al., 2017), *Eupenicillium hirayamae*, *Phialophora alba*, *Paecilomyces variotii* (Ojha et al., 2017) etc. identified for the microplastics degradation. Hydrophobicity and chemical structure (containing non-phenolic aromatic rings, ether linkages, and a carbon skeleton that is oxidised during lignin breakdown) are two ways in which lignin resembles plastic (Ali et al., 2020). These properties enable laccase and manganese peroxidase enzymes to easily degrade polyethylene and polypropylene due to structural similarity to lignin (Jeyakumar et al., 2013). However, few investigations on the fungal-mediated degradation of microplastics have been conducted, demonstrating the difficulties in finding fungal strains with good microplastics-degrading activity by ectopic screening (Yuan et al., 2020).

7.3. Degradation by invertebrates

Recently, biodegradation of microplastics specially PS in the guts of invertebrates has gained significant attention. The larvae of *Tenebrio molitor* (yellow mealworms), *Zophobas atratus* (superworms), *Plodia interpunctella* (Indian mealmoths), *Galleria mellonella* (greater waxworms), *Achroia grisella* (lesser waxworms) ingested microplastics and biodegraded it in their guts. *T. molitor* and *Z.atratus* larvae (species of darkling beetle) biodegraded PS and LDPE in a matter of hours. *T. molitor* larvae could also biodegrade polyvinyl chloride (PVC), polypropylene and hydrolyzable bioplastic polylactic acid (PLA). Yang et al. (2015) reported rapid biodegradation of PS in the larval gut of *T. molitor* Linnaeus. The higher PS degradation ability within the gut of *T. obscurus* (26.03%) than *T. molitor* (11.67%) was demonstrated by Peng et al. (2019). Yang et al. (2021b), suggested the intestinal digestive system could perform LDPE depolymerization in *T. obscurus* via *Enterobacteriaceae, Enterococcaceae* and *Streptococcaceae* (at bacterial family's level) and *Spiroplasma* sp. and *Enterococcus* sp. (at the genus level). Biodegradation of EPS and LDPE by larvae of *Z. atratus* supported gut microbe-dependent LDPE and EPS biodegradation (Peng et al., 2020b). However, Yang et al. (2021a) confirm the Polypropylene (PP) biodegradation in both *T. molitor* and *Z. atratus* larvae via gut microbe-dependent depolymerization with diversified microbiomes. Polyethylene film (PE, 100 mg) biodegradation in the gut of the larvae of *P. interpunctella* (waxworms, or Indian mealmoths) was executed by two bacterial species *Enterobacter asburiae* YT1 (6.1 $\pm$ 0.3%) and *Bacillus* sp. YP1 (10.7 $\pm$ 0.2%) over a 60-day of incubation period (Yang et al., 2014). Brandon et al. (2018) reported that the larvae of *T. molitor* conversed up 49.0 $\pm$ 1.4% (mass balances) of the ingested PE into a putative gas fraction (carbon dioxide). PVC depolymerization/biodegradation by *T. molitor* larvae involve gut microbes. Further *T. molitor* larvae can undertake extensive depolymerization/biodegradation of PVC microplastics but only a little amount of mineralization (Peng et al., 2020a). Lou et al. (2020) revealed that Bacillus and Serratia were significantly associated with the PS and PE biodegradation in the gut of larvae of *Galleria mellonella*. The second generation of PE-WC (wax comb as co-feed) fed larvae of *A. grisella* efficiently degrades PE at par with first generation counterparts (Kundungal et al., 2019). Bombelli et al. (2017) reported fast bio-degradation of PE by larvae of the waxmoth *Galleria mellonella*, producing ethylene glycol. First time, *Achatina fulica* (Land snails) ability to degrade PS were tested by (Song et al., 2020) and observed that the gut microorganisms (family *Enterobacteriaceae*, *Sphingobacteriaceae*, and *Aeromonadaceae*) were associated with PS biodegradation.

8. Mitigation strategies for microplastic pollution

Plastic waste disposal has become a significant concern due to inadequate legislation and lack of inefficient disposal techniques. To minimize the adverse effects of microplastics, we must efficiently manage plastic waste in an eco-friendly and cost-effective manner. Because plastic pollution affects neighbouring countries and international waters, mitigation efforts to decrease plastic pollution must be stringent. Plastic removal from aquatic and terrestrial systems could be considered clean-up activities, but it is inadequate for this widespread problem. Some of the mitigation strategies are suggested to tackle microplastic pollution.

1. A well-established efficient management system adheres to the 5 R's: reuse, reduce (reducing the production of plastic waste at the source), recycle (reducing the number of plastics released on a daily basis to reduce their deleterious impacts on the environment), recover (waste conversion to energy), refuse, and finally, eco-friendly disposal of plastic waste would result in an environment, free of plastic.

2. Microbeads in cosmetics and other personal care products such as toothpaste, face wash, and shampoos should be banned immediately (Chatterjee and Sharma, 2019). Another way to reduce plastics in the environment is to incur a tax on plastic products such as microbeads in the cosmetic industry, daily care products, and plastic bags for groceries.
3. Further, emphasis should be given on consumer education and awareness, reducing plastic discharge in wastewater treatment plants, improving plastic product life-cycle and end-of-life management, and national and international governance (Prata et al., 2019).
4. All industries should adopt extended producer responsibility (EPR) because it is a public policy tool that makes every producer legally and financially responsible for mitigating the environmental impacts of their products throughout their life cycle stages (Eriksen et al., 2018).
5. Thermoplastics such as polyethene terephthalate (PET), polyethene (PE), and polypropylene (PP) all have the high potential to be recycled mechanically (Ogunola et al., 2018).
6. The biggest menace in plastic pollution is the use of single-use plastic bags. If plastic bags are completely banned, or their use is restricted, and some user fees are applied to their use, then definitely there will be a reduction in their usage and accumulation in the environment.
7. Ecolabelling could be another measure for controlling plastic pollution. The eco-labelled products are eco-friendly, recyclable, and consume less energy (Ogunola et al., 2018). The main aim behind the ecolabelling of plastic products is to create awareness among the consumers so that they buy products that pose no harm to the environment on disposal, which would create environmentally conscious behaviour in them. Secondly, this scheme would introduce biodegradable plastics (that could be composted) (Ogunola et al., 2018). Bioplastics have gained popularity in recent years due to their eco-friendliness and ability to be degraded by microorganisms. Petroleum, starch, vegetable fats, and oils can all be used to produce these bioplastics (Ogunola et al., 2018). Chitosan, crustacean shells, polysaccharides, and insect cuticles are other materials that can be used to produce bioplastics that are biodegradable in the environment within 2 weeks (Kumar et al., 2021b).
8. A few pieces of research have appeared that have used bacteria, fungus, and some worms to degrade plastics (Karbalaei et al., 2018). These include a variety of *Pseudomonas*, *Flavobacterium*, *Arthrobacter*, and *Agromyces* species, most of which live in soil or sediment (Bassi, 2017). Bombelli et al. (2017) found that the larvae of the wax moth (*Galleria mellonella*) decomposed PE, producing ethylene glycol as the end product. Hadad et al. (2005) demonstrated that in the presence of UV light, gram-positive thermophilic soil bacteria (*Brevibacillus borstelensis*) degrade branched-chain low-density polyethene. In another study, Lwanga et al. (2018) found that bacteria isolated from the earthworm's gut (*Lumbricus terrestris*) degraded low-density polyethene (LDPE). Scientists have recently attempted to degrade microplastics using various microorganisms (bacteria and fungi) such as *Streptomyces setonii*, *Pseudomonas aeruginosa*, *Rhodococcus Ruber*, *Pseudomonas stutzeri*, *Streptomyces badius*, *Aspergillus niger*, *Aspergillus flavus*, *Fusarium lini* (Tiwari et al., 2020). A mutant enzyme has been found by scientists that breaks down the plastic bottles in few days than it takes to do the same in oceans (Lamichhane et al., 2022). Biodegradable plastics can be produced from microalgae which could substitute synthetic plastics (Roy et al., 2022).
9. There are various social platforms such as Plastic Pollution Coalitions, Plastics for change, Plastic Oceans, Surfers Against Sewage, Greenpeace, By the Ocean We Unite, One More Generation, One Green Planet, Surf Rider Foundation, and Earth Guardians. They are working on the issue of microplastic pollution and contributing substantially (Chatterjee and Sharma, 2019). The World Economic Forum and the Ellen MacArthur Foundation brought a joint initiative in terms of The New Plastic Economy, which proposed redesigning the manufacture of plastic products. Redesigning means that these products should be biodegradable so that they are not harmful to the environment when disposed of (Mehmandost et al., 2019). In 2015, the United States approved the "Microbead-Free Waters Act, 2015," which states that plastic microbeads should not be added to products. This legislation came into force in July 2017 for manufacturers and in July 2018 for retail sales (Masia et al., 2020). With the target year of 2016, the Netherlands was the first to produce microbead-free cosmetic products. With this in perspective, the Delhi Plastic Bag Act, (2000) was enacted to stop consuming foods in recycled plastic bags and dispose of nonbiodegradable waste in toilets, highways, and sites (Bundela et al., 2010). In 2002, India similarly prohibited the use of ultra-thin plastic bags. In 2017, India also banned disposable plastics in Delhi and the National Capital Region. In India, just 7% of total plastic gets recycled, and 65% of plastic waste ends up in landfills. So, developing countries like India should focus on material and energy recovery rather than landfilling because landfilling produces toxic leachate, contaminating soil and groundwater. Looking into international efforts to manage plastic waste, in 2008, Rwanda became a pioneer in banning disposable plastics among developing nations and has been declared the cleanest nation on the globe. Indonesia introduced a new policy in one of its cities to collect used plastic bottles for free bus rides across the city. South Korea has launched an emerging practice of "precycling" (bringing own mugs and reusable bags) in supermarkets, grocery stores, and cafes to curb the consumption of disposables (Santhosh and Shrivastav, 2019). In 2011, the governments of Rwanda, Kenya, Uganda, South Sudan, and the United Republic of Tanzania signed the East Africa Community Polythene Material Control Bill to halt the illegal movement of plastics in cross-borders and promote sustainable packaging substitutes (Santhosh and Shrivastav, 2019). European Union introduced Directive 85/339/EEC to address the issues of production, use, recycling, and refilling containers for consumption and disposal of post-consumer plastic waste. The directive such as Directive EU 2015/720 (2015) and the amendment of Directive 94/62/EC define measures how to tackle the problem regarding the consumption of lightweight plastic carrier bags. This Directive obliges member states to reduce the per capita consumption of plastic bags to 90% by the end of 2019 (Santhosh and Shrivastav, 2019). The Plastic Waste Management Rules, 2016 (in India) stipulate that urban local bodies (ULBs) should ban less than 50 μm thick plastic bags and not allow the usage of recycled plastics for packing food, beverage, or any other eatables (Manuja 2020). Single-use grocery bags, shopping bags, and plastic bottles have been outlawed in California (USA), plastic packaging materials in Massachusetts (USA), non-biodegradable tableware in France, and plastic-containing cosmetic products in Canada (Kumar et al., 2021b). The United Nations Environment Programme (UNEP) initiated a global campaign to eliminate primary sources of plastic waste by 2022 (Llorca et al., 2020).
10. The most basic and important step in mitigating microplastics pollution is to develop a sanitary waste management system i.e. sanitary landfills, and organise waste collection. Secondly, leakage to the environment can be reduced significantly by banning and taxing most frequently littered items. Microplastics can be put into the Guppyfriend washing bags in order to reduce microfibre shedding during washing. After washing, the microfibres retained in the bags can be disposed of in the residual waste bin. The laundry bag produced from a plastic woven polyamide is

user friendly. An example of an organisational innovation is Loop in which the overall environmental footprint is reduced and the business model for a delivery system is changed to avoid single-use packaging. Loop delivers online orders to households in reusable containers that are collected afterwards, cleaned and refilled (OECD, 2022).

In nutshell, in order to reduce the ever increasing microplastic pollution, stringent national and international laws should be implemented, change in human lifestyle and behaviour is the utmost need of the hour in order to deal with this waste. The recycling and recovery are some of the basic things which if emphasized upon can really bring a drastic change in handling this pollution as a whole.

9. Bottleneck and current challenges

It is crucial to realize that not all plastic products are the same, and not all have the same service life to comprehend the life cycle of plastic products. When plastic reaches the end of its useful life, it becomes waste. Primary microplastics are made tiny in size for use in personal care products, medicines, and industries. In contrast, secondary microplastics are formed when larger plastic objects are gradually fragmented by mechanical, chemical, and biological processes. The linear economy model associated with single-use plastic contributes a significant amount to plastics in the natural environment, such as oceans or landfills. According to a study (Cordier and Uehara, 2019), the production of commercial plastics and products, their distribution, and consumption cause about $13 billion of damage to the marine environment each year, although approximately 95% (equal to $80–120 billion) of all packaging plastic material is discarded.

This research also predicted that around 0.7–1.0% of total global GDP in 2017 (approximately 492–708 billion euros) is required to clean the approximately 135 Mt of waste plastic from oceans, representing just 15% of total ocean plastics for the years 2020–2030 (Watt et al., 2021). So, the cost is always a huge challenge associated with removing plastics and microplastics from the natural environment. Microplastic is a global threat due to its quantity, persistence, and widespread distribution with potential geophysical and biological impacts (Galloway and Lewis, 2016). Microplastics are a cause of concern due to their size range coinciding with the ideal particle size swallowed by creatures of the marine food web. Detritus, suspension, and filter feeders easily ingest microplastics, resulting in bioaccumulation, biomagnification, and trophic transfer to the highest food level consumers (Galloway and Lewis, 2016). Furthermore, microplastics serve as a vector or carrier for heavy metals (such as Cr, Cd, Cu, Ni, Pb, etc.) and organic contaminants (Pesticides, Polychlorinated biphenyl, Perfluorinated compounds, etc.). The adsorption of these contaminants on microplastic is a complex process that associates the physico-chemical properties of microplastics and factors associated with microplastic surface properties, including the biofilm attachment and environmental ageing process (Hou et al., 2021). Moreover, the uptake and transfer of contaminant-loaded microplastics still require intensive exploration to understand their behaviour in the living body. Moreover, effective sampling and identification protocols for toxic chemicals related to microplastic pollution are in the developing stages (Yu et al., 2018). The trophic transfer of microplastics to predators has been investigated in laboratory conditions, either alone or in combination with xenobiotics, which is far from real-world scenarios (Arienzo et al., 2021). Every stage of microplastic analysis, including sampling, extraction, separation, and identification, takes time, indicating a considerable barrier to large-scale monitoring. Furthermore, the methodology used in these processes varies; therefore, the results are not always comparable. Because it is difficult to collect enough microplastic particles, especially in complex samples for chemical analysis, small microplastics have low detection frequencies and high detection limits (Yu et al., 2018). For instance, quantifying microplastics in a terrestrial environment is challenging because of sorting them from the huge amount of biomass. However, methods used to characterize and quantify the microplastics in sediments and water samples were modified and applied to soil samples. Although various techniques for extracting microplastics are fast, cheaper, and efficient. However standard approach is still missing (Miri et al., 2021). Visual identification provides a rapid screening of microplastics based on their type, shape, size, and colour, which is often enough to distinguish plastic particles in solid matrices ranging from 5 to 0.25 mm (Rocha-Santos and Duarte, 2017). However, below 0.25 mm, their identification is based on colour and shape, leading to misclassification of the microplastics because of our inability to differentiate them inside the bulk sample (Lavers et al., 2016). However, they can be identified using an optical microscope and SEM. However, a pre-treatment is necessary to eliminate organic materials from microplastic, which might also result in changes in the morphological features of plastics, such as color or size, leading to microplastic misidentification (Ruggero et al., 2020). Surprisingly, various factors influence the effectiveness of identification, such as plastic additives, which hinder the identification of polymers like pigments, and interfere during identification due to their small size and widespread in the environment (Yu et al., 2018). Furthermore, microplastics look similar to non-polymeric and sediment materials, making the visual procedure more time-consuming and error-prone (Miri et al., 2021). Other approaches, such as infrared spectroscopy, thermogravimetric analysis, differential scanning calorimetry, and Raman spectroscopy, may detect particles as small as 20 μm, but they are costly and restricted to certain applications. Recently, the use of micro-FTIR (FTIR linked to microscope) has increased as it facilitates sample mapping, characterization of multiple polymers, and identification of irregular-shaped microplastic. However, it is quite an expensive and time-consuming process (Miri et al., 2021). There are no standard authorized protocols to determine small microplastic (<1 micron) in natural environmental samples, and the use of large quantities of microplastics in ecotoxicological investigations may only be described as a proof of concept as there is not enough information to estimate the potential risk (Huvet et al., 2016). Furthermore, to minimize misunderstanding of non-ecologically realistic results, microplastic concentrations in exposure experiments should be closer to environmentally realistic amounts. Previous research on microplastic contamination has shown their potential toxicity; nevertheless, additional information is needed to understand their toxicity and harm to human health. Previous reports on microplastic pollution demonstrated their potential toxicity; however, more data should be collected to clarify their toxicity and threat to human health. Recently, a research reported the presence of microplastics in the human placenta (Ragusa et al., 2021) but their presence in the placenta and associated potential harm to human health remains poorly understood. The trophic transmission of microplastics is the major challenge in understanding the uncovered possible health hazards. On the practical ground, the major issue in preventing microplastic contamination in water bodies is the absence of technology that successfully retains microplastic at wastewater treatment plants. Generally, microplastic studies reported the presence of microplastics in the final effluent; nevertheless, the details on microplastic removal at each step of the wastewater treatment plants (WWTPs) are still inadequate. The majority of microplastics in WWTPs are removed through screening, grit removal, grease removal, skimming, sedimentation (preliminary and primary treatment), activated sludge (secondary treatment), and membrane bioreactor (MBR), rapid sand filter (RSF), disc filter (DF), dissolved air flotation (DAF), ultrafiltration (UF), reverse osmosis (RO), gravity filter, etc. (tertiary treatment). Moreover, the current technologies used in the wastewater treatment plant are not designed to effectively remove microplastic, which allows a large portion of microplastics to be discharged into water bodies. Further, sludge retains more microplastics and acts as a microplastics source as it releases microplastics into the environment during conventional sludge management practices such as landfilling. The technologies specially targeted for microplastic removal are still in the preliminary stages of research (Sun et al., 2019). Furthermore, they can be costly, difficult to integrate into existing facilities, and only employed when high-quality standards are necessary, e.g., membrane bioreactors. The membrane bioreactor uses cross-flow filtration to remove small microparticles but demands high operational energy, which increases the operational cost. The combination of different unit operations can be effective for microplastic removal but increases the complexity and capital cost of microplastic elimination in WWTPs. Investigations towards boosting enzyme degradation efficiency and developing bioreactors for microplastic enzymic/biotic depolymerization are still at the laboratory scale. Furthermore, most bioplastic degradation studies have not considered the formation of microplastics (Miri et al., 2021). Recently, the United Nations Environment Program (UNEP) presented a report which assessed the substantial economic losses caused by plastic contamination in marine ecosystems (Smith 2014). Microplastic contamination in the aquatic environment was also highlighted in the Nature journal, which called for more attention to these pollutants and their toxicity (Marris 2014). Fortunately, along with global climate change, ozone depletion, and ocean acidification, microplastic pollution was identified as one of the biggest environmental challenges in 2015 (Yu et al., 2018). However, various laws and policies to control plastic pollution are not enough to regulate plastic production, its consumption, and finally, proper disposal, causing a significant increase in microplastics in the natural environment. Recently, the COVID-19 pandemic has changed the dynamics of microplastic exposure to humans and created a new source of a huge amount of microplastics in the environment in a short time. As a result, there is a greater chance of interaction, ingestion, and hazardous effects across food webs, which has become a new challenge to explore the significant and potential impact of reusable face masks on humans and the environment (Shruti et al., 2020b).

10. Conclusions and future perspectives

The degradation of plastics has become an issue of concern because it produces microplastics that are a potential risk to the ecosystem, humans, animals, and plants. Another area of research could be an evaluation of the potential effects of advanced treatment on microplastic levels in treated water. It has been found that wastewater treatment plants (WWTPs) release microplastics into the environment. Therefore, detecting microplastic occurrences in WWTPs is highly important for their effective control. Smaller microplastics of size smaller than 20 μm should be the focus of future studies as these being abundantly present in water, enter the circulatory system of aquatic organisms. Removal of microplastics from wastewater should be the target area as wastewater treatment plants act as pathways for entering microplastics into natural aquatic systems.

Future research should emphasize the fate of plastics in various environments and technology development to reduce plastic pollution. More research is required on the formation and degradation of nanoplastics to assess their fate and environmental risks. There is a crucial need for appropriate sampling, identification, and removal methods to detect microplastic pollution. The impact of microplastics on humans is not adequately understood, and this field needs to be explored further. Reuse, refuse, reduce, recycle, and rethink could be key factors in reducing microplastic pollution. Plastic products that are harmful to the environment should be banned or taxed. The ban offers comprehensive protection against plastic microbead pollution in the nations. Microplastic production and consumption could be reduced by improving design and using alternative materials, resulting in long-lasting products that can replace plastics. A deeper understanding is required to identify and effectively remove hotspots of plastic pollution in water. Scientific research should be conducted to minimize their discharge into rivers. Standard scientific microplastic sampling and extraction techniques should be explored to monitor risk assessment.

The potential toxicity of contaminants adsorbed on microplastic in the environment needs to be investigated further to understand point/non-point sources, particle interactions, and transport mechanisms for modelling and identifying contaminants source and accumulation in the river environment, including river toxicity and biodiversity. Ultimately, reducing consumption, production, and effective waste management is an effective strategy to reduce microplastic pollution. The transportation of microplastics in sediments should be investigated to improve risk assessment in aquatic environments. Microplastic monitoring is essential in areas where the seafood is harvested to develop strategies for future management. The cosmetic and beauty products industries have already begun to phase-out microplastics and replace these additives with more environmentally benign alternatives. Some countries have even banned the use and sale of cosmetics containing microplastics. Positive human behavioural change is highly recommended as it will provide a long-lasting solution to the microplastic problem.

Declaration of competing interest

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

Data availability

No data was used for the research described in the article.

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