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REVIEW PAPER

Micro/nano-plastics occurrence, identification, risk analysis and mitigation: challenges and perspectives

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Abstract Micro/nanoplastics (MP/NPs) are emerging global pollutants that garnered enormous attention due to their potential threat to the ecosystem in virtue of their persistence and accumulation. Notably, United Nations Environment Programme (UNEP) yearbook in 2014 proposed MPs as one among ten emergent issues that the Earth is facing today. MP/NPs can be found in most regularly used products (primary microplastics) or formed by the fragmentation of bigger plastics (secondary microplastics) and are inextricably discharged into the environment by terrestrial and land-based sources, particularly runoff. They are non-degradable, biologically incompatible, and their presence in the air, soil, water, and food can induce ecotoxicological issues and also a menace to the environment. Due to micro size and diverse chemical nature, MP/NPs easily infiltrate wastewater treatment processes. This communication reviews the current understanding of MP/NPs occurrence, mobility, aggregation behavior, and degradation/assimilation in terrestrial, aquatic (fresh & marine), atmospheric depositions, wetlands and trophic food chain. This communication provide current perspectives and understanding on MP/NPs concerning (1) Source, occurrence, distribution, and properties (2) Impact on the ecosystem and its services, (3) Techniques in detection and identification and (4) Strategies to manage and mitigation.

Keywords Plastic pollution · Microfiber plastics · Ecological risk · Wastewater treatment · Circular economy

Abbreviations

AS	Acrylonitrile styrene copolymer
Cd	Cadmium
Co	Cobalt
Cr	Chromium
$FeSO_4$	Iron (II) sulfate
H_2O_2	Hydrogen peroxide
HBrCDs	Hexabromocyclododecanes
HDPE	High density polyethylene
HNO_3	Nitric acid
KI	Potassium iodide
KOH	Potassium hydroxide
LDPE	Low density polyethylene
LC-MS	Liquid chromatography-mass spectrometry
NaCl	Sodium chloride
NaI	Sodium iodide
PA	Polyamide

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PAH	Polycyclic aromatic hydrocarbon
PBB	Polybrominated diphenyl
PBDEs	Polybrominated diphenyl ethers
Pb	Lead
PC	Polycarbonate
PCB	Polychlorinated biphenyl
PE	Polyethylene
PE-PP	Polyethylene and polypropylene copolymer
PEST	Polyester
PET	Polyethylene terephthalate
PMMA	Polymethyl methacrylate
PP	Polypropylene
PS	Polystyrene
PU	Polyurethane
Py–GC/MS	Pyrolysis–gas chromatography/mass spectrometry
PVC	Polyvinyl chloride
Ti	Titanium
TiO_2	Titanium oxide
WWTP	Wastewater treatment plant
Zn	Zinc
$ZnCl_2$	Zinc chloride

1 Introduction

Plastics have diversified to infiltrate all human activity and utilities by replacing wood and metals due to their affordability and versatility. The world's first synthetic plastic was produced by Leo Baekeland in 1907. The manufacture of plastics and use was phenomenal between 1950 and 2015, with terrestrial emissions accounting for 80% of plastic waste globally (Geyer et al. 2017). Plastic pollution has become a ubiquitously distributed and severe transboundary threat to natural ecosystems (Thompson 2015). Packaging materials (39.5%), building materials (20.1%), fishing gears (10%), automotive parts (8.6%), electronic utilities (5.7%), and agro-industry components (3.4%) account for the majority of plastic depositions, with the remainder consisting of household, medical waste, and sports equipment (Horton et al. 2017; Plastics Europe 2017). Recent Covid-19 pandemic upsurged the consumption of single-use plastics, namely personal protective equipment (PPE) kits, facemasks, and gloves, contributing to an additional 3.5% of global solid waste fraction (Patricio Silva et al. 2021). An estimated 4.90 trillion tonnes of plastics are dumped into oceans alone (O'Neill and Lawler 2021).

Microplastics (MPs; <5 mm in diameter) and nanoplastics (NPs; <100 nm in length) are becoming major concerns (Mariano et al. 2021), which can be found in the majority of regularly used products (primary) or can be formed by the fragmentation of bigger plastic litter (secondary) (Kokalj et al. 2021). Primary MPs are mass-produced from manufacturing and domestic applications, which includes exfoliating facial scrubs, toothpaste, detergents, personal care products, abrasive cleaning agents, plastic powder for molding, and synthetic clothing (nylon/polyester) (Wang et al. 2020a; Birch et al. 2020; Sun et al. 2019). Paints, adhesives, electronics, etc., are responsible for NPs release (Kihara et al. 2020). Secondary MP/NPs are formed by fragmentation of macroplastics (200–1000 µM) through shear forces (Auta et al. 2017) which account for 70–80% of all plastic released into the environment, while primary MPs contribute only 15–30% (Mariano et al. 2021). MP/NPs infiltrate terrestrial, freshwater, and marine ecosystems from the equator to the poles and surface water to deep-sea sediments (Wang et al. 2016).

In general, MP/NPs enter the water bodies through domestic waste, sewage treatment plants (STPs), industrial effluents, stormwater, estuaries, and riverine transport, surface runoff, wind currents, and disposal practices (Liu et al. 2021; Sutton et al. 2016). MP/NPs are present as fragments, pellets, fibres, films, granules, and Styrofoam that vary with surface-mass area ratios (Koelmans et al. 2019). The chemical variation found in STPs includes polyethylene, polystyrene, and polypropylene which tend to float while polyethylene terephthalate (PET) and polyvinylchloride (PVC) are more likely to sink due to their density (Wu et al. 2017; Avio et al. 2017; Carr et al. 2016). The presence of MP/NPs in the air, water, and food can cause ecotoxicological issues (Zhang et al. 2020). Influx of MP/NPs causes growth reduction, fecundity, reduced immunity, and malformation in animals and human reproductive systems (Wong et al. 2020; Guzzetti et al. 2018). MP/NPs also act as vectors to adsorb tenacious organic pollutants, trace metals, and harmful additives that are multiple times higher than natural sediments (Jiang et al. 2020). Presence of MP/NPs in WWTPs, impact microbial communities, inhibit sludge hydrolysis and accumulate acids (Zhang and Chen 2020).

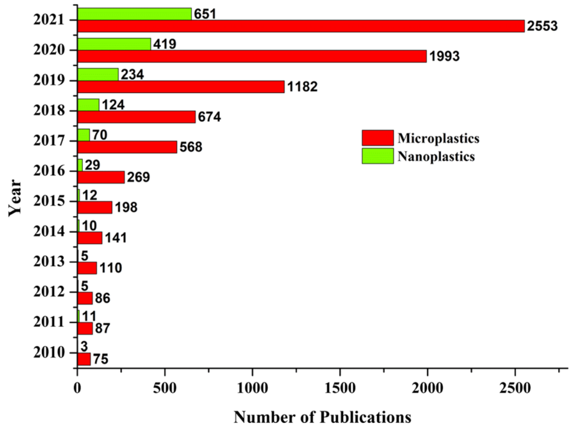

Fig. 1 The cumulative total of annual research papers published on microplastics and nanoplastics. Keyword used = Microplastics and nanoplastics. Data source: www.sciencedirect.com

Plastic occurance, accumulation and persistence as MP/NPs in terrestrial and marine environments is becoming a major global concern. The issue of plastics was recognized by the United Nations (UN) Sustainable Development Goals (SDGs), specifically Goal 14—'Conserve and sustainably use the oceans, seas, and marine resources for sustainable development (Walker 2021). Using the keywords "microplastics" and "nanoplastics" as database search in indexed journals (ISI Science Direct), the publication number (MPs—7136/NPs -1369) is soaring exponentially, which elucidates growing interest in the scientific community (Fig. 1). This review aims to provide current perspectives on MP/NPs concerning (1) Source, occurrence, distribution, and properties (2) Impact on the ecosystem and its services, (3) Techniques in detection and identification of MP/NPs and (4) Remediation and management strategies to mitigate MP/NPs. The current understanding in the literature is highlighted, and future perspectives are indicated.

2 Occurrence and distribution of Micro/nanoplastics

2.1 Terrestrial system

The terrestrial domain is more susceptible to MP/NPs and the annual inputs from land exceeds the total MP/NPs floating in the global ocean (Yee et al. 2021; Wang et al. 2021; Hu et al. 2019). Soil bound MP/NPs originate from multiple sources, which includes sewage sludge/biosolids, domestic waste, irrigation, land fillings, fertilization, plastic film mulch, greenhouse materials, atmospheric deposition, tire abrasion and garden organic waste (Koutnik et al. 2021; Guo et al. 2020; Wang et al. 2020b). MP/NPs penetrate vertically via water infiltration, facilitated by tilling and action of soil microorganisms (O'Connor et al. 2019). The major source of MP/NPs in the soil is from WWTPs as 95% of MPs are retained in biosolids, and treated biosolids, in addition, are used as fertilizer in agriculture (van den Berg et al. 2020; Horton et al. 2017). Furthermore, a single-use facial scrub could release up to 10^6 primary NPs into the domestic sewage system (Shen et al. 2019). Around 306.9 tonnes of microbeads are discharged into the environment, of which 80% are derived from STPs annually (Cheung and Fok 2017). Biosolids alone introduce approximately 44,000 to 430,000 tonnes of MP/NPs into North American and European grasslands each year (Nizzetto et al. 2016). Textiles contribute 35% of the MP/NPs, specifically in the oceans, in the form of synthetic microfibers (Xu et al. 2020). A single garment produces about 1900 microfibres, and around 700,000 microfibres can be released from a single load of laundry, according to the International Union for Conservation of Nature (IUCN) (Boucher and Friot 2017).

Plastics have a lower density than soil minerals, and once they enter the soil, it forms soil aggregates and alter soil physicochemical properties. It effects water-binding capacity, soil density, porosity, soil integrity, soil surface cracking, and organic matter and influences the nutrient cycling process in the soil ecosystem (de Souza et al. 2019; Liu et al. 2017). Polyethylene, polystyrene, and polyvinylchloride nanoplastics (20 nm to 150 nm) indirectly influence soil productivity and directly alter the soil microorganisms and fauna communities (Wahl et al. 2021). Organic fertilizers and compost act as vectors for MP/NPs into the soil (Bläsing and Amelung 2018). Plastic mulch films are used extensively as soil conditioners in agriculture (Gao et al. 2019). Farmlands in China used ~1.47 million tonnes of plastic mulch with film content reaching 502 kg ha^{-1} soil in all provinces and regions (Zhang et al. 2016). Farmlands reported 62.5 particles kg^{-1} in deep soil, 78.0 particles kg^{-1} in shallow soil, and 1.9 ± 0.87 particles kg^{-1} in garden soils (Liu et al. 2018). A study on the rice-fish co-culture ecosystem showed MPs concentration in soil ranges between 10 to 78 particles kg^{-1} (Lv et al. 2019; Liu et al. 2018). MPs congregations in Sydney's industrial zones ranged from 300 to 67,500 mg kg^{-1} (Fuller and Gautam 2016). The dispersal of plastic debris in coastal belts, vegetable farmlands, and riparian forest zones found MPs concentration ranged from 7100 to 42,960 particles kg^{-1} (Zhang and Liu 2018). Due to its large mass-surface ratio and hydrophobic nature when combines with other persistent organic pollutants (POPs), heavy metals, and antibiotics, they serve as vectors for toxic chemicals (Wang et al. 2020b; Hodson et al. 2017). MPs-POP mixtures threaten the terrestrial environment by adsorbing various POPs (Yang et al. 2019a, b). Microplastics absorb pesticides and hinder the chemical degradation of polyethylene film residues, resulting in pesticides accumulation (Ramos et al. 2015). MPs affinity to adsorb antibiotics was also reported (Li et al. 2018).

Soil act as a preliminary sink and most MP/NPs are deposited in landfills. Between 1950 and 2015, global trash production was expected to total 6300 million tonnes, with landfills and other environmental compartments accounting for 79% of the total (Patricio Silva et al. 2021; Ng et al. 2018). A study in Finland enumerated MP/NPs particles up to 4.5 items L^{-1} with size >1 mm in diameter (Kilponen 2016). Plastic trash blowing from landfills, if not adequately buried, can affect the soil and reach aquatic habitats (Peng et al. 2017). The particle and mass concentrations of MP/NPs in the untreated leachate were 235.4 ± 17.1 item/L and 11.4 ± 0.8 g/L, respectively, with particles size less than 50 µm (Sun et al. 2021). Fresh Kills landfills leached most of the medical waste on New York beaches in 1987–1988 (Hale et al. 2020). Natural catastrophes (hurricanes, tsunamis, and wildfires), construction and demolition landfills frequently aid in the deposition of plastic debris (Brand et al. 2018). PE and PP were the predominant polymer types in municipal solid waste (MSW) landfills with sizes ranging from 100 to 1000 µm (He et al. 2018).

2.2 Air

Micro/nanoplastics are released into the air due to wastewater treatment plant operation and (Chang et al. 2020) mechanical activities such as severe wind events, sea surface spray and wave breaking which often entrain them into the atmosphere (Brahney et al. 2021). Sea-spray releases MP/NPs from the maritime environment into the atmosphere, resulting in a global extrapolated value of 136,000 tonnes blowing on shore per year (Allen et al. 2020). Fibre, film, foam, and fragment morphologies have been dominant in atmospheric deposition and air masses (Zhang et al. 2020). MPs have a higher density than air (1.225 g/L at 15 °C at sea level), and their density diminishes as altitude rises (Revel et al. 2018). The commonly distributed MP/NPs in environment are 0.91–0.93 g/cm^3 of low-density polyethylene, 0.94 g/cm^3 of high-density polyethylene, 0.85–0.83 g/cm^3 of polystyrene, 1.02–1.05 g/cm^3 of polyamide, 1.37 g/cm^3 of polyester, 1.38 g/cm^3 of PVC, 1.2 g/cm^3 of polycarbonate and 1.42 g/cm^3 of polyformaldehyde (Rezaei et al. 2019). MP/NPs suspended in atmospheric air correspond to synthetic fibres and, upon inhalation, cause respiratory disorders (Wright et al. 2020). A significant amount of MPs are ejected from tyres (100,000 metric tonnes) and brakes (40,000 tonnes) each year which get transported through the air (Evangeliou et al. 2020). Car tyres and brake systems are responsible for 5–10% of worldwide ocean plastic pollution and 3–7% of PM2.5 in the air (Evangeliou et al. 2020). According to a study, each person in the United States generates 4.7 kg of tire wear microplastics per year^{-1}, equating to 1.8 million metric tonnes per year^{-1}(Hale et al. 2020). Road dust has been recognized as an important source of MP/NPs in urban/sub-urban areas. The atmospheric deposition of MP/NPs is higher in indoor environments, and about 33% of microfibres contain petrochemicals (Wu et al. 2019). On normal exposure, > 900 MP/NPs particles are ingested by a child per year (Abbasi et al. 2019).

2.3 Aquatic system

In the early 1970s, the first report on MPs occurance in marine system was published (Carpenter and Smith 1972). MPs enter into fresh and marine environments mainly through natural erosion and human activities (industrial and domestic wastewaters, aquaculture, and tourism) (Guo et al. 2020; Birch et al. 2020). "River to the ocean" is one of the major pathways for plastic reservoirs (Wu et al. 2019). Eight rivers in Asia (Yangtze, Yellow, Indus, Hai He, Ganges, Mekong, Amur, and Pearl), and two rivers in Africa (the Nile and the Niger) are the sources of 90% of the plastics that reach the sea every year (Sadeghi et al. 2021). Every year river-fed estuaries dump 1.15 to 2.41 million tonnes of plastic trash into the sea (Lebreton et al. 2018). The North-western Pacific Ocean and Arabian Bay are widely polluted by MP/NPs with concentrations ranging from 640 to 42,000 items km^{-2} and 4.38 × 10^4 items per km^{-2} (Xu et al. 2019). An estimated 15–51 trillion MP/NPs particles (93–236 thousand metric tons) were found in oceans (Naik et al. 2019). Plastic pollution, dumping, and trash eventually goes downstream into estuaries and coastal seas. Changing rainfall, wind speed, waves, and oceanic currents contribute to the transfer of pollutants into seawater (Cózar et al. 2014). Agricultural runoff, soil erosion, and atmospheric deposits also transfer MP/NPs in marine ecosystems (Hale et al. 2020). Field studies conducted on the banks of Ciwalengke River, Indonesia, showed the presence of MP/NPs as primary fibres with concentrations ranging from 5.85 particles per litre in surface water and 3.03 particles per 100 g of sediment in sizes ranging from 50 to 2000 µm (Alam et al. 2019). The abundance of MPs ranged from 112 to 234 particles kg^{-1} dry weight in lake Bolsena shoreline sediments of central Italy (Lake Chiusi) (Fischer et al. 2016). Microalgae also can colonize MP/NPs and increase their specific density through biofouling and hetero-aggregation (Mateos-Cárdenas et al. 2021). In 2011, Japan's Tsunami injected 5 million tonnes of plastic debris into the Pacific, which is approximately the amount of plastic waste entering the ocean each year (Murray et al. 2018).

2.4 Wastewater systems

Anaerobic digestion is a prevalent approach for handling sewage sludge (in WWTPs), wherein particulate organic waste gets converted to soluble substrates is the first step. Polyethylene, polyamide, polyvinyl chloride, and polystyrene in activated sludge impede anaerobic digestion and have varying effects depending on their physicochemical nature (Zhang and Chen 2020). Additives or toxic chemicals released from MP/NPs rupture microbial cells or inhibit key enzymes related to acidogenesis (hydrolysis and fermentation) and methanogenesis and hinder anaerobic digestion (Azizi et al. 2021). MP/NPs also penetrate the exopolysaccharide matrix, alter protein secondary structures, and damage phospholipids (Azizi et al. 2021). Following MP/NPs exposure, changes in the activity of key enzymes such as protease, cellulase, glucosidase, acetate kinase, butyrate kinase, coenzyme F420, etc. were observed to cause changes in the digester performance (Azizi et al. 2021). The effects of various doses of polyvinyl chloride and polystyrene (10–200 particles g^{-1}) on waste-activated sludge's anaerobic digestion negatively affected hydrolysis rate and biogas generation (Li et al. 2020). After exposing digesters to polystyrene nanoparticles, marked variations in pH, volatile fatty acids, and ammonia nitrogen profiles were reported (Zhang et al. 2020). Microbial communities also resulted in a considerable decrease in the digester upon short and long-term exposure to MP/NPs (Azizi et al. 2021).

3 Micro/nanoplastics degradation

Plastics can disseminate micro-and nanoplastics either via biodegradation or non-biodegradation processes (Yee et al. 2021). Plastics are long-chain organic compounds, and their degradation is determined by the change in physical properties (shape, size, porosity, surface area, and crystallinity) and their ability to interact with other pollutants (Cai et al. 2017; Campanale et al. 2020). Once disposed of, plastic waste undergoes chemical, biological, and environmental changes, breaking large pieces of plastics into micro and nanoplastics (Allen et al. 2020). Thermal degradation, physical deterioration, photodegradation, thermo-oxidative degradation, biodegradation, sand friction, and hydrolysis are some of the weathering mechanisms that degrade larger plastic polymers to MPs (Hale et al. 2020; Wu et al. 2019) (Fig. 2). NPs are formed when MPs are exposed to light (visible and UV rays) at moderate temperatures (30 °C) (Jiang et al. 2020). Hydrolysis and photodegradation are natural chemical reactions that use water molecules and UV–visible light to break down chemical bonds in polymers and turn them into monomeric forms (Yee et al. 2021). The fragmentation of polystyrene down to the nanoscale occurred in four weeks inside a weathering chamber (Kihara et al. 2020). Throughout the backbone of plastic polymers, heteroatoms (O, N, and S) serve as sites for hydrolytic/enzymatic processes and facilitate breakdown into low molecular weight

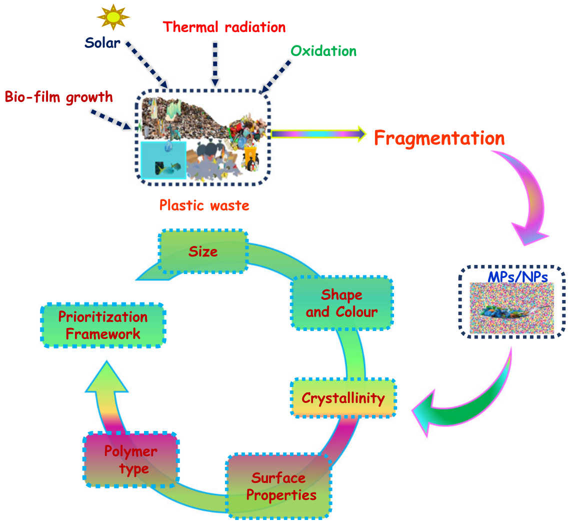

Fig. 2 Micro and nanoplastics fragmentation in terrestrial and aquatic systems and associated physcio-chemical properties

fragments that microbial cells can assimilate (Ng et al. 2018). The chemical structure of polymers changes at the molecular level, such as chain scission, cross-linking, and the inclusion of oxygen-containing functional groups (esters, ketones, and alcohols) into the surface of plastic particles lowering their hydrophobicity (Yee et al. 2021). Non-biodegradation methods break down polymeric structures, changing mechanical characteristics and increasing surface zone, ensuing increased physico-chemical reactions and interfaces with microbes (Lucas et al. 2008). Microorganisms such as bacteria, fungi, and other eukaryotic colonizers secrete extracellular enzymes and mediate biodegradation (Naik et al. 2019; Enfrin et al. 2019). Biofilm formation also promotes the formation of hetero-aggregates where MP/NPs become sticky because of extracellular polymeric substance matrix (Wu et al. 2019). High saline content in marine environments, combined with naturally present microbes, promotes plastic disintegration faster than in terrestrial ecosystems (Ng et al. 2018). The structural integrity of the plastics is further weakened by subsequent interactions with wind, waves, solar UV radiation, temperature, additives, environmental conditions, and other abrasive interactions released through delamination (Jiang et al. 2020).

4 Properties of micro/nanoplastics

Microplastics' physical properties are generally associated with particle size, shape, and color, whereas chemical properties associate with crystallinity, surface properties, toxic metals, and additives adhered on MP/NPs surfaces (Hildebrandt et al. Lambert et al. 2017).

4.1 Size

Micro/nanoplastics sizes are often determined by the sampling and analysis methods. MPs come in various sizes, but the most common are 1 µm to 5 mm, whereas NPs are <100 nm in length (Murray and Örmeci 2020; Hartmann et al. 2019). The National Oceanic and Atmospheric Administration proposed these dimensions. The Joint Group of Experts later adopted the Scientific Aspects of Marine Environmental Protection (GESAMP 2016) and the United Nations Environmental Program (UNEP 2014). Currently, the two most common MP/NPs classification methods are sieve retention and microscopic imaging. The dimensions 25 mm, 100 mm, and 500 mm were the most frequently identified sizes in WWTP influent and effluents (Mintenig et al. 2017). Samples from the Atlantic Ocean revealed MPs in size 20 to 40 µm, accounting for 64% of the total detected particles (Simon et al. 2018; Enders et al. 2015), where atmospheric depositions accounted for microfibers of ~5000 µm in size (Cai et al. 2017). The size and ubiquitous distribution of MP/NPs in the aquatic and marine environment facilitate their availability to pelagic and benthic habitats (Campanale et al. 2020). Several species of zooplankton (*Calanusfinmarchius, Euphausia pacifica,* and *Neocalanus cristatus*), bivalves, macro-sized invertebrates, and fish ingest a wide range of MP/NPs ranging from 0.5 nm to 816 µM (Botterell et al. 2019; Lehtiniemi et al. 2018).

4.2 Shape

The shape is one of the important indicators used for microplastics classification. MP/NPs come in a wide range of shapes (fragment, foam, paint, pellet, foil, sphere, fibre, film, line, bead, flake, sheet, granule, and nurdle) (Campanale et al. 2020; Zhang et al. 2020; Koelmans et al. 2019). The primary form of microplastics, degradation, erosion process and the duration time in the environment all influence the shape of MP/NPs (Murray and Örmeci 2020). Fibres and fragments are the dominant shapes and account for 52.7% of wastewater and urban atmospheric depositions (Annenkov et al. 2021). Fibre particles form due to mechanical abrasion and chemical weathering, whereas fragmented microplastics may form due to greater plastic items exposed to fatigue or UV light (Zhou et al. 2020). The shape of MP/NPs influences pollutant transport in the environment. Films are thin and have a bigger surface area than fragments of the same mass for atmospheric conveyance (Allen et al. 2020). Microfibres and fragments are dominant shapes found in international waters (Wang et al. 2020a).

4.3 Colour

Colour is one of the most important parameters for identifying plastic debris and potential contaminations in sample preparation. Various MP/NPs have been documented including red, orange, yellow, brown, tan, off-white, white, grey, blue, and green (Murray and Örmeci 2020; Rochman et al. 2019). Particles with dark, transparent, white, or translucent may be underrepresented (Hartmann et al. 2019). Blue and red fibres are the most commonly reported colours (Bergmann et al. 2019). Polypropylene has been clear and transparent, while polyethylene and polyethylene with low-density have been assigned opaque colors (LDPE) (Zhang et al. 2020). Microplastic discoloration can occur during weathering and sample preparation (H_2O_2– oxidative digestion), which should be considered when reporting and interpreting data (Allen et al. 2020). Marine organisms commonly ingest bright color NPs particles similar to natural foods. Zooplankton, euphausiid, copepods, and fish larvae ingest MPs predominantly in blue, red, green, and black (Botterell et al. 2019).

4.4 Chemical additives

MPs and NPs are mostly additives and basic polymeric materials generated from plastics and chemicals absorbed from the environment (Table 1). Plastic toxicity is associated with built-in chemicals (i.e., monomers, solvents, catalysts, additives-dyes, plasticizers) (McGivney et al. 2020). Low molecular weight additives entrenched into the polymer matrix are weak and leaches readily into surrounding water bodies (flame retardants from electronic items; PET oligomers from bottles and food trays; Pb from unplasticized PVC pipes; nonylphenol from food contact materials and Sb from PET water bottles) (Schwinghammer et al. 2020; Dobslaw et al. 2020). During plastic production, additives are added for desired colour, transparency and to improve the working liability of the product, which makes the polymer more resistant to degradation from physical (temperature, ozone, and light radiation), mechanical, electrical, and biological (fungi, bacteria) agents (Hale et al. 2020). Inert or reinforcing fillers, dyes, UV stabilizers, plasticizers, lubricants, and flame retardants are additives. In addition, wood, graphite, glass fibres, rock flour, kaolin, cotton flakes, jute, clay or linen, cellulose pulp are also used as additives (Campanale et al. 2020; Dobslaw et al. 2020). These additives upgrade the plastic properties, but most are toxic and potential soil, water, and air contaminants. Ingestion or inhalation of these chemicals disrupts the endocrine system and causes hormonal imbalance, asthma, reproductive problems, metabolic disorders, and neuro-developmental conditions (Schwinghammer et al. 2020). Bisphenol A, phthalates, heavy metals, and brominated flame retardants are most common additives (Li et al. 2021).

4.5 Crystallinity

Crystallinity is ordered structural linkages that influence the plastics' density, permeability, and swelling behavior (Botterell et al. 2019; Lambert et al. 2017). The crystallinity of MP/NPs changes with residence time in the environment. The breakdown of the amorphous region in the polymer promotes overall crystallinity and reduces the size of MP/NPs (Campanale et al. 2020). The materialization of crystallites might differ in toxicity and will alter MP/NPs from their counterparts. It eventually influences particle shape, size, surface area, and chemical properties like adsorption of pollutants, additives and consecutively affects ingestion rate (Botterell et al. 2019; Gao et al. 2019).

4.6 Surface properties

The surface property primarily involves surface area and surface chemistry. With the decrease in particle size, MP/NPs surface area increases, and particles at

Table 1 Various Chemical additives, polymers and metals with plastics and their effect on the human health

Additives		Effect on human health	References
Plasticizers	1,2-benzenedicarboxylic acid	Neuronal toxicity	Campanale et al. (2020) Engwa et al. (2019) Verla et al. (2019) Jeong and Choi (2020)
	Chlorinated paraffins, Dicyclohexyl phthalate (DCHP)	Breast cancer	
	di-C7-11-branched and linear alkyl esters (DHNUP)	Cardiovascular	
	Butyl benzyl phthalate (BBP)	Kidney diseases	
	Diethyl phthalate (DEP)	Metabolic and mental disorders	
	Diheptyladipate (DHA)		
	Formaldehyde, 4,4′-methylenedianiline (MDA)	Neuro-degenerative disorder	
	Dipentyl phthalate (DPP)		
	Diisobutyl phthalate (DiBP)		
	Diisoheptylphthalate (DIHP)		
	Heavy metals—Zinc, Cadmium, Tin, Lead, Titanium, Barium		
Biocides	Arsenic trioxide	Metal–estrogen	
	Triclosan	Mutagen	
	Triphenyltin hydroxide	Carcinogen	
	Butyltin trichloride	Brain damage	
	Dimethyltin dichloride	Congenital disabilities	
	Dibutyltin dichloride	Lung, skin, liver, bladder, kidneys, Gastrointestinal damage	
	Tetrabutyltin		
	Tributyltin chloride		
	Heavy metals- Antimony, Copper, Mercury, Arsenic, Tin		
Flame retardants	2,2′,4,4′-Tetrabromodiphenyl ether (BDE 47)	Carcinogen	
	3.3′-5.5′-Tetrabromobisphenol (TBBPA)	Apoptosis	
	1,2,3,4,5,6-Hexabromocyclohexane	Genotoxicity	
	Tris(2-chlorisopropyl)phosphate (TCPP)	Osteomalacia and bone fractures	
	Boric acid		
	Decabromodiphenylethane (DBDPE)	DNA methylation	
	2,2′,4,4′,5,5′-Hexabromobiphenyl ether (BDE 153)		
	Heavy metals—Antimony, Bromine, Aluminum, Zinc, Bromine		
Stabilizers, Antioxidants and Organic pigments	Bisphenol A	Metabolism changes	
	Fatty acid amides	DNA methylation	
	2-tert-Butyl-4-methoxyphenol	Anemia	
	Triglycidylisocyanurate (TGIC)	Neurological disorder	
	2-t-butyl-4 hydroxyanisole (BHA)	Cardiovascular andendocrine deficits	
	Tris(2,4-di-tert-butylphenyl) phosphate		
	Butylated hydroxytoluene (BHT)	Hypertension	
	Tris-nonyl-phenyl phosphate (TNPP)	Miscarriages	
	4-Nonylphenol	Disruption of nervous	
	Irganox 1010	Brain damage	
	4-Octylphenol	Infertility	
	Heavy metals—Aluminum, Manganese, Barium, Cobolt, chromium, Lead, Titanium, Tin, Cadmium, Aluminum		

the nanoscale have a marked impact (Gao et al. 2019). Although the surface area is not widely recognized, microbeads can be calculated using the spherical equivalent diameter (Botterell et al. 2019). Nanoscale particulates are assumed using geometrical estimates, resulting in a sevenfold interpolation of the surface area (Campanale et al. 2020). The plastic surface is affected by photo- and oxidative degradation processes that generate new functional groups when reacted with OH radicals, O and N oxides, and other photo-generated radicals (Schwinghammer et al. 2020). These pathways disrupt the plastic surface area, and microscopic particles are released upon ingestion, chemical leaching, and formation of angular-shaped NPs from primary microbeads (Lambert et al. 2017). Furthermore, surface chemistry significantly impacts particle-biota interactions as the microbial community utilizes the oxygenated binding sites (McGivney et al. 2020; Campanale et al. 2020).

5 Impact on the environment

5.1 Soil

Micro/nanoplastics are a serious threat to soil biota because they inhibit plant growth, organism reproduction, and soil biodiversity (Hale et al. 2020). Soil serves as a habitat for terrestrial animals, and the ecotoxicological effects of MP/NPs on soil fauna are exuberantly increasing (Fig. 3). MP/NPs are ingested by soil invertebrates and poultry, providing a potential gateway to enter humans and animals due to their small size (Cox et al. 2019). Species like nematodes, snails, mice, isopod, collembolan, and chicken are studied with different MP/NPs (HDPE, PVC, PE, PS, PA, and PET), of which earthworm studies are predominant (Ng et al. 2018). The toxicity of MP/NPs on soil biota includes growth suppression, energy metabolism disturbance, immunological responses, locomotion reduction, gastrointestinal disorders, gut microbiota dysbiosis, metabolic abnormality, and mortality (Wang et al. 2020b). In vertebrates, leaching from MP/NPs additives such as bisphenol A and phthalates disrupt the endocrine system through the estrogenic effect (Zhang et al. 2020). Polystyrene MPs showed damage to cholinergic, GABAergic neurons, and oxidative stress kinase (Lei et al. 2018). *Lumbricus terrestris* (earthworm), when exposed to 28% PE-MP in dry plant litter, experienced growth reduction and eventually cell death (Huerta Lwanga et al. 2016). A cross-section study reported the existence of MPs debris in the gastrointestinal tract and gut system of domestic live-stock and terrestrial birds (Omidi et al. 2012). Sunlight, oxygen availability, high temperature, soil microbes, and terrestrial biota in the top soil layer function as plastic garbage degradative habitat (Wang et al. 2020a). Antibiotic resistance genes are transported, deposited, and dissipated in soil by MP/NPs (Sun et al. 2019). Microbial enzymes (dehydrogenase and fluorescein diacetate hydrolysis) tend to increase plastic mulch residues (Wang et al. 2016). MP/NPs can affect gene expression and alter the microbial cellular structure. They penetrate cells through endocytosis (Zhang and Chen 2020). MP/NPs with exopolysaccharides (EPS) in wastewater systems are mediated by functional groups such as carbonyl, amide groups, and side chains of lipids or aminoacids (Feng et al. 2018). PS-NPs showed a change in the secondary structure of proteins in EPS (Catarino et al. 2021).

5.1.1 Plant and macrophytes

Plants metabolize a wide range of pollutants (MPs, NPs, and POPs) either by avoiding its uptake or detoxifying by various enzymatic activities in the plant cell (Dobslaw et al. 2020; Ng et al. 2018) (Fig. 3). In the plant root zone, soil microorganisms and animals participate in MP/NPs degradation and protection from toxic effects (de Souza et al. 2019). MP/NPs alter soil fertility and significantly affect growth, biomass, root traits, nutrient uptake, and soil microbial activities (Bianco and Passananti 2020; Zhang et al. 2020). MP/NPs adsorption and internalization have been observed in plants used for human consumption (*Triticum aestivum, Allium cepa*, *Allium fistulosum*, *Lactuca sativa*, *and Zea mays*) and macrophytes (*Lemna minor* and *Fucus vesiculosus*) (Mateos-Cárdenas et al. 2021). Polystyrene (PS), polyethylene (PE), polypropylene (PP), cotton, and rayon fibres were commonly found adhering to plant surfaces (Mateos-Cárdenas et al. 2021). *Triticum aestivum* and *Lactuca sativa* internalize 200 nm PS nanobeads and 2.0 µm polymethylmethacrylate microbeads, respectively (Li et al. 2020). PS nanobeads (10 nm to 200 nm) was internalized in the root maturation zone and translocated to leaf vessels via

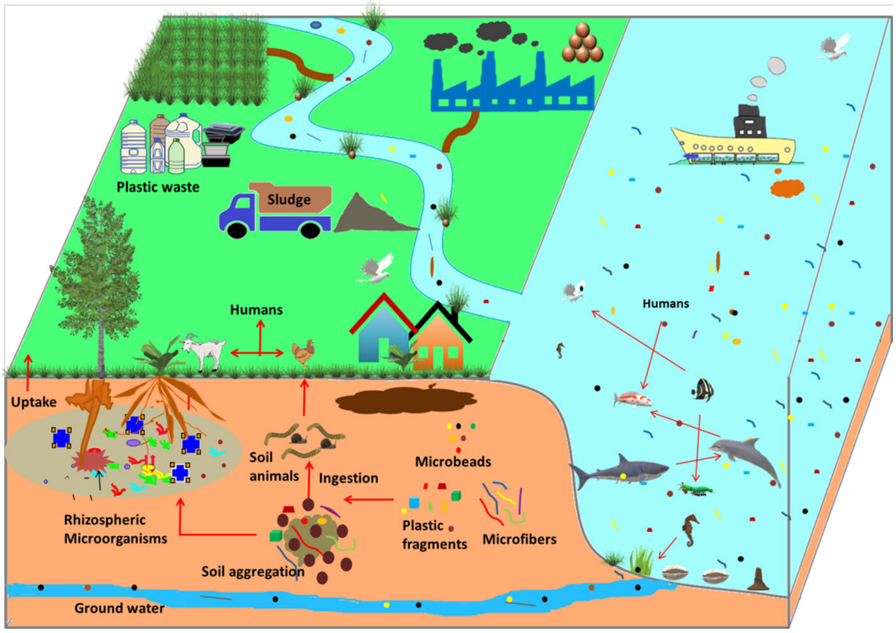

Fig. 3 Micro/nanoplastics distribution across different food trophic levels and their interaction with biological processes in terrestrial, fresh and marine systems

vascular tissues in *Zea mays* L. (Sun et al. 2021), *C. pteridoides* (Yuan et al. 2019), and *Vigna radiata* (Chae and An 2017). Apple, broccoli, and carrots showed significant quantities of MPs per gram (Conti et al. 2020). In a study, co-culturing *Lolium perenne* (ryegrass) with *Aporrectodea rosea* (rosy-tipped earthworm) with different MPs (PE, HDPE, and fibres) showed reduced plant biomass, pH, and inhibited growth (Boots et al. 2019). Various plant species and soil types have different reactions. Plants and macrophytes act as potential vectors for trophic transfer of MP/NPs into natural food webs. Duckweed is toxic to amphipods (Mateos-Cárdenas et al. 2021), *F. vesiculosus* is toxic to *Littorina littorea* (Gutow et al. 2019), and *V. radiate* is toxic to *Achatina fulica* (Mateos-Cárdenas et al. 2021. High doses of MPs are used on a laboratory scale to identify and impact soil biota easily. However, MPs levels in human-impacted soil are unlikely to be greater than 0.1% dry weight (Sun et al. 2021).

5.1.2 Mangroves

Mangrove wetland is an intertidal ecosystem that acts as a barrier for retaining land-based pollutants, including MP/NPs (Helcoski et al. 2020; Li et al. 2020). Nor and Obbard (2014) first reported MPs in mangroves with concentrations ranging from 12 to 62.7 items per kg. Cienaga Grande de Santa Marta (CGSM), one of the coastal lagoons declared as RAMSAR wetland, consists of MP/NPs concentrations oscillated between 31 and 2863 particles per kg of dry sediment (Garcés-Ordóñez et al. 2019). The plastic debris is always accumulated on the leaf's surface, trunk, pneumatophores and transported by ocean currents and storm runoff from the land (Zhou et al. 2020). Fibres, foams, and films are dominant MP/NPs detected in mangrove sediments worldwide (Duan et al. 2020). MP/NPs inhibit the growth of mangrove seedlings, loss of foliage, and negatively affect mollusks, crabs of the Uca genus, and crustacean habitats that live in soils associated with mangroves (do Sul et al. 2014). Four species of juvenile fish (viz. *Ambassis dussumieri*, *Terapon jarbua*, *Mugil sp., and Oreochromis mossambicus*) growing in KwaZulu-Natal mangroves unveiled fibres (68%) and fragments (21%) with 0.79 ± 1 particle per fish (Naidoo et al. 2020). Similar findings were found in each fish's gills, stomach, and intestine in the Zhanjiang mangrove wetlands, where 2.83 ± 1.84 items per individual MPs were reported (Huang et al. 2020). *Gerreidae* sps. (*Eugerres brasilianus,* *Eucinostomus melanopterus, and Diapterus rhombeus*) found in estuaries and mangroves ingested MPs and showed a reduction in gut weight and false satiation (Ramos et al. 2012).

5.2 Aquatic biota

Plastic pollution is an upsurging global concern in aquatic systems, spreading even to the most remote habitats (Fig. 3). Plastic debris ranging from microscopic to high density is prevailing in benthic and pelagic habitats of rivers, seas, and remote regions of Artic to Antarctica (Bianco and Passananti 2020). MP/NPs have been identified as a major concern for biodiversity loss, climate change, and threatening living biota.

5.2.1 Fresh water systems

In recent years, freshwater systems such as ponds, lakes, estuaries, and rivers have attracted as much attention as oceans in terms of MP/NPs flux (Wong et al. 2020) through domestic waste, industrial effluents, and sewage disposal sites (Meng et al. 2020; Hu et al. 2019). Particle qualities (size, shape, and density) and physical forces (flow velocity, water depth, sediment topography, tidal cycles, and urban runoff), and anthropogenic activity (e.g., dam release) influence MP/NPs movement in freshwater systems (Helcoski et al. 2020; Ziajahromi et al. 2017). MP/NPs have been found in freshwater systems across the continents (Lahens et al. 2018). The Danube delivers 4.2 tonnes of plastic each day to the Black Sea (Lechner et al. 2014). Microbeads, raw plastic (pellets and flakes), films, fibres, and fragments are types of MP/NPs found across the freshwater streams, where microbeads and fibres are dominant (Li et al. 2021). Plastic debris accumulation in aquatic habitats alters light penetration and affects biogeochemical cycles in the water column (Chen et al. 2020). NPs absorbed on the surface of microalgae cause a shading effect and decrease the fluidity of cell membranes (Zhu et al. 2021). Internalized NPs reduce carbohydrate metabolism and decrease cellular esterase activity, electron transport rate, and lipid reserves, thus modulating algal energy metabolism (Zhu et al. 2021). Large plastic particles, such as fishing ropes and nets, entangle turtles, birds, and mammals, whereas small particles are ingested directly, causing a gut obstruction (Horton et al. 2017). MP/NPs in freshwater systems severely impact the ecosystem as they are the major reliable sources of food and drinking water for humans (Senathirajah et al. 2021). NPs last longer in the body than MPs and are transported to the digestive gland. Trophic transfer of NPs was investigated, and increased uptake causes tissue inflammation, reduced lipid stores, nutrient uptake, and membrane instability in digestive cells (O'Neill and Lawler 2021). *Daphnia magna* and *Danio rerio* (Zebra fish) are widely used biological and toxicological research models to assess MP/NPs in freshwater systems (Jiang et al. 2020). When exposed to 0.1, 1, and 10 ppm of polystyrene NPs, Zebrafish embryos, accumulation in embryonic tissues and choroid membranes was observed, thereby influencing the inter or trans generational toxicity (Pitt et al. 2018). Freshwater fauna (*Daphnia magna*, *Gammarus pulex*, and *Lumbriculus variegutus*) ingested MPs and translocated them from cells to oil storage droplets and faces (Imhof et al. 2013). A field study in French water streams showed 7 out of 11 gobies (*Gobio gobio*) contained MPs in their tissues (Sanchez et al. 2014). Polystyrene NPs of sizes 20 nm to 39.4 nm affect the biomass, photosynthetic efficiency, and mortality rate of *Chlorella* sp., *Daphnia magna*, *Raphidocelis subcapitata*, and *Scenedesmus obliquus* (Chae and An 2017). Organisms predate freshwater species at higher trophic levels, and MP/NPs consumed have cascade impacts on the ecosystem's health(Nizzetto et al. 2016; Li et al. 2015).

5.2.2 Marine systems

Marine litter is a huge environmental and economic issue around the world. Plastic garbage reaches the ocean at a pace of 4.8 to 12.7 million tonnes per year, with 80% of it coming from land-based sources worldwide (Mofijur et al. 2021; Raju et al. 2018). Rivers, lakes, sewage effluents, and anthropogenic activities are major gateways to transport plastic debris to oceans (Xu et al. 2019). The European Union's Marine Strategy Framework Directive (MSFD), the OSPAR commission, the Stockholm Convention, and International Pellet Watch have all focused on marine litter problems to protect and conserve its resources (Stockholm convention, UNEP 2018; OSPAR 2014). Aerial surveys in the Pacific Ocean discovered 79 thousand tonnes of plastic floating over a 1.6 million km^2 area, dubbed the "Great Plastic Garbage Patch." (Lebreton et al. 2018). The density of sea water is 1.02–1.07 g/cm^3, while that of plastic is 0.8–1.5 g/cm^3, which tends MPs to float on water surfaces (Wang et al. 2020a). PS, PP, PE, PU, PVC, and PET are among the most common plastics contributing to marine pollution (Annenkov et al. 2021; Birch et al. 2020). Nine sediment samples at 2340–5570 m depth analyzed in the Arctic region showed widespread NPs (42–6595 MPs kg^{-1}) (Bergmann et al. 2019). MPs will outnumber fish in oceans by 2050, according to World Economic Forum 2016.

Once plastic debris enters the marine environment, they interact with various marine species across trophic levels (Wang et al. 2020a; Guzzetti et al. 2018). MP/NPs have been found in a wide variety of marine species, including cetaceans, copepods, chaetognaths, shrimps, echinoderms, zooplankton, corals, fishes, turtles, seabirds, and mammals, causing cascading effects in the marine food web due to their small size (Botterell et al. 2019). Phagocytosis and pinocytosis are two pathways for marine organisms' uptake of MP/NPs (Allen et al. 2020). They are transported through several physiological routes and translocate mainly to the stomach, intestine, gills, digestive tract, and sometimes to the liver (Allen et al. 2020). Ingestion of MPs causes deleterious effects in marine taxa, mainly impairment, reduced feeding behavior, inhibited growth, nutrient uptake, decreased immune response, oxidative stress, cellular toxicity, pseudo-satiety sensation, fertility reduction, malnutrition, and lastly, death (Gonçalves and Bebianno 2021). *Tigriopus japonicas,* when exposed to polystyrene microbeads, exhibited a decrease in fecundity across two generations (Lee et al. 2013). *Brachionuskoreanus* ingestion with fluorescent-labeled PS nanobeads (50 nm, 500 nm, and 6 µm) causes increased oxidative stress and decreased growth rate, fecundity, and body size (Jeong et al. 2016). Adult oysters and blue mussels on ingestion with 50 nm NPs lead to a substantial decrease in embryo-larvae development and fertilization, which results in complete stagnation (Rist et al. 2019). Similarly, MPs act as vectors for *Aeromonas salmonicidia* (a fish bacterial pathogen), which can form biofilms and transfer to the food chain upon consumption of infected fish (Viršek et al. 2017). To investigate the anthropogenic impacts, 41 fishes were sampled from Giglio Island after refloating project of Coasta Concordia Wreck. Out of 41, 85% of examined fish, i.e., benthonic species *Phycis phycis, Scorpaena* sp., and *Uranoscopus scaber* showed 77%, 84%, and 86% plastic particles. In comparison, benthopelagic *Spondyliosoma cantharus* exhibited 100% plastic particles in the gastrointestinal tract dominated by fragments, lines, and films (Avio et al. 2017). Microorganisms and fisheries habiting coral reefs are eight times more likely to be affected by plastics. MP/NP loads correlate to terrestrial dispersal into the season coral reefs. The amount of plastic objects entangled on coral reefs in the Asia–Pacific region is predicted to reach 11.1 billion, with the number expected to rise by 40% by 2025 (Lamb et al. 2018). Due to the high surface-volume ratio, hydrophobicity, and lipophilicity, MP/NPs can incorporate heavy metals (Ni, Pb, Zn, Cu, Cd, and Ti), PCBs, PAHs, POPs, phthalates, and bisphenol A on their surface (Prata 2018). Co-contamination of mercury and microplastic in marine environments causes behavioral changes and reduced swimming velocity in *Dicentrarchus labrax* (Barboza et al. 2018). *Oryzias latipes* (Japanese medaka fish), when exposed to polyethylene and chemical pollutants (PCBs, PAHs, and PBDEs), showed symptoms of tumor formation, cellular necrosis, down regulation of choriogenin (chgH) in males, and vitellogenin (VTgI) gene expression in females (Guzzetti et al. 2018). Perfluorooctane sulfonic acid and benzo[a]pyren) interact with polyethylene MPs, and are known to induce cancer in humans (O'Donovan et al. 2020). MP/NPs also function as medication delivery vectors entering marine water as medical waste or human faces (Campanale et al. 2020).

5.3 Food web

Micro/nanoplastics enter the food chain because of their wide availability in aquatic and terrestrial environments. MP/NPs were observed in honey, beer, poultry, salt, sugar, teabags, milk, salmon, seaweed, shrimps, and bivalves among other things (Markic et al. 2020; Cox et al. 2019; Santillo et al. 2017). According to reports, the average MP/NPs in food products is as follows: seafood (1.48 items g^{-1}); sugar (0.44 items g^{-1}); honey (0.10 items g^{-1}); salt (0.11 items g^{-1}); alcohol (32.27 items L^{-1}); bottled water (94.37 items L^{-1}); and tap water (4.23 items L^{-1}) (Yee et al. 2021; Zhang et al. 2020). Plastic teabags, bottled water, and seafood are among the sources of exposure (Kosuth et al. 2018), and unusually high MP/NPs in fruits and vegetables have also been observed (Conti et al. 2020).On average, human beings consume 39,000 to 52,000 MP/NPs particles per year, whereas individuals who drink bottled water, in addition, consume 90,000 particles (Mason et al. 2018). MP/NPs contamination in drinking water bottles resulted in 40.1 mg kg^{-1}body-weight day^{-1} for adults and 87.8 mg kg^{-1} body-weight day^{-1} for children (Zuccarello et al. 2019). MPs occurrence was also reported in salt between 56 and 103 particles kg^{-1} of salt (Seth and Shriwastav 2018).

6 Health risk

Plastic usage in daily human activities (drinking water bottles, soft drinks, food packaging, and medical device, etc.) increases their abundance and human health risks (Jiang et al. 2020). The major routes for MPs intake are inhalation, skin perfusion, and ingestion, whereas NPs enter organisms through physical piercing and endocytosis/phagocytosis (O'Neill and Lawler 2021). The most prevalent route of MP/NPs exposure is through contaminated food, associated with gastrointestinal consequences, including reduced epithelial permeability, localized inflammatory processes, and changes in gut microbiota composition (Campanale et al. 2020). Most European countries rely on shellfish and crustaceans as their food diet and are estimated to ingest 1800 MPs/year per person (Barboza et al. 2018). MPs were identified in the tissues of commercially cultivated bivalves, *M. edulis*, and *Crassostrea gigas*, with quantities ranging from 360.07 to 470.16 particles per gram, collected from mussel farms and supermarkets (Zhang et al. 2020; van Cauwenberghe and Janssen 2014). Rapid bronchospasm, diffuse interstitial fibrosis, inflammatory and fibrotic changes in bronchial and peribronchial tissue, and interalveolar lesions are caused by inhaling MP/NPs (Mariano et al. 2021). The accumulation of NPs by primary producers *Chlamydomonas reinhardtii* and *Daphnia magna* starts the trophic transmission of NPs in the food web (Zhu et al. 2021). The trophic transfer occurs when primary producers are eaten by secondary (*Oryza sinensis*) and tertiary (*Zacco temminckii*) consumers before being passed to humans (Zhu et al. 2021). NPs transferred from the primary producers to top consumers make morphological alterations and strongly affect their behavior and metabolism (Zhu et al. 2021).

On the other hand, MPs are unlikely to be absorbed via the skin since stratum corneum diffusion is limited to nano-sized polymers with a diameter of less than 100 nm (Gonçalves and Bebianno 2021). Personal and cosmetic goods, particularly in the body, include nanoplastics, as face washes are administered topically to the skin. Urea, glycerol, and -hydroxyl acids, all common constituents in body lotions, improved the nanoparticle's capacity to penetrate the skin barrier (Jatana et al. 2016). Oral administration of MP/NPs produces redox imbalance, disruption of energy homeostasis, and neurotoxicity in the gut, intestine, and kidney (Deng et al. 2017). Endocytic mechanisms allow polystyrene and PVC particles (150 nm) to penetrate the gut wall and end up in lymph nodes and the blood vascular system (Xu et al. 2019). A significant reduction in testosterone, luteinizing hormone, and follicle-stimulating hormone was reported with mice given 40 nm polystyrene NPs for 35 days (Amereh et al. 2020). Recent research found 20 MPs (50–500 μm) per 10 g of human feces, with nine distinct plastic types (most common of which were polypropylene and polyethylene terephthalate) (Shen et al. 2019). Organochlorines are well-known endocrine-disrupting compounds that increase cytotoxicity and inflammatory response (Prata 2018). NPs affect neustonic/planktonic habitats and act as surface receptors for accumulating organochlorines (OCs) and cause toxicological effects in organisms such as *Balaenoptera physalus* and *Cetorhinus maximus* (Fossi et al. 2014).

7 Sampling and analysis

Organizations, namely ICES, GESAMP, UNEP-MEDPOL and JPL Oceans, developed standardized methodologies on MP/NPs monitoring (GESAMP 2016).

7.1 Sampling

Wastewater streams, surface water, bottled water, sediments, untreated and treated tap water were collected by pumping, trawling, followed by sieving to separate MP/NPs particles of the desired size range

(Yang et al. 2019a, b; Koelmans et al. 2019; Raju et al. 2018). Auto-sampler, surface filtration, separate pumping, and containers are employed in collecting sampling for MP/NPs quantification and characterization from wastewater streams (Fig. 4). Separate pumping and filtration processes are efficient methods for collecting wastewater due to high solids, organic matter, and sampling volume ranging from hundreds to cubic metre litres (Song et al. 2018). Neuston nets, manta trawls, etc., are often used for sampling MP/NPs from aqueous samples (Song et al. 2018). Samples above 500 µm are fractionated with a stainless-steel filter and sorted under a stereomicroscope (Yang et al. 2019a, b; Koelmans et al. 2019). For identification, samples with size fractions less than 500 µm underwent enzymatic and oxidative purification. Sand samples from the shoreline identified MPs when sieved through 5 mm mesh followed by 2 mm (Ceccarini et al. 2018). Post-treated sewage sludge and anaerobically digested sludge were passed through 250 µm sieve to trap MPs and later filtered through 212, 63, and 45 µm sieves for particle size fractionation (Mahon et al. 2017). Samples from various WWTPs passed through the customized filtration device with an electric pump and identified MPs with a particle size of > 300 mm, 100–300 mm, and 20–100 mm (Talvitie et al. 2017; Li et al. 2021; Yang et al. 2019a, b; Cabernard et al. 2018; Song et al. 2018; Ceccarini et al. 2018; Dyachenko et al. 2017). Hermsen et al. (2018) suggested using fixatives like ethanol, formalin, or methyl aldehyde in sampling containers avoids retention of MPs. Ethanol and formalin have less effect on polymer, but it tends not to lose their characteristics (Sun et al. 2019). For NPs separation, magnetic field flow fractionation (MFFF), gel electrophoresis, and size-exclusion chromatography (SEC) were used (Li et al. 2021; Nguyen et al. 2019).

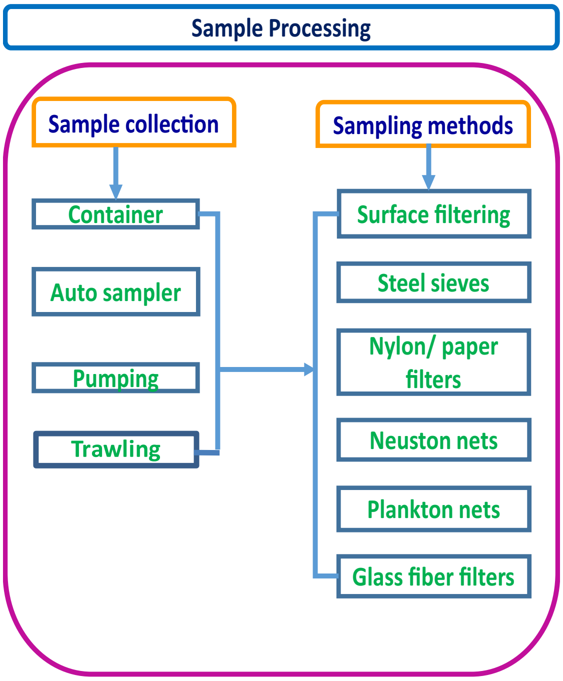

Fig. 4 Schematic representation of samplecollection and processing routesof micro/nanoplastics in the environment

7.2 Density separation

Density separation is also used to isolate MP/NPs from wastewater to limit their number in WWTPs (McGivney et al. 2020; Schwinghammer et al. 2020). This method separates MP/NPs from sludge, sediments, and other inorganic materials that have not been degraded during pre-treatment/breakdown (chemical or enzymatic) processes (Hermsen et al. 2018). The density of most plastics is almost equal to water, and MP/NPs particles suspended in water tend to float, making their removal process easy (Enfrin et al. 2019). High-density solutions like sodium chloride (NaCl), sodium iodide (NaI), potassium formate (K(HCOO)), zinc chloride ($ZnCl_2$), sodium tungstate dehydrates and sodium polytungstate are used in the density separation of MP/NPs (Table 2). NaI is expensive and hazardous to the environment according to the GHS classification (Enfrin et al. 2019). NaCl is recommended for sediment samples due to its low cost and non-toxic nature. Due to their lower density, most plastics tend to float on the surface and can be reduced up to 80% (Li et al. 2021; Zhao et al. 2017). $ZnCl_2$ is one of the inexpensive methods with a good recovery rate (Li et al. 2021). K(HCOO), after utilization, can be filtered and reusable (Mahon et al. 2017). However, sodium polytungstate is relatively expensive, whereas sodium tungstate dihydrate is recommended in standardized protocols for sediments (McGivney et al. 2020; Schwinghammer et al. 2020). Canola oil was also used to separate MPs (fibres and fragments) with 92–97% recovery (Stock et al. 2019). The dual-density isolation mechanism for separating MP/NPs was also reported (Zhao et al. 2017). First, a higher-density medium is used to separate plastic particles

Table 2 Various processes involved in density separation and pre-treatment methods of micro/nanoplastics in the environment

	Type of Polymer	Solution	Density (g/cm^3)	Advantage	Disadvantage	References
Density Separation	PP, PE	Water	1	Simple; Easy to operate	Limited with high-density plastics and less recovery rates with more errors	Lv et al. (2021); Li et al. (2021); Stock et al. (2019); Enfrin et al. (2019); Sun et al. (2019); Munno et al. (2018); Lares et al. (2018); Zhao et al. (2017); Coppock et al. (2017); Mahon et al. (2017); Löder et al. (2017); Dehaut et al. (2016); Avio et al. (2017)
	Microplastics with a density below 1.2 g/cm^3 (eg: PA, PE, PP etc.)	NaCl	1.2	Economically feasible and Non-toxic, Easy available and eco-friendly	Limited with high-density plastics and less recovery rates with more errors	
	Nylon, PVC, PET	Sodium polytungstate	1.4	Non-toxic and Highly efficient to extract certain high-density plastics	Not specified	
	PVC, PS, PET, PE,PUR, PP	NaI	1.6	Recovery rates are very high $> 99\%$	Cross contamination (Reacts with cellulose matter) Not eco-friendly and expensive	
	Almost all plastics	$ZnCl_2$	1.7	Easy and applicable for almost all plastics	High economic inputs and Toxic	
	Microplastics with a density below 1.8 g/cm^3 (eg: PA, PE, PP etc.)	Potassium formate	1.9	More microplastics can be extracted	Not specified	

	Digestion	Reagents	Effect on polymer
Pre-treatment	Oxidative	Fenton reagent; $K_2S_2O_8$; 5% NaClO; 30–35% H_2O_2	Decolouration of polyethylene, PET and polyester Degrades NY6 and NY66 and difficult in complete digestion of biomaterials at any temperature. Digestibility rates are $< 95\%$
	Acid	37% HCl; H_2SO_4; 65% HNO3	Digestibility rates are $> 95\%$. Higher concentrations of acids cause degradation of polymers and significant decolouration of polymers. Acids are Corrosive to NY6 and NY66 and melts PET as well

Table 2 continued

Digestion	Reagents	Effect on polymer
Alkali	10% KOH; NaOH	Higher concentrations of NaOH causes partial destruction of the nylon fibers, melting of the polyethylene, and colour change of plastic particles. KOH at 40 °C is measured to be the destructive treatment of all polymers
Enzymatic	Proteinase K; Trypsin; Cellulose; Chitinase; Lipase rtc	Low concentration of trypsin, Proteinase K etc. has higher digestion. Enzymatic digestion efficiencies are 88% and does not effect on microplastics. The optimized concentrations can digest > 95% in the phytoplankton of seawater samples without damaging any MPs fragments

such as polyethylene (PE), polypropylene (PP), and polystyrene (PS) from heavier components (minerals). Second, adding calcium fluoride to a low-density solvent (pure methanol) enables selective sedimentation of plastic enabling effective recovery via centrifugation. These solvents don't harm the plastic polymers, except polypropylene, since the density is relative to solvent (Fries et al. 2013). The density separation is not ideal for larger WWTPs due to MP/NPs, and the procedure must be done in the static mode because rushing water would disperse the particles (Stock et al. 2019).

7.3 Pre-treatment

Wastewater in general consists of suspended matter, sediments, solid biomass, and organic material. Sediments and solid biomass can be easily separated by picking or filtration, but the complexity comes with suspended and organic matter. Heating of the sample at higher temperature damages MP/NPs due to degradation or mechanical friction (Sun et al. 2019). Prior digestion, samples need to be treated due to high concentrations of organic matter or inorganic solids materials. The biofilm formation on MPs with smaller size (0.063 mm) is an alternative to the digestion process and enables recovery to some extent. However, this method has many drawbacks like inefficiency for large-scale samples, cost-effectiveness, time-consuming, and limited recovery (Felsing et al. 2018; Rummel et al. 2017). Organic content in the waste samples is pre-treated/digested mostly by chemical digestion and enzymatic digestion.

7.3.1 Chemical

The chemical digestion methods can be classified into acid, alkali, and oxidation treatments. Acidic digestion was performed to reduce organic matter in the samples using H_2SO_4, HNO_3, and HCl (Li et al. 2021; Avio et al. 2017). A maximum of 98% degradation was noticed using HNO_3 compared to other acids. It has a disadvantage with polymers PS and PE since these particles get agglutinated during dissolution. HCl is less preferred due to its inefficiency in destroying bulk organic matter (Cole et al. 2014), while H_2SO_4 deform and degrade the plastic along with the organic compounds. Alkali digestion was carried out using strong base solutions like NaOH or KOH. The use of 1 M NaOH shows the efficiency of $\geq 90\%$ organic digestion (Cole et al. 2014). Further increase in molarity, polymers like PE, PET, PVC, and cellulose acetate start degrading into simple forms (Munno et al. 2018). Upon exposure for a longer time, NaOH degrades the MPs, but KOH retains them from degradation except for cellulose acetate (Dehaut et al. 2016). Oxidizing digestion is one of the commonly used methods for digesting organic matter. NaClO and H_2O_2 (30%) are efficient oxidizers (Li et al. 2021; Sun et al. 2019; Lares et al. 2018). Fenton reagent ($FeSO_4/H_2O_2$) is generally referred to as an advanced oxidizing agent used in digesting soil organic matter, compost, and large volumes of wastewater (Tagg et al. 2017; Masura et al. 2015). The reagent degrades MPs with less than 5 mm and facilitates biological samplings (gut or tissues) (Tagg et al. 2017). MPs can be recovered up to 70% when digested with 30% H_2O_2 for seven days (Liu et al. 2019). Further increase in H_2O_2 concentration, deformation of plastics was observed. So, using a lower concentration of H_2O_2 and Fenton reagent is recommended with an exposure time of 12–24 h (Munno et al. 2018; Zhao et al. 2017). Recently, a metal-doped particle approach was adopted for NPs extraction. Pd-doped nanoplastic particles were spiked into ultrapure water with microwave-assisted acid digestion to assess the presence of NPs (Hildebrandt et al. 2020).

7.3.2 Enzymatic

Enzymatic degradation is the alternative method for chemical digestion for removing organic matter. It digests biological/aquatic tissues and organic matter in the waste samples. Enzymes such as cellulose, chitinase, protease, trypsin, papain, collagenase, and lipase were used in various studies (Sun et al. 2019; Courtene-Jones et al. 2017). Enzymatic digestion using protease showed >97% degradation (Cole et al. 2014). Löder et al. (2017) conducted experiments using a combination of enzymes. It modified the process and incubation conditions, where the whole degradation procedure of organic matter could not take more than one week. Utilizing enzymes for MP/NPs digestion is safe, but this process is limited to low quantity samples as it is an expensive process. This process does not apply to large-scale treatment as each enzyme requires its optimum pH for its functioning and digestion of samples (Stock et al. 2019; Löder et al. 2017).

7.4 Identification

Detection and predicting the formation of MP/NPs in wastewater is complex to perform (Zhao et al. 2017). Various analytical protocols to identify the size, quantity, and types of plastic particles have been used (Table 3). Sieve pattern and size create a primary cut-off for particles present in the samples (Li et al. 2021; Lv et al. 2019; Magnusson and Norén 2014). The simple method for identifying plastic matter is the ‘hot-point test’ where heated needle, when placed in the sample, plastic gets melted and masks over the needle, which will not permit identifying the nature and type of plastic (Hebner and Maurer-Jones 2020). Transmission electron microscopy (TEM), Field emission scanning electron microscopy (FE-SEM), dynamic light scattering (DLS), and multi-angle light scattering (MALS) with pyrolysis detect particles of size (10 nm) and provide finite pictures/patterns. Energy-dispersive X-ray spectroscopy (EDX) coupled with FE-SEM reveals the elemental composition of MP/NPs (Li et al. 2021; Sun et al. 2019; Long et al. 2019; Corcoran et al. 2009). Fourier transform infrared spectroscopy (FTIR), focal plan array-based systems (FPA), micro-FTIR, and Raman spectroscopy are used to identify functional groups of polymers (Sun et al. 2019; Duemichen et al. 2015; Fries et al. 2013), while the structural architect of plastics can be identified using thermogravimetric analysis-gas chromatography-mass spectrometry (TGA-GC–MS), pyrolysis–gas chromatography-mass spectrometry (GC/MS) and thermogravimetric analysis (TGA) with thermal desorption–gas chromatography-mass spectrometry (TED-GC/MS) (Li et al. 2021; Lv et al. 2019; Duemichen et al. 2015). However, these techniques are sensitive to analyzing MP/NPs with impurities. Prior pre-treatment processes like wet peroxide oxidation, enzymatic digestion, and oxidizing agents facilitate the detection of MP/NPs. For nanoplastics, identification through these methods becomes more challenging when the sample size exceeds 500 mm, and the weight is less than 10 mg.

7.4.1 Microscopy

Identification of MPs using a microscope can be classified mainly by Optical/visual microscopy, Scanning electron microscope (SEM), and Transmission electron microscopy (TEM). The plastic particles were identified based on surface, and chemical structures as MPs do not shine and exhibit ductility/ malleability under physical stress (Zhao et al. 2017). MPs collected from various wastewater streams were passed through a filtered mesh and categorized into wires, needles, fibres, clusters, fragments, irregular partials, etc., with two or three-dimensional shapes (Hebner and Maurer-Jones 2020). However, visual observation of MPs is difficult due to the lower magnification factor of microscopes. MPs are size constrained and have a high error ratio (Uurasjärvi et al. 2020; Lv et al. 2019). Filtered MPs from the secondary wastewater treatment plants were observed under stereo-microscope, confirming bead-shaped MPs with 87% recovery (Dyachenko et al. 2017). Waste sludge samples free of organic matter observed fibres and particles when visualized under a digital optical microscope with

Table 3 Various methods of identification and characterization of micro/nanoplastics in environmental samples

Processes	Technique	Range/ Limits	Advantage	Drawbacks	References
Pre-concentration	Membrane filtration/ Ultra-filtration	> 10 nm 5–50 nm	Easy and cheap process, Process larger volumes No sample damage/ aggregation	Frequent interaction with membrane. Low flow rates with small pores and less volumes	Li et al. (2021); Schwinghammer et al. (2020); McGivney et al. (2020); Uurasjärvi et al. (2020); Lv et al. (2019); Schwaferts et al. (2019); Xu et al. (2019); Lv et al. (2019); Erni-Cassola et al. (2017); Ziajahromi et al. (2017); Shim et al. (2016); Fries et al. (2013)
	Dialysis	5–50 nm	Performed at mild condition	Slow process and requires large volume of counter dialyzing medium	
	Ultracentrifugation and Analytical ultra-centrifugation	1 nm–1 μm	Simple and easy operation Can provide more information on MPs/NPs Very effective in separation	Operated at harsh conditions and no separation from particulate matrix Difficult to obtain complete separation	
Separation techniques	Solvent evaporation	All particulates	Easy and economic	Requires more energy Difficult to remove dissolved matter	
	Asymmetric Flow Field Fractionation	1 nm–1 μm	There is no stationary phase and done by online monitoring	Difficult in operation, Interaction with membrane and Steric inversion	
	Size Exclusion Chromatography	1–100 nm	Fixed with Coupled detectors	Operated at small range of samples and stationary phase	
	High Performance Liquid Chromatography	1–40 nm	Fixed with Coupled detectors	Operated at small range of samples and stationary phase	
	Capillary Electrophoresis	5–500 nm	Fast and high separation rates Fixed with Coupled detectors	External charge required, Electrolyte/surface modification Might damage sample by Interaction with capillary/clogging	

Table 3 continued

Processes	Technique	Range/ Limits	Advantage	Drawbacks	References
	Hydrodynamic Chromatography	5 nm–1.2 µm	Minimal interaction with stationary phase	Application is very less	
Characterization technique	Dynamic Light Scattering (DLS) (Size (dh), PSD, aggregation behaviour)	1 nm–3 µm conc. 10^{-6} to 10^{-1}	Easy, fast and economic Process is In- situ, Non-invasive, Aggregation and direct coupling	Large particles, Polydispersity with complex matrix, Non-spherical particles	
	Electrophoretic Light Scattering (ELS) (Surface charge, stability)	1 nm–3 µm	Fast and cheap Non-invasive	Involves Electro-osmotic effect and Sensitive to environment	
	MALS (Size (dg), PSD)	10–1000 nm	Fixed with Coupled detectors	Prior sample preparation is done, requires neat and clear samples	
	Laser Diffraction (LD) (Size)	10 nm–10 mm	Measures large size range Process is easy, fast and automated	Only spherical model	
	Nanoparticle Tracking Analysis (NTA) (Size (dh), PSD, number concentration)	30 nm–2 µm	Better with polydisperse samples, complex, media, Particle corona	Complexity in operation	
	FPA-FT-IR (Vibrational spectrum, Pigments, Additives, Ageing)	> 10 mm	Non-destructive and Automated process	Not applicable for single subµ- and nanoplastic Easy interference from water takes place	
	ATR-FT-IR; AFM-IR (Spectrum, imaging)	> 50 nm	Advanced, Simple and fast performing method AFM-IR Spectrum, imaging > 50 nm, High resolution and Chemical imaging	Slow and covers small area	
	Raman Microspectroscopy (Fingerprint spectrum, Pigments, Additives)	> 0.5 mm, Bulk	Non-destructive, Easy sample preparation, Fast and no interference from water	Works on Fluorescence	
	XPS (Binding energies of orbitals)	Bulk	Surface scanning and measurement	UHV and Laborious	

Table 3 continued

Processes	Technique	Range/ Limits	Advantage	Drawbacks	References
	Py-CG-MS (Mass Polymer type Additives)	Bulk Limit: ng-mg	Measurement with matrix, Fast, Higher sample masses can be done	Dependent on type of polymer, Dry sample needed Prior sample preparation is necessary	
Imaging studies	Optical Microscopy and Fluorescence Microscopy (Size, shape, morphology)	> 1 µm	Non-destructive, Cheap, easy to handle Operated at sub-diffraction variants	Limited Diffraction, Environmental plastic is not fluorescent	
	AFM (Size, shape, topography, aggregation)	> 0.1 nm	High resolution and AFM-IR TERS and liquid	High resolution, AFM-IR, TERS and done in liquid	
	STM; SEM (Size, shape, surface morphology, aggregation)	> 1 nm	High resolution imaging	Conductive samples, Slow and covers small area Prior sample preparation is involved and has charge effect	
	TEM (Size, shape, aggregation, imaging)	< 1 nm	High resolution imaging, Precise size information can be produced	Quantification is difficult Prior sample preparation is involved and expensive process	
	EDS (Elemental composition)	nm range	Complementary to SEM, measurement of all elements	Few Elemental information is not given	
	Confocal Laser Scanning Microscope (CLSM); Near-field Scanning Optical Microscopy (NSOM) (Size, shape, location in)	> 1 µm	Non-destructive Fluorescence imaging	Covers small area and Diffraction limit	
	Environmental scanning Electron Microscopy (ESEM) (Size, shape, imaging, surface morphology)	> 0.1 nm	Wet samples can be done at environmental conditions	Reduced in resolution	

magnification up to 1000X using built-in software (Lares et al. 2018). Particles with shapes such as round, flat and oval fibres have a twisted, layered, and tapered nature. Hebner and Maurer-Jones (2020) conducted experiments to observe MPs photo-degradability in simulated moving water systems and noticed cracks in polymer linear or structural alignments irrespective of their thickness upon exposure to UV light. Distinguishing the difference between transparent, pale-colored MPs and natural polymers (cellulose and fibrils) microscopy requires higher magnification (Dris et al. 2015; Magnusson and Norén 2014). Dyes such as Rose Bengal, Oil red EGN, Eosin B, Hostasol Yellow3G, fluorescein, safranine, isophosphate, and Nile Red are used for staining MP/NPs (Lv et al. 2019; Ziajahromi et al. 2017; Shim et al. 2016). Nile Red is used to identify MPs, i.e., polyethylene, polypropylene, polystyrene, polycarbonate, polyurethane, nylon particles, and poly (ethylene–vinyl acetate) with sizes ranging from 20 µm to 1 mm. Its fluorescence is stable for two months and is mostly used for marine and environmental pollutants (Lv et al. 2019; Ziajahromi et al. 2017). Rose-Bengal solution (4,5,6,7-tetrachloro-20,40,50,70-tetraiodo-fluorescein) is used to differentiate natural fibres and MPs since the stain gets adsorbed only by natural polymers and allows easy visual separation (Lv et al. 2019). Karakolis et al. 2019 carried out their experiments by staining more than ten fluorescent dyes on plastics, and excitation and emission colours were studied using fluorescence microscopy. Application of dyes could identify MPs in the micrometre to nano scale level (Karakolis et al. 2019).

Scanning electron microscopy (SEM)is used to characterize the surface topography of MPs. The pretreated polymers samples were first coated with gold (Long et al. 2019; Corcoran et al. 2009). SEM analysis showed the dominant shapes like fragments, pellets, and fibres with average particle diameters of 11 mm from sediments and 130 mm from water suspensions, including polypropylene, polyethylene, polystyrene, and polyethylene terephthalate (Long et al. 2019). Transmission electron microscopy (TEM) effectively visualizes NPs for less amorphous particles; simplified sample preparation to electron microscopy and heavy-metal stains may be necessary (Hebner and Maurer-Jones 2020). By combining the Brownian motion properties, the degradation studies of NPs can be analyzed effectively (Hildebrandt et al. 2020).

7.4.2 Elemental composition

The elemental analysis gives the individual composition of elements, whereas CHNSO analysis predicts only carbon, hydrogen, nitrogen, sulfur, and oxygen ratios present in the sample. However, detection of MPs could be difficult if sampling matter contains impurities or organic compounds and can be analyzed only by comparing with standard polymers (Li et al. 2021; Sun et al. 2019). Energy-dispersive X-ray spectroscopy (EDS) was used to assess the elemental composition of MPs based on diffraction and reflection of radiation emitted from MP/NPs surfaces (Sun et al. 2019; Fries et al. 2013; Dubaish and Liebezeit 2013). Fries et al. (2013) applied this technique to identify the inorganic plastic additives (IPAs) present in the samples. The elements such as aluminum, zinc, sulfur, barium, and titanium dioxide nanoparticles (TiO_2-NPs) were detected in marine (Fries et al. 2013). TiO_2-NPs are formed due to the degradation of polymer–TiO_2 composites in the marine environment.

7.4.3 Fourier transforms infrared (FTIR) spectroscopy

Fourier transforms infrared (FTIR) spectroscopy is used to detect MP/NPs chemical constituents/moieties (carbon-linked substituents) of MP/NPs in wastewater, sediments, and biota samples. It aids in the identification of plastic polymer particles by comparing their characteristic infrared spectra of carbon attached functional groups to reference spectral libraries, where each characteristic peak corresponds to chemical connections between atoms (Schwinghammer et al. 2020; McGivney et al. 2020; Zhang et al. 2016; Murphy et al. 2016). Cincinelli et al. 2017 identified rayon, polyamide, and semi-synthetic cellulose in gastrointestinal fish tracks using FTIR. Uurasjärvi et al. (2020) identified polyethylene, polypropylene, polymethyl methacrylate, polyvinylchloride, polyethylene terephthalate, and polystyrene in surface waters of lake samples. Lares et al. (2018) followed FTIR spectroscopic approach to confirm the presence of plastics and non-plastic organic particles (cellulose) in wastewater sludge. However, it is labor-intensive and needs sample preparation before identification. Attenuated total reflection FTIR spectroscopy (ATR) µ-FTIR can detect irregular shaped and scattered MPs effectively than FTIR (Hidalgo-Ruz et al. 2012). Hebner and Maurer-Jones. (2020) conducted their studies to assess the photo-oxidation of plastic fragments collected from the North Pacific Gyre sample using (ATR) µ-FTIR spectra and compared them against the spectral library. Carbonyl content was seen at the maximum value of 1700–1800 cm^{-1}. Chen et al. (2019) analyzed synthetic PVC MPs exposed to light and UV radiation using ATR-FTIR. PVC polymer has a characteristic absorption peak at 600−700 cm^{-1} region associated with C−Cl stretching vibration. Upon UV irradiation, C−Cl bonds at 613 and 711 cm^{-1} were decreased and replaced by C=C or C=O bonds resulting in dehydrochlorination of PVC. Similarly, Dyachenko et al. (2017) examined MPs from secondary wastewater treatment plant (WWTP) effluent using (ATR) FTIR spectrometer combined with continuum microscope in transmission mode. Polyoctadecyl methacrylate, amides, polyacrylonitrile, polyethylene, and non-plastic cellulosic cotton were detected compared with library references. Focal plane array (FPA) detector-based micro-FTIR imaging could detect MPs with sizes smaller than 20 µm (Schwinghammer et al. 2020; McGivney et al. 2020; Lv et al. 2019; Tagg et al. 2017). Mintenig et al. (2017) used FPA-based transmission micro-FTIR to identify MPs in wastewater and sludge samples, limiting fibre size (10–20 µm) and lateral resolution. Xu et al. (2019) collected 68 influent and 72 effluent samples from WWTPs and, upon analysis, found 112 plastics of 14 different types, which includes polyethylene (PE), polyamide (PA), polypropylene(PP), polystyrene (PS), polyethylene terephthalate (PET), rayon, polyvinyl chloride (PVC), poly methylmethacrylate (PMMA), rubber, polyethylene and polyether urethane (PU), polypropylene copolymer (PE-PP), acrylonitrile styrene copolymer (AS) and polyacrylate.

7.4.4 Raman spectroscopy

Raman spectroscopy analyzes MPs based on the inelastic scattering of light and delivers the evidence of chemical compounds in the form of molecular vibrations by a spectrum. Plastic particles of size less than 1 µm are detected with better response and reliability to other non-polar functional groups, and it is unaffected towards signals caused by H_2O and atmospheric CO_2 (Schwinghammer et al. 2020; McGivney et al. 2020; Li et al. 2018; Zhao et al. 2017; Hidalgo-Ruz et al. 2012). Raman spectroscopy linked with a microscope favors robust and non-presumptive identification of plastic components with various sizes (Lv et al. 2019). Raman spectroscopy was performed with excitation at 455−532 nm on gold-coated polycarbonate filters or silicon filters (Kelkar et al. 2019; Schymanski et al. 2018). Several studies often used excitation at near-infrared (NIR, 785 nm) with laser intensity of 5–7 mW and integration times of 15–30 s in the wavelength range between 200 and 3000 cm^{-1} to identify MPs without deformation and compared with Raman library (Li et al. 2021; Cabernard et al. 2018; Zhao et al. 2017; Hidalgo-Ruz et al. 2012). Cole et al. (2014) suggested coherent anti-Stokes Raman scattering (CARS) microscopy based on confocal laser scanning microscopy (CLSM) and a synchronized dual-wavelength picosecond laser source. It can localize polymeric units in biological tissues with subcellular precision and gives spectral signals, but this process is limited in sensing the fluorescence samples. In such cases, samples free from fluorescence are recommended for Raman measurements (Sun et al. 2019; Li et al. 2021). Micro-Raman spectroscopy for High-density polyethylene (HDPE), Polypropylene, Polystyrene polymers before and after chlorine disinfection during sterilization and detected changes in Raman intensities by total loss of existing Raman peaks and emergence of new chemical bonds resulting aggressive effect of chlorine during water treatment (Kelkar et al. 2019). HDPE has prominent Raman peaks at 1064 cm^{-1}, 1130 cm^{-1}, 1295 cm^{-1}, and 1416 cm^{-1}. A shift was noted with increased chlorine dosage, and a new chlorine carbon bond peak at 678 cm^{-1} was detected. Polypropylene did not show any degradation effect on chlorination, and no new bonds formation was seen in Raman spectra. While in Polystyrene, new intense Raman peaks at 400 cm^{-1}, 445 cm^{-1}, and 348 cm^{-1} were observed due to the oxidative stress created by excessive chlorination. Long et al. (2019) performed micro-Raman spectroscopic analysis for WWTPs and reported the presence of plastics such as 21% polyethylene, 31.6% polypropylene, 10.1% polystyrene, 7.5% polyethylene terephthalate, and 9.2% propylene copolymer. Zada and co-workers used Stimulated Raman scattering (SRS) microscopy and worked on coherent interaction of two separate laser beams with vibrational levels to identify MPs. Different polymers such as polypropylene, high-density polyethylene, polyamide 6, 6, and polyethylene terephthalate are common in environmental or consumer products. (Li et al. 2021; Schwinghammer et al. 2020; McGivney et al. 2020; Zada et al. 2018). Zhao et al. (2017) identified 11 chemical matrices including 6 polymers, 5 plastic additives, and pigments 100 to 500 mm in size from marine sediments using Raman microscopy.

7.4.5 Mass spectroscopy

Some of the commonly reported analyses for MP/NPs identification are s liquid chromatography (LC), gas chromatography coupled with mass spectrometry (GC–MS), liquid chromatography-mass spectrometry (LC–MS), and Pyrolysis- gas chromatography–mass spectrometry GC–MS (Py-GC/MS) (Sun et al. 2019; Nguyen et al. 2019). Among them, the Pyrolysis GC/MS technique gained much attention in recent years as GC–MS, LC–MS analytical methods consist of multiple steps such as solvent extraction, filtration, and sample preparation before chromatographic injection, which are time-consuming processes (Li et al. 2021; Sun et al. 2019; Lv et al. 2019). In the Pyrolysis GC/MS technique, samples were analyzed by injecting MP/NPs directly into the pyrolyzer, preventing contamination during sample preparation by improved analytical efficiency (Nguyen et al. 2019; Schwinghammer et al. 2020; McGivney et al. 2020). Hermabessiere et al. (2018) worked extensively on Py-GC/MS method and determined the limit of detection (LOD) values for eight commercial plastics, i.e., polyethylene (PE), polystyrene (PS), polypropylene (PP), polycaprolactam (PA-6), polyethylene terephthalate (PET), polycarbonate (PC), poly(methyl methacrylate) (PMMA), and unplasticized polyvinyl chloride (uPVC). They developed LOD standards ranging from 1 μg and applied them to environmental samples from beach sediments, surface seawater, and organisms. The average of all obtained masses was calculated by using intensity (h), time (t), and response factor (R) (Hermabessiere et al. 2018).

$$\text{Response factor (R)} = \text{Weight/Height}$$

$$\text{Limit of detection (LOD)} = 3 \times \text{R} \times \text{H}_{20\text{FWHM}}$$

Fries et al. (2013) conducted experiments on MPs and bound additives in the marine environment using Pyr-GC/MS. Pyrolysis results selected from standard polymers showed the presence of polystyrene, polypropylene, polyethylene, polyamide, chlorinated and chlorosulfonated polyethylene, diethyl phthalate, diethylhexyl phthalate, 2,4-di-tert-butylphenol, etc. from the analyzed samples. However, environmental samples contain a mixture of complex structured molecules where the difficulty arises in determining the exact polymer upon pyrolysis. Duemichen et al. (2015) used thermogravimetric analysis (TGA) with thermal desorption– gas chromatography-mass spectrometry (TED-GC/MS) for handling the complex structured environmental samples of soil suspended solids of wastewater and identified mostly PE along with PS, PET, PS, and polyamide 6. NPs with lower concentrations can be analyzed by using Thermal desorption coupled with gas chromatography−mass spectrometry (TDS-GC–MS) with greater sensitivity of masses less than ~50 μg. This may be applicable for simple matrices like drinking water, where separation is straightforward (Nguyen et al. 2019).

7.4.6 Differential scanning calorimetry (DSC)

Differential scanning calorimetry analysis is used to evaluate polymeric MP/NPs by glass transition temperatures (Tg) and depicts chemical structures upon exposure to higher temperatures. Kelkar et al. (2019) applied DSC for physical changes and conversion of virgin plastics to other forms while sterilizing drinking water and wastewater disinfection using different chlorine concentrations. Initial analyses were performed with high-density polyethylene and polypropylene standards, and their respective glass transition temperatures (Tg) were noted. Concerning the traditional values, experimental samples were analyzed, and types of polymers present were identified with heating from ambient temperature to 300 °C at a heating rate of 5 °C/min and cooled to −80 °C and enthalpy changes were noted. The DSC analysis upon plastic particles exposure to extreme chlorination conditions resulted in a substantial change, denoted by the difference in melting points and characters of stressed plastics. Likewise, Hebner and Maurer-Jones (2020) calculated DSC exotherms heating and cooling cycles of plastics in moving water. The polymer samples identified have an increased crystallinity, which might be due to the photodegradation of polymers. Plastic polymers melting at low temperatures or glass transition temperature (Tg) show that these plastics having lower molecular weight polymers were attributed to polypropylene and polyethylene samples.

8 Interaction energy calculation by DLVO theory and kinetic model

MP/NPs generation and interactions are necessary to assess its surface chemistry, size and shape in WWTPs processes to predict the existence of plastic nature, which is either single or agglomerate state particles in water (Andrady 2017; Li et al. 2019). Derjaguin, Landau, VerweyOverbeek's (DLVO) theory is used to study the surface energy interactions between suspended solids-polystyrene nanoplastics (PSNPs) and suspended solids-polyethylene MPs (PMPs) particles by applying particle-plate geometry in DLVO calculations (Enfrin et al. 2019). Using theoretical calculations, the interaction energies were treated as particle–particle geometry for PSNPs-PSNPs and plate-plate geometry for PEMPs-PEMPs. Enfrin et al. (2019) explained homogeneous and heterogeneous interactions between the nano/microplastics in water. In homogenous interaction, the surface interaction energies of colloidal MP/NPs have less than 1 μm in water as per DLVO theory. From the interparticle distances, the electrostatic repulsion and van der Waals attraction energy between the particles results in agglomeration/aggregation rate in the sample. The potential energy between interparticle distance and kinetic energy due to motion forms a huge effect on dense aggregation/agglomeration and other parameters such as surface area, charge, and ionic strength of sample suspension. In heterogeneous interaction, the aggregation occurs between NPs/MPs and organic matter or microorganisms. Zhao et al. (2017) studied the marine organic aggregate encapsulate MPs like PP, PS, and PE resulting in the formation of agglomeration. Here, the secretions of microbial extracellular polysaccharides form an interfacial bonding between the plastic particles and the chemicals in the water create the surface interactions between plastics, resulting in agglomeration. As a result, the dispersion pattern can be improved by inserting nucleophile chloro-halides groups to boost negative charge surface density on MP/NPs. Li et al. (2018) looked at how different monovalent and divalent electrolytic solutions affected the aggregation behavior of polystyrene MPs. The aggregation kinetics were achieved by DLVO theory. They have also studied critical coagulation concentration for the aggregation and interaction energies between polystyrene MPs using a combination of theoretical calculations and experimental measurements that could be useful for assessing MP/NPs fate in aquatic environments.

9 Removal of microplastics in WWTPs

Wastewater treatment plants (WWTPs) interface natural and mechanical systems. Large quantities of particulate plastic resulting from industrial, domestic, and surface runoff were treated before release into the environment (Frehland et al. 2020; Hu et al. 2019). MPs, unlike suspended solids in standard WWTPs, necessitate the use of settling and filtration equipment to remove MPs (Iyare et al. 2020). Three distinct WWTPs in New York recorded daily particle discharges of 109, 556, 81, 911 particles (Blair et al. 2017). The WWTPs may not completely remove MP/NPs, but treatment processes involving degradation, generation, and advanced treatment sources should be considered. Nevertheless, MP/NPs concentration downstream is higher than those upstream, suggesting the enormous release of particulate plastic into the natural system is conceivable. The removal of MP/NPs by the WWTPs can be observed from primary, secondary, and tertiary treatments.

9.1 Primary treatment

Primary treatment removes suspended solids and most of the MP/NPs in wastewater. It was evident that preliminary (mechanical) treatment could remove 35–59% of the microplastics, while primary treatment could remove 23–53% (Hale et al. 2020). Pretreatment was able to remove bigger microplastics effectively. Rough grille bars (16–25 mm) and fine grille bars (3–10 mm) used in WWTPs independent MPs of smaller size (Raju et al. 2018; Michielssen et al. 2016). According to Dris et al. (2015), the fraction of massive particles (1000–5000 μm) reduced from 45 to 7% after the preliminary treatment. Surface skimming on primary clarifiers removed light floating MPs, whereas gravity parting removed MPs bound in solid flocs (Simon et al. 2019). Microbeads made of PE were effectively removed by skimming due to their buoyancy and floating on the top of water, grease, and oil particles (Murphy et al. 2016). These findings were consistent with Michielssen et al. (2016) and Sutton et al. (2016), where a considerable decrease of microbeads was noticed in the outflow of the WWTPs after primary treatment. In terms of MPs morphologies, pre-treatment may be more effective at removing fibres from wastewater than pieces (Ziajahromi et al. 2017). This is most likely because fibres are abundantly captured in flocs and sedimentation (Talvitie et al. 2015; Magnusson and Norén 2014).

9.2 Secondary treatment

Secondary treatment typically includes biological treatment/clarification, which reduces MPs in wastewater by 0.2% to 14% (Sun et al. 2019; Besseling et al. 2017). Plastic debris production in the secondary clarifying tank is likely to be aided by sludge flocs or bacterial extracellular polymers in the aeration tank (Carr et al. 2016). Microplastics have an impact on microbial-mediated processes that affect ammonium synthesis (ammoniation) and reduction, as well as inorganic nitrogen biological conversion efficiency (nitrification and denitrification) (Zhang and Chen 2020). Furthermore, the concentrations of microplastics are positively linked with BOD, DO, and total phosphorous. Chemicals (ferric sulfate) and flocculating agents employed during secondary treatment may help to remove MPs by causing suspended particulate matter to combine and form a “floc” (Murphy et al. 2016). Due to protozoa or metazoan feeding, MPs might be trapped in sludge flocs (Scherer et al. 2018; Jeong et al. 2016). Secondly, prolonged interaction of MPs with wastewater aids secondary discharges and removes MP/NPs escaped from skimming/settling practices. These bio-coatings may operate as wetting agents, altering micro/nano plastics surface characteristics and relative densities (Rummel et al. 2017; Fazey and Ryan 2016). Large MPs particles can be removed even more thoroughly during subsequent treatment, resulting in a low abundance. The secondary treatment eliminates additional fragments than fibres due to their abundance and neutral buoyancy (Ziajahromi et al. 2017; Talvitie et al. 2015, 2017).

9.3 Tertiary treatment

The tertiary treatment provides further polishing to wastewater and removes microplastics by 0.2–2% relative to the influent (Sun et al. 2019). In this stage, MP/NPs suspended in the wastewater hinder coagulation, air flotation, microfiltration and affects the disinfection process (chlorine and UV disinfection) (Zhang and Chen 2020). Membrane-related technologies such as disc-filter (DF), rapid sand filtration (RSF), dissolved air flotation (DAF), and membrane bioreactor (MBR) are utilized in the treatment of tertiary effluents (Mintenig et al. 2017). The performance of several membranes in removing MP/NPs found thatMBRshowed relatively higher removal efficacy (99.9%) trailed by RSF (97%) and DAF (96%) (Talvitie et al. 2017). The exclusion of DF varied from 40 to 98.5%. Similar studies were conducted by Schneiderman (2015) in WWTPs and found an absence of microbeads with membrane filters while other advanced filters did. MPs concentrations were considerably lower following ultrafiltration and reverse osmosis (Ziajahromi et al. 2017). In some cases, fibres with fractions (20–190 mm) were found to be abundant in the final effluent of tertiary treatment (Ziajahromi et al. 2017). This is most likely since fibres can readily pass a filter or membrane longitudinally. MPs were reported to be removed from WWTPs in 98–99.9% of cases (Simon et al. 2018; Lares et al. 2018; Carr et al. 2016).

10 Strategies to reduce MP/NPs

Most interventions to prevent plastic leakage are related to removing MP/NPs from wastewater infrastructure (Pico et al. 2019). One of the effective solutions is to eliminate MP/NPs at the source itself. Improved solid waste infrastructure and management will reduce plastic debris dispersion into environmental matrices. The “GoJelly Project,” which employs jellyfish mucus to trap microplastics, is one example of microplastic prevention technology (Schmaltz et al. 2020). The “Cora Ball” and “Fibre Free” are laundry balls used to capture synthetic fibres in the washing machine. At the same time, the “Lint LUV-R” and “Showerloop” are household water systems that filter out microfibers (Schmaltz et al. 2020). Ban on the usage of microbeads alone prevents >2.9 trillion pieces of MPs from entering water bodies per year (Prata et al. 2019). Ban or imposing taxes on non-biodegradable plastics, single-use plastic cutlery, packaging materials, and labeling have been passed recently in various global jurisdictions (Karasik et al. 2020). Bio-based polymers and biodegradable plastics derived from renewable starting materials (starch, cellulose, lignin, etc.) are mostly biodegradable (Amulya et al. 2021; Kopperi et al. 2021). Bio-based polyethylene terephthalate (bio-PET), Bio-based polyethylene (bio-PE); Polyethylene furanoate (PEF), Polyhydroxyalkanoates (PHA), Poly(butylene succinate) (PBS), and Polycaprolactone (PCL) accounts for 40% of the global bioplastics used (Hahladakis et al. 2020; Prata et al. 2019). Swallowing biodegradable MP/NPs also affects health and the environment (Sadeghi et al. 2021).

Mitigation can be achieved through legislation (regulations on the management of plastic waste and products in various industries), technical (bio-innovations including bioplastics), and social awareness (reducing the use of single-use plastics or disposable plastics and adopting recycling habits) (Allouzi et al. 2021). The United States put forth a microbead ban in 2015 (H.R.1321 Microbead—Free Waters Act of 2015), and the European Union launched the single-use plastics Directive and proposed a ban on intentionally added MPs (EU 2018, Directive (EU) 2019/904) (Catarino et al. 2021). Between 2000 and 2019, near to 28 policies were come into force established on reducing and managing plastic pollution around the globe (Karasik et al. 2020). The Antarctic Treaty, London Convention and Protocol amendments, G20 Action Plan on Marine litter, International Convention for the Prevention of Pollution from Ships (MARPOL) Annex V, and the United Nations Environment Assembly (UNEA) on marine plastic pollution (Karasik et al. 2020). As of 2019, 39 regional policy documents have been adopted in Europe (Schmaltz et al. 2020). In India, the ’Plastic Waste Management Rules, 2016’ was implemented to remove all disposable plastic products by 2022 and prohibit the production of plastics below 50 µm (Ministry of Environment and Forest, India 2016; Veerasingam et al. 2020). Malaysia has implemented a “No Plastic Bag” campaign by imposing a minimum charge per plastic bag (Allouzi et al. 2021).

10.1 Circular economy (CE) approach

Recycling techniques cannot be a comprehensive solution to plastic reduction. Operational issues, infrastructure for collection and sorting, and lack of green methods are inherent limitations of recycling (Bishop et al. 2020). Circular Economy (CE) may be a viable option for reducing the harmful impacts of mass-produced plastics, rising consumption, and the usage of plastic items for short periods (Sandhiya and Ramakrishna, 2020). The end of the product life is one of the most significant contrasts between linear and circular economies. In the linear economics model, the management of a “post-use” product takes the form of dumping, whereas, in the CE model, end-of-life is determined as the product’s “first life” (Sadeghi et al. 2021). CE focuses on repair and reconstruction with the aim of continuous use. Managing a finite supply of natural resources and renewing natural systems, closing loops, and waste design are the essential principles of CE (Chae and Hinestroza 2020; Venkata Mohan et al. 2019). Economic considerations are impending for relooping low-value plastic products in the consumption cycle (Fadeeva and Van Berkel 2021). Retailers are crucial in establishing a consumption loop by establishing a viable link between producers and consumers (Fadeeva and Van Berkel 2021). Therefore, manufacturers should create products that cater to consumer preferences while using few virgin resources, reducing toxicity, and increasing recyclability. Adapting technology change in process control, equipment modification, onsite reuse, and recycling helps achieve production loss and curtailing plastic waste at the point source (AliAkbari et al. 2021; Venkata Mohan and Katakojwala, 2021). Mechanical, physical, chemical, biological, and energy recovery are the main recycling technologies employed for recovery (Shamsuyeva and Endres 2021). Recycling can be achieved through the “close-loop” and “open-loop” approaches. In closed-loop, plastic’s inherent properties stay unchanged, allowing it to be reused as a primary material (bottle-to-bottle recycling), whereas recyclate is used to produce new plastic products (bottle-to-fiber recycling) in an open-loop system (Shamsuyeva and Endres 2021). The recycling technologies are well established for managing single-polymer plastic waste (PET, PE, PP, PS, etc.) and mixed plastic waste (Hahladakis et al. 2020). Recycling rates for PET exceed 10%, while PS and PP are not yet started (AliAkbari et al. 2021). Recycling plastics will reduce GHG emissions from 8 $GtCO_2e$ to 4.9 $GtCO_2e$ by 2050 (Zheng and Suh 2019). From an environmental standpoint, the use of microbial polymers and plant-based fibres should be encouraged. The use of biodegradable materials, sustainable management of landfills, designing efficient recycling methods, providing support, incentive laws, and creating awareness among stakeholders can help gain traction in adopting CE (Sadeghi et al. 2021). Reprocessing and recycling also create jobs and save energy while downing resource consumption and waste (Chae and Hinestroza 2020).

11 Conclusion and future perspectives

Misuse and mismanagement of plastics have led to the accumulation of MP/NPs in the environment, posing a risk to the ecosystem and living beings. They are ubiquitous and released into different environmental compartments (terrestrial, aquatic, and atmospheric depositions) via rivers, lakes, stormwater runoff, sewage, sludge, and wastewater treatment plants. The fate and transport of MP/NPs strongly depend on the physicochemical properties of the plastics in soil and water. MP/NPs accumulation in soil affects the soil productivity, microorganism fecundity and inhibits plant growth. The trophic transfer of MP/NPs to human beings leads to cancer, digestive problems, and cardiovascular diseases. Research priority can be given to MP/NPs transport and accumulation in soil microbiome, crop plants, and trophic transfer of plastic-derived contaminants (additives and chemicals) in the food web. In MP/NPs analysis analytical front, developments in simple and precise tools, validations, and implications in detection, characterization, and quantification still need focus. The traditional use of Raman and FT-IR is already reinforced by the emergence of some promising techniques such as GC–MS performed with pyrolysis or TDS and especially new hyperspectral imaging techniques. Assessing and quantifying MP/NPs in the soil-microbiome-plant system still needs interventions in sample preparation, pre-treatment, and pre-concentration to meet the detection limit of the analytical instruments. In the context of NPs, sampling, pre-treatment, and characterization still lack standardization and homogenization of the analyte methods. In addition, it is necessary to establish matching libraries to identify MP/NPs from environmental samples. Particulate plastics from industrial, domestic, and surface runoff are entering wastewater treatment plants (WWTPs) before releasing into the environment. Increasing loads of MP/NPs in sludge digestion system reduces the effectiveness of process and raise operational expenses. Remediation technologies, prioritizing recycling, sustainable waste management, education and awareness, infusing circular models (reduce, reuse, recycle and recover), influxing bio-based materials (bioplastics), legislation, policy, and road map are some of the essential prerequisites that may be critically considered for the establishment of sustainable practices in plastic management domain.

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Declarations

Conflict of interest The authors declare no conflict of interest.

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