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Microplastic Toxicity in Aquatic Organisms and Aquatic Ecosystems: a Review

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Abstract Microplastics are pervasive pollutants and have been found in all environmental compartments globally, including aquatic ecosystems. Ingestion and trophic transfer of microplastics through aquatic species have been widely reported. Although a plethora of studies have reported that microplastics can be transferred through higher trophic level food webs with the potential for accumulation and toxicity, most microplastic aquatic toxicity studies have been conducted in laboratory studies. This means that studies within entire ecosystems or at environmentally relevant concentrations are lacking, representing a critical knowledge gap for ecotoxicological impact of microplastics on aquatic species and higher trophic level consumers (including humans). Thus, an understanding of aquatic ecosystem toxicity is still relatively unknown. To address this knowledge gap, this study provides a non-exhaustive summary of microplastic transport pathways, ecotoxicology, food web transfer, and examples of toxic pollutants sorbed onto microplastics in aquatic food webs. This study will

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guide future research priorities to address microplastic toxicity through aquatic food webs.

Keywords Microplastics (MPs) · Exposure · Ecotoxicology · Aquatic toxicity · Aquatic food webs · Food safety

1 Introduction

Plastic contamination affects growth, development, and survival of many aquatic species (Cole et al., 2011; Karbalaei et al., 2018). In 2019, global plastics production was 368 million tons (MT) and has grown 20-fold (Plastics Europe, 2020). Mismanagement of plastic waste has resulted in widespread plastic pollution in terrestrial and aquatic environments (Jambeck et al., 2015; Geyer et al., 2017; Borrelle et al., 2020; De-la-Torre et al., 2021; Rakib et al., 2021a,b; Rakib et al., 2022). Once in the environment, larger plastics continue to degrade into smaller fragments (Barnes et al., 2009). Microplastics (MPs), <5 mm, are the major contributor of plastic pollution in aquatic ecosystems (Thompson et al., 2004; Karbalaei et al., 2018). MPs pose threats to aquatic biota including phytoplankton, zooplankton, molluscs, and fish species, which may enter the human food chain (Wagner et al., 2014; Horton et al., 2017; de Souza Machado et al., 2018).

MPs are pervasive pollutants and have been found in all environmental compartments globally, including aquatic ecosystems (Allen et al., 2022). Although occurrence and presence of MPs have been widely reported in aquatic biota, transport pathways, toxicity profiles, and subsequent trophic transfer are still largely unexplored. Despite a plethora of studies on the presence or occurrence of MPs in aquatic biota, studies on exposure or potential toxicological effects are lacking. MPs can be mistaken for food and ingested by zooplankton, which poses a threat to higher trophic level aquatic consumers and food chains (Cole et al., 2013). Primary and secondary consumers, such as aquatic plankton, fish, and crustaceans, in marine and freshwater ecosystems may accidentally ingest MPs during feeding (Moore et al., 2011; Lechner et al., 2014; Wang et al., 2017). This poses safety concerns for MPs in seafood destined for human consumption or indirectly via aquaculture-based feed (Rochman et al., 2015; Hanachi et al., 2019; Karbalaei et al., 2019, 2020; Sequeira et al., 2020).

The occurrence of macroplastic and MP pollution has been widely documented in marine and freshwater environments (Rakib et al., 2022; Hatami et al., 2022; Walker et al., 2006; Corcoran et al., 2009; Imhof et al., 2013; Biginagwa et al., 2016; Ambrose et al., 2019; Enyoh et al., 2021), and in wastewater (Horton et al., 2017). MPs have also been reported throughout the entire water column, with the highest concentrations usually found in bottom sediments (e.g., Bergmann et al., 2017). For example, approximately 62,200 MPs/kg was reported in benthic river sediments in the River Mersey, UK (Hurley and Nizzetto, 2018). MPs also sorb inorganic and organic contaminants at varying concentrations onto their surfaces, acting as vectors for trophic transfer of pollutants within different media (Van et al., 2012; Rochman, 2015). Although there is increased knowledge on the prevalence and abundance of MPs in freshwater and marine habitats, there is still a lack of data on monitoring and behavior of associated hazardous pollutants sorbed by MPs (Hurley et al., 2017; Windsor et al., 2019). Freshwater macro-invertebrates and fish have been reported to ingest MPs with subsequent trophic transfer to higher predators (Biginagwa et al., 2016; Peters and Bartton, 2016; Horton et al., 2018), including transfer to predatory birds (Hollander et al., 2016; Gil-Delgado et al., 2017; Bourdages et al., 2021).

Physical and chemical damage to aquatic organisms following MP exposure resulting in lethal and sublethal effects on aquatic food web consumers (e.g., zooplankton and phytoplankton) has also been widely reported (Au et al., 2015; Nobre et al., 2015; Watts et al., 2015; Jeong et al., 2016; Duis and Coors, 2016; Rehse et al., 2016), although most laboratory MP toxicity or trophic transfer studies have focused on freshwater algae (Farrell and Nelson, 2013; Nelms et al., 2018; Hanachi et al., 2022). Aquatic invertebrates have also been used as prey for predators to study MP trophic transfer across food webs resulting in lethal or sublethal effects, highlighting the threat to aquatic biota and aquatic ecosystems (Windsor et al., 2019).

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To better understand MP threats to aquatic biota and aquatic ecosystems, a plethora of studies have described the occurrence, abundance, fate, and transfer of MPs in aquatic ecosystems (e.g., Cole et al., 2011; Auta et al., 2017; Karbalaei et al., 2018), including ecotoxicology of MPs and subsequent transfer through food webs in aquatic and terrestrial environments (e.g., Anbumani and Kakkar, 2018; Wang et al., 2019). Consequently, MP toxic impacts including alteration of metabolism in fish species have also been widely reported (e.g., Ahrendt et al., 2020; Karbalaei et al., 2021; Xu et al., 2020).

Although ecotoxicological studies on the distribution, fate, and interaction of MPs in aquatic environments exist, an updated and critical overview of MP interaction with inorganic and organic contaminants in aquatic ecosystems and their potential impact on food web toxicity, including food consumed by humans, is lacking (Karbalaei et al., 2018). National governments, such as the Canada Plastics Science Agenda (CaPSA), and international agencies, such as the United Nations Environmental Programme, have also recognized this as an important knowledge gap that needs to be addressed (ECCC, 2019; UNEP, 2020). Future research should investigate relationships between MP sorbtive potential and particle size, as well as standardizing laboratory MP toxicity studies on aquatic species throughout entire ecosystems. To highlight this research gap, this non-exhaustive critical review aimed to evaluate contemporary research related to MP transport, MP interaction with organic and inorganic contaminants, MP ecotoxicity, MP behavior, and potential MP impacts on food safety in aquatic ecosystems. This review also identifies future research priorities to address MP toxicity and food safety through aquatic food webs.

1.1 Methodology

This non-exhaustive critical review focused on contemporary literature to summarize MP studies on transport pathways, ecotoxicology, and food web transfer, along with effects of toxic organic and inorganic pollutants sorbed onto MPs in aquatic food webs. Peer-reviewed articles (i.e., both original and review papers) were reviewed from Google Scholar, Science Direct, Web of Science, and Scopus databases using some the following keywords: "fate of microplastics in aquatic ecosystems," "toxicity of microplastics," "vector effect of microplastics," "ecotoxicology of microplastics," "sorption of pollutants by microplastics," "mechanism of microplastic ecotoxicology," and "microplastics and human toxicology." Although this review did not adhere to protocol requirements (e.g., PRISMA), the main screening and acceptance/rejection of referral data were performed based on relevance of key findings to this review theme. Although not exhaustive, this critical review was designed to highlight some key studies and to help identify future research priorities to address MP toxicity and food safety through aquatic food webs. MP contamination in various environmental components and resulting food web toxicity has been extensively studied and reported. Thus, a focus of this review was to identify contemporary research uncertainties (i.e., no specific timeframe was used to limit the literature review) of MP contamination, MP vector effects of associated toxins, and MP ecotoxicology.

2 Transport Pathways of MP Through Different Environmental Matrices

MPs intentionally manufactured such as microbeads are primary MPs (Dris et al., 2016; Li et al., 2016; Gasperi et al., 2018). Degradation of larger macroplastics in the environment from heat, UV light, oxidation, and biodegradation create secondary MPs (Thompson, 2015). However, most MPs in aquatic and terrestrial ecosystems are secondary MPs derived from larger plastic fragments (Su et al., 2017).

Dominant modes of MP transfer from terrestrial to aquatic ecosystems include wind transport, soil migration, and via water transport (Boucher and Friot, 2017; Gasperi et al., 2018; He et al., 2018). Aquatic ecosystems accumulate MPs from multiple sources (e.g., wastewater treatment plants, sewer floods, soil washout, precipitation and from agricultural plastic products such as plastic mulching sheet, plastic packaging for fertilizers or pesticides, and plastic fragments in compost) (Duis and Coors, 2016; Horton et al., 2017; Ng et al., 2018) (Fig. 1). MP density and dimensions govern their transport and deposition within aquatic and marine ecosystems (Wagner et al., 2014). For example, wind transport and can recirculate MPs from coastal locations to the open ocean and vice versa. For example, ocean surfaces have been reported as both sinks and sources of MPs to

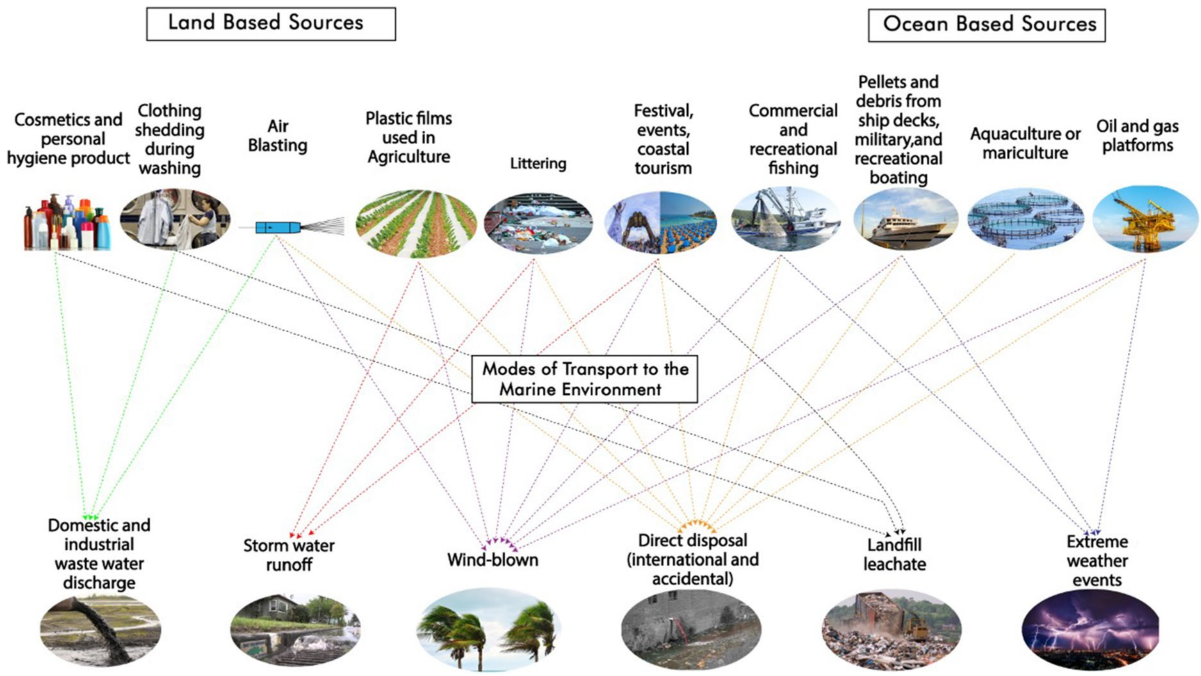

Fig. 1 Identification of sources and fate of MPs through multiple transport pathways in aquatic ecosystems

terrestrial ecosystems via seaspray and atmospheric wind transport (Allen et al., 2020). Storm events are also responsible for atmospheric MP transport across huge distances (Sul and Costa 2014; Li et al., 2016). Accordingly, there has been a plethora of marine MPs studies conducted in the Pacific Ocean, Arctic Ocean, Atlantic Ocean, and Indian Ocean (Lusher et al., 2015; Imhof et al., 2017; Desforges et al., 2014).

One study in the northern Gulf of Mexico documented 5.0–18.4 MPs/m^3 in sediment samples (Di Mauro et al., 2017). Similarly, Jang et al. (2020) reported a positive correlation between MP concentrations near densely populated coastal areas and underlying sediments. Sources of coastal polystyrene (PS) and polypropylene (PP) MPs are often derived from fishing activities from plastic polypropylene rope. Yin et al. (2020) reported MP concentrations in freshwater lakes of 180 to 693 MPs/kg of sediment in East Dongting Lake in northeastern Hunan Province, China. Similarly, a wide range of MP concentrations have been reported in beaches, coasts, and sediments of major oceans throughout the world indicating the ubiquity of MPs in marine ecosystems (Kunz et al., 2016; Young and Elliott, 2016; Guven et al., 2016; Haave et al., 2019) (Table 1).

MP ingestion or inhalation can occur from household dust (Catarino et al., 2018). Several studies have reported MPs in drinking water (Oßmann et al., 2018; Mintenig et al., 2019). Accordingly, the World Health Organization (WHO) revised safety guidelines for the presence of MPs in drinking water and the State Water Board in California, USA, has adopted requirements for four years of testing and reporting of MPs in drinking water (WHO, 2019; Coffin et al., 2022). Additionally, a systematic review by Danopoulos et al. (2020) reported that polyethylene terephthalate (PET) and PP were the most abundant MPs in drinking water samples from 12 separate studies. However, there are currently no specific safety guidelines or threshold limits for MPs in drinking water (Coffin et al., 2022). Thus, further research is required to address this uncertainty.

3 Interaction of MPs with Contaminants as Potential Vectors in Aquatic Ecosystems

MPs also pose additional toxicity to aquatic biota as they can transfer and act as vectors of sorbed organic

Table 1 Examples of occurrence and distribution of different MPs within different ecological matrices in aquatic ecosystems (ocean, sea, beach, and sediments)

Location	Ecosystem types	Abundance (mean)	Types of plastics	References
Baltic Sea	Sea, subsurface waters	0.19–7.73 items m^3	Fibers and fragments	Gewart et al. (2017)
Arctic	Surface waters	31.3 items L^{-1}	Fibers	Barrows et al. (2018)
Baltic Sea	Sea, surface waters	0.04–0.09 (0.07) items m^3	Fibers	Tamminga et al. (2018)
Arctic	Subsurface waters	0–7.5 (1.15) items m^3	Fibers	
Arctic	Surface waters	0–320,000 items km^2	Fragments	Cozar et al. (2017)
North Atlantic	Subsurface waters	0-8-5 (1.15) items m^3	Fibers	
Baltic Sea	Sea, surface waters	48 items L^{-1}	Fibers and fragments	Karlsson et al. (2017)
Arctic	Surface waters	0–1.31 (0.34) items m^3	Fragments and fibers	Lusher et al. (2015)
North Sea	Sea, surface waters	0–1.5 items m^3	Fragments	Maes et al. (2017)
North Atlantic	Deep-sea	70.8 items m^3	Fibers	Courtene-Jones et al. (2017)
Bay of Brest, North Atlantic	Sea, surface waters	2.4×10^{-4} items m^3	Fibers	Frere et al. (2017)
Hausgaten, Arctic	Offshore	42–6595 items kg^{-1} (4365)	Fragments	Bergmann et al. (2017)
North Sea	Inshore	100–3600 items kg^{-1}	Fibers	Leslie et al. (2017)
Greenland Sea, Adventfjord, Svalbard	Coastal sediment	9.3 kg^{-1}	Fibers	Sundet et al. (2015)
Baltic Sea	Beach sediment	88.1 items kg^{-1}	Fibers	Hengstmann, et al. (2018)
Norwegian Sea	Inshore	1–50 items kg^{-1}	Fragments	DNV-GL and NGI (2018)
North Sea	Offshore	0–3146 items kg^{-1}	Fibers, spheres	Maes et al. (2017)
Indian Ocean and marginal seas, ship-breaking yard, Alang-Sosiya, India	Beach	81 mg kg^{-1}	Fragments	Reddy et al. (2006)
Mediterranean Sea Izmir, Turkey	Sea	-	Pellets	Ogata et al. (2009)
Nile deep-sea fan, Mediterranean	Sea	40 items m^{-2}		Van Cauwenberghe et al. (2013)
Mexico gulf, Atlantic sea	Gulf	4.8 to 8.2 particles m^{-3}	Fibers, beads	Di Mauro et al. (2017)

and inorganic contaminants. For example, a recent review by Menéndez-Pedriza and Jaumot (2020) recognized the sorption potential of MPs to adsorb pollutants. MP characteristics (polymer type, size, and chemical properties), environmental factors (temperature, wind, and rainfall), and pollutant factors (sorption potential, molecular weight, and solubility) all contribute to sorption of pollutants by MPs (Hartmann et al., 2017; Menéndez-Pedriza and Jaumot, 2020). Organic pollutants including polycyclic aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs), and organochlorine (OCs) pollutants and metals (e.g., Cd, Ni, Cr, and Pb) are adsorbed by MPs (Godoy et al., 2019; Campanale et al., 2020a; Wang et al., 2020). Additionally, sorption and desorption of associated toxic pollutants to MP surfaces are driven by pH, temperature, dissolved organic matter, and artificial gut surfactant conditions (Bakir et al., 2014; Godoy et al., 2019). Lower pH and higher temperature were identified to increased sorption and desorption of associated toxic pollutants. Simulated equilibrium models such as Freundlich or Langmuir models to predict the mode of adsorption of hydrophobic organic pollutants on MP surfaces were reported (Bakir et al., 2014; Velez et al., 2018). Thus, sorption of hydrophobic organic contaminants (HOC) by MPs is dependent on various factors such as MP polymer size and type and surface affinity of the adsorbent and contaminants (Hartmann et al., 2017). Liu et al. (2019) reported differences in sorptive properties of hydrophilic organic contaminants in various MPs both in freshwater and marine ecosystems. Therefore, further coordination of laboratory studies with MP types and concentrations found in natural ecosystems may help elucidate MP-pollutant vector pathways and associated food web toxicity.

3.1 Vector Effects of MPs with Emerging Organic Pollutants

A plethora of research has documented interactions between toxic chemicals (e.g., organic pollutants and pesticides) with MPs (Mato et al., 2001; Rios et al., 2007; Sheavly and Register, 2007; Teuten et al., 2007, 2009; Betts, 2008; Andrady, 2011; Hirai et al., 2011; Karbalaei et al., 2021; Khoshnamvand et al., 2021). Although MPs themselves are sources of toxic chemicals (Andrady, 2011, 2017; Cole et al., 2011), MPs also sorb harmful chemicals that can be released into surrounding environments (Takada et al., 2005; Gouin et al., 2011), with some scientists calling for removal of harmful chemicals from plastic production (Dey et al., 2022). Wildlife ingestion of MPs at various concentrations may have negative effects from sorbed hazardous compounds that can be leached or desorbed from plastic fragments. When aquatic biota ingest toxic substances, they may be transferred through food chains into human food. Therefore, a major area of interest is to understand relationships between hazardous chemicals and plastic pollution. Persistent and bioaccumulation of toxic substances, such as PCBs and dichloro diphenyl trichloroethane (DDT), pose both ecological and human health risks (Engler et al., 2012).

Polybutylene terephthalates (PBTs) can enter the environment via pathways such as industrial pollutants, dispersive application of pesticides, or direct release of PCBs and polybrominated diphenyl ethers. Persistent organic pollutants (POPs) and bio-accumulative substances are comparatively well-studied, and their toxicity impacts are well understood. POPs include dichloro diphenyl dichloroethane (DDD), dichloro diphenyl dichloroethylene (DDE), DDT and its degradants, polychlorinated dibenzo-p-dioxins ("dioxins"), PCBs, and polychlorinated dibenzofurans ("furans") (Davis et al., 2019; Hoffman et al., 2019; Zhang et al., 2019a). Many are even banned in some jurisdictions. However, some POPs are still widely used, particularly in developing countries where MPs can act as vectors for POPs and other bio-accumulative toxins (Sheavly and Register, 2007; Teuten et al., 2009; Engler, 2012).

Hazardous chemicals (e.g., pesticides, trace metals, and emerging environmental pollutants) can be absorbed by MP (Besseling et al., 2017; Srain et al., 2021). MPs are considered a vector in aquatic ecosystems, accumulating pollutants such as metals and organic pollutants, resulting in increased transfer and bioavailability of hydrophobic and hazardous contaminants to aquatic species (Batel et al., 2016; Prata, 2018). When compared to seawater, contaminants can be up to 30 times more sorbtive in the gut of aquatic biota (Ma et al., 2020). Consequently, MPs play a role in the bioaccumulation and trophic transfer of environmental pollutants, as well as altered metabolic activities such as endocrine system disruption and gene expression through contaminant toxicity (Syberg et al., 2017; Zhang et al., 2019b). MP vector effects for organic pollutants and pesticides sorbed onto MP surfaces are summarized in Table 2.

3.2 Sorption Potential of MP for Metals

Accumulation of plastics in aquatic ecosystems serves as vectors collecting metal pollutants resulting in long-range transport and increased bioavailability of hydrophobic metal contaminants for aquatic species. Although accumulation of plastic debris leads to increasing levels of contaminants, the toxicity can be exacerbated in the presence of metals. For example, Kim and Chae (2017) reported that the combined effect of PS and Ni (Nickel) has an antagonistic effect on Ni toxicity. Mercury (Hg) is toxic for humans and biota. The effect of Hg and MPs may lead to higher bioaccumulation of Hg in tissues of *Daphnia magna* (Barboza et al., 2018a). Histological effects of MPs can result oxidative stress in zebrafish (e.g., Qiao et al., 2019). Similarly, exposure from metals such as Cr (chromium) and methyl mercury (Me-Hg) can cause oxidative stress, reduction of predatory activity, growth inhibition, and altered organ homeostasis (Khan et al., 2017; Luis et al., 2015; Bellingeri et al., 2019; Davarpanah and Guilhermino, 2019). However, the combined effects of these metals with MPs are poorly understood (Rainieri et al., 2018), but studies focusing on the interaction of different MP types with metals and their combined toxicity profiles within aquatic species are summarized in Table 2.

Table 2 Adsorption of organic pollutants, pesticides, and metal pollutants adsorbed on surface of MPs on aquatic species

MP types	Pollutants	Species	Toxic effect	References
PE	Polychlorinated biphenyls (PCBs)	Lugworm *Arenicola marina* (L.)	Bioaccumulation and influence feeding activity	Besseling et al. (2014)
PVC	Chiral antidepressant venlafaxine and its metabolite O-desmethylvenlafaxine (pharmaceuticals)	Loach *Misgurnus anguillicaudatus*	Accumulation in loach tissues and subcellular liver	Qu et al. (2018)
PE	Chlorpyrifos	Microalgae	Growth inhibition	Garrido et al. (2019)
PE microbeads	Triclosan (TCS) hydrophobic organic contaminants (HOC)	Marine copepod *Acartia tonsa* (Dana)	Bioaccumulation metabolic activity and mortality	Syberg et al. (2017)
PE, PS, PVC, and PVC800	Triclosan (TCS)	Microalgae *Skeletonema costatum*	Toxicity of MPs on microalgae mainly resulted from physical damage, and triclosan showed an inhibition effect on growth of microalgae	Zhu et al. (2019)
PS	Roxithromycin	*Daphnia magna*	Activate the activities of CAT and GST and MDA levels	Zhang et al. (2019a)
PE	Endocrine disruptors (PAHs, PCBs, PBDE)	Japanese medaka (*Oryzias latipes*)	Disrupted the functioning of the endocrine system, altered gene expression, and affected the proliferation of germ cells	Rochman et al. (2014)
PE, PS, and PVC	Triclosan	*Skeletonema costatum*	Growth inhibition and oxidative stress	Zhu et al. (2019)
PS	14C-phenanthrene	*Daphnia magna*	Bioaccumulation of phenanthrene-derived residues in *Daphnia* body and inhibited the dissipation and transformation of phenanthrene in the medium	Ma et al. (2016)
Fluorescent MPs	PAH benzo[a]pyrene (BaP)	*Artemia nauplii* and zebrafish	Desorbed in the intestine and transferred to intestinal epithelium and liver	Batel et al. (2016)
PE, PS	PAHs	Blue mussel *Mytilus galloprovincialis*	Accumulation in hemolymph, gills, and especially digestive tissues. Alterations of immunological responses, lysosomal compartment, peroxisomal proliferation, antioxidant system, neurotoxic effects, onset of genotoxicity	Avio et al. (2015)

Table 2 (continued)

MP types	Pollutants	Species	Toxic effect	References
PE	PAHs/PCBs/PBDEs	Japanese medaka *Oryzias latipes*	-	Rochman et al. (2014)
PE, PP	Benzo[a]pyrene (BaP)	Fish cell line	Virgin industrial MP extract has no toxic effect on fish cell line	Rochman et al. (2014)
Polystyrene (PS)	Ni	*Daphnia magna*	Abnormalities, including immobilization and changes in morphology	Kim et al. (2017)
Fluorescence red polymer microspheres	Hg	European seabass *Dicentrarchus labrax*	Accumulated metal in the brain and muscles, causing neurotoxicity, oxidative stress and damage, and changes in the activities of energy-related enzymes in juveniles of this species	Barboza et al. (2018b)
PS	Cu	Zebrafish *Danio rerio*	Accumulation in tissue and toxicity to guts and liver	Qiao et al. (2019)
PE	Ag	Rainbow trout *Oncorhynchus mykiss*	MP may facilitate the adsorption of labile contaminants and Ag desorption was evident at lower pH conditions	Khan et al. (2017)
PE fluorescent microspheres	Cr	Lima *Pomatoschistus microps*	Reduced predatory performance	Luis et al. (2015)
PS-NPs	Cu	Algae	Growth inhibition	Bellingeri et al. (2019)
Fluorescent polymer	Nano gold	*Tetraselmis chuii*	Growth inhibition	Davarpanah and Guilhermino(2019)
PE	Methyl mercury	Zebrafish	Altered organ homeostasis	Rainieri et al. (2018)
Fluorescent PE	Nano gold	Juveniles	No influence on toxicity	Ferreira et al. (2016)

PE, polyethylene; *PVC*, polyvinyl chloride; *PS*, polystyrene; *PP*, polypropylene

4 Ecotoxicity of MPs in Aquatic Food Webs

MPs are ingested by diverse vertebrate and invertebrate biota inhabiting freshwater and marine ecosystems. Ecotoxicological studies of MPs in seabirds, mammals, and other marine organisms have also been widely reported (Browne et al., 2008; Boerger et al., 2010; Fossi et al., 2014; Baak et al., 2020; Provencher et al., 2020). MP effects can be classified into two categories: primary effects (immediately after ingestion) and secondary effects (physiological and biological changes after ingested MPs enter digestive systems of aquatic biota) (Oehlmann et al., 2009). Since MPs are omnipresent in freshwater streams and seawater, buoyant floating MPs may be mistakenly ingested by aquatic fauna instead of prey or by plankton (Provencher et al., 2022). With the pervasive occurrence and similarity with plankton, MPs can easily be consumed by aquatic biota (Cole et al., 2011). For example, a total of 443 different MPs were identified in digestive organs of marine fish species in the Bay of Bengal (Hossain et al., 2019).

MP toxicity in aquatic organisms can be driven by accidental ingestion followed by physical harm leading to obstruction of intestinal tracts (Li et al., 2018; Provencher et al., 2022). Thus, ingested MPs can hinder growth and development of aquatic biota (Patra et al., 2022). For example, PS MPs resulted in substantial histomorphometrical changes in juvenile rainbow trout (Karbalaei et al., 2021). However, the combined toxicity of MPs with contaminants may be more hazardous as they can cause abnormal metabolism, accumulation of metals, and pollutants in ingested species, and may also be fatal in severe circumstances (Lga et al., 2018). Potential risk of MPs for aquatic organisms and food safety has been reported by the Food and Agriculture Organization (FAO, 2017). Food safety may be threatened through seafood consumption containing MPs and NPs (nanoplastics) (EFSA, 2016; Karbalaei et al., 2019). Furthermore, MPs may form microbial biofilms which can support harmful microbial pathogens (McCormick et al., 2016; Wright et al., 2021).

4.1 MP Toxicity in Lower Trophic Aquatic Species

Adverse impacts of MPs and NPs on phytoplankton (microalgae) have been widely documented (e.g., Khoshnamvand et al., 2021). MP deposition reduces development of microalgal tissues (Anbumani and Kakkar, 2018; Li et al., 2018; Lga et al., 2018). Positive correlations exist between MP concentrations and decreased microalgal growth (Besseling, et al., 2014). MP impacts on algal growth are lower with increased particle size (Sjollema, et al., 2016). Thus, MP toxicity is size dependent (Khoshnamvand et al., 2021). Previous studies have focused on physiological and biochemical impacts of MPs on phytoplankton (e.g., Mao et al., 2018). Studies on freshwater microalgae demonstrated that MP exposure not only poses physical harm and oxidative stress to algal cells but also influences gene expression transcriptomics engaged with certain metabolic pathways (Lagarde et al., 2016). Algae exposed to PVC microspheres resulted in decrease in chlorophyll *a* content and photosynthetic activity of *Skeletonema costatum* (Zhang et al., 2017). Co-development analysis of *Chaetoceros neogracile* and MP revealed the arrangement of hetero-aggregates comprising both algal cells and MPs because of the arrival of extracellular polysaccharides by green algal cells (Long et al., 2017).

Reactive oxygen species (ROS) can be affected in algal cells by MPs (Bhattacharya et al., 2010). MP toxicity to phytoplankton varies depending on particle size (Zhang et al., 2017) and polymer type (Lagarde et al., 2016; Mao et al., 2018). However, ecological implications and toxicity mechanisms are still poorly understood. In an ecotoxicological study, algae exposed to polystyrene MPs for 72 h resulted in decreased photosynthetic activity in *Chlorella vulgaris*, *Thalassiosira pseudonana*, and *Dunaliella tertiolecta* (Sjollema et al., 2016). The degree of MP toxicity is relative to particle size and absorption potential at the primary producer level (Sjollema et al., 2016). Similarly, micro-sized PS particles (0.22 and 103 mg/L) influenced algal development and diminished chlorophyll *a* content prompting reduction of photosynthesis in *Scenedesmus obliquus* (Besseling et al., 2013).

In the marine environment, MP impacts have been observed with exposure of 1-μm PVC particles on *Skeletonema costatum*. PVC MPs <1 μm decreased development rate by 39.7% after a 96-h incubation, while larger PVC MPs (>1 mm) had no harmful impact (Zhang et al., 2017). No reduced development rate was observed in *Tetraselmis chuii* after exposure to fluorescent red polyethylene microspheres (Davarpanah and Guilhermino, 2015). An expanded uptake assay using 10 μm and fluorescent polystyrene particles (1–5 µm) was performed with algal species *Oxyrrhis marina* and *Rhodomonas baltica* which resulted in the loss of juvenility and substitution of food (Lyakurw, 2017). PS MP–derived aquatic toxicity may cause biofouling development and shedding of aquatic phytoplankton that can result in reduced chlorophyll production in marine autotrophs (Long et al., 2015). Therefore, toxicity of MPs on aquatic phytoplankton, zooplankton, and other aquatic organisms may vary based on MP particle size (Patra et al., 2022). Thus, MP ecotoxicity in aquatic ecosystems remains a critical research gap. Figure 2 shows hypothetical interactions of MPs with multiple species throughout aquatic food webs.

4.2 Impact of MPs on Other Aquatic Consumers

Incidental ingestion of MPs by the other higher trophic aquatic consumers (e.g., zooplankton, shrimp, and fish species) has been reported in the South China Sea and Bay of Bengal (Sun et al., 2017; Hossain et al., 2020). Table 3 shows examples of incidental MP ingestion included translocation and resulting adverse impacts on the aquatic food chain may lead to complex ecotoxicology in aquatic and marine biota. Although ecotoxicology of MPs to higher trophic consumers is limited, in vitro studies have been conducted on aquatic organisms representing lower trophic levels. For example, Li et al. (2018) and Murphy and Quinn (2018) studied scavengers (e.g., *Hydra attenuata*) and cnidarians. Uptake of MPs has been reported in freshwater consumers such as *Daphnia magna*, *Gammarus pulex*, *Notodromas monacha*, and *Potamopyrgus antipodarum* under field conditions (Imhof et al., 2013). Ingestion of polyethylene MPs (<400 µm) in the freshwater sponge, *Hydra attenuata*, and feeding frequency were documented by Murphy and Quinn (2018). MPs between 0.01 and 1 mm were found in gut epithelia of *Daphnia magna* and were deposited in lipid storage droplets, disrupting filtration in this crustacean (Rosenkranz et al., 2009). DNA damage was also observed in rainbow trout that ingested MPs in a review by Alimba and Faggio (2019). The combined biochemical and additive impacts of MPs and NPs with hydrophobic phenanthrene have been documented in *Daphnia magna* (Ma et al., 2016). A review by Alimba and Faggio (2019) reported increased damage and mortality after MP exposure for long periods in seals, dolphins, and sea snakes. Bioaccumulation of MPs in the digestive tract of zebrafish (*Danio rerio*) and nematode (*Caenorhabditis elegans*) along with physiological impacts was reported by Lei et al. (2018). Polystyrene MPs (5 and 70 nm) were also found in gills, liver, and gut of zebrafish (*Danio rerio*) by Lu et al. (2016).

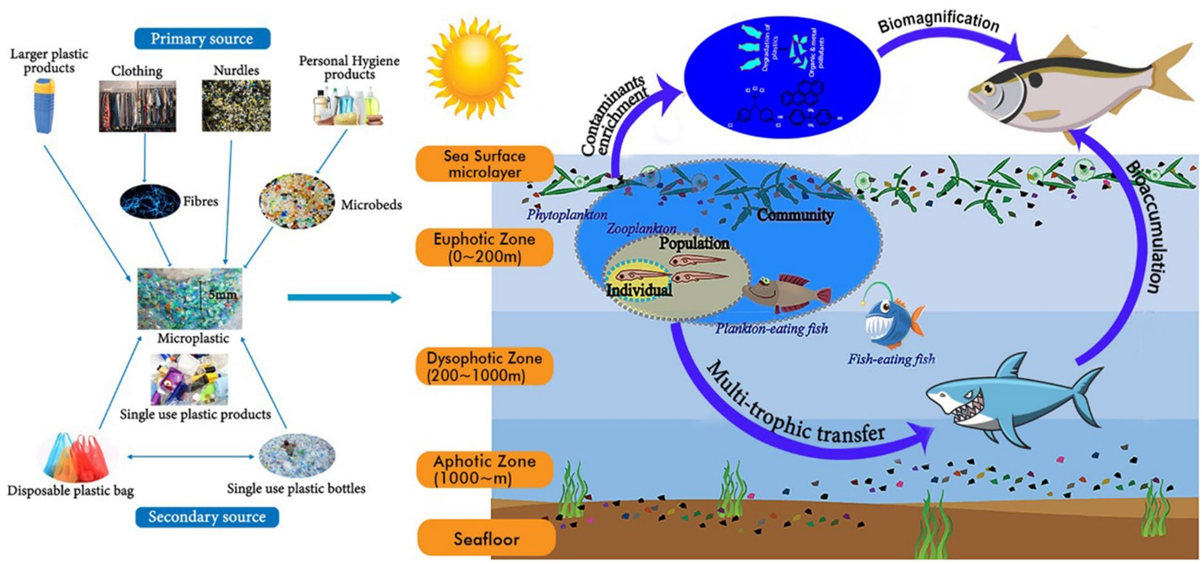

Fig. 2 Hypothetical interactions of MPs with diverse lower trophic (aquatic producers) and higher trophic (aquatic consumers) including food web transfer

Table 3 Ingestion, translocation, and impact of MPs on marine organisms

Species name	Size of ingested material	Exposure concentration	Effects	References
Scenedesmus spp.	130 µm Unplasticized polyvinylchloride (UPVC)	0–5% by weight	Ingestion decreased feeding, phagocytic activity upregulation, reduced available energy reserves, lower lipid reserves	Wright et al. (2013)
Isochrysis galbana, *Chaetoceros neogracilis*, *Rhodomonas salina*, *Heterocapsa triquetra*	2 µm PS	104 mL^{-1}	Attachment of microspheres to algae; no negative effect observed	Long et al. (2014)
Strombidium sulcatum	0.41–10 µm	5–10% ambient bacteria concentration	Ingestion	Christaki et al. (1998)
Tintinnopsis lobiancoi, *Synchaeta* spp. *Marenzelleria* spp. *Bosmina coregoni*	10 µm PS	1000, 2000, 10,000 mL^{-1}	Ingestion	Setälä et al. (2014)
Arenicola marina	20-2000 µm	1.5 g L^{-1}	Ingestion	Thompson et al. (2004)
Arenicola marina	130 µm UPVC	0–5% by weight	Ingestion decreased feeding, increased phagocytic activity reduced available energy reserves, lower lipid reserves	Wright et al. (2013)
Arenicola marina	230 µm PVC	1500 g of sediment mixture	Ingestion, oxidative stress	Browne et al. (2013)
Arenicola marina	400–1300 µm	0, 1, 10, 100 g L^{-1}	Ingestion, reduced feeding, weight loss	Besseling et al. (2013)
Galeolaria caespitosa	3–10 µm	5 microspheres μL^{-1}	Ingestion	Bolton and Havenhand (1998)
Galeolaria caespitosa	3 and 10 µm PS	635, 2240, 3000 beads mL^{-1}	Ingestion, size selection, egestion	Cole et al. (2013)
Mytilus edulis	30 nm PS	0, 0.1, 0.2, and 0.3 g L^{-1}	Ingestion, pseudofeces, reduced filtering	Wegner et al. (2012)
Mytilus edulis	0–80 µm HDPE	2.5 g L^{-1}	Ingestion, retention in the digestive tract, transferred to lymph system, immune response	von Moos et al. (2012) Köhler (2010)
Mytilus edulis	3, 9.6 µm	0.51 g L^{-1}	Ingestion, retention in the digestive tract, transferred to lymph system	Browne et al. (2013)
Placopecten magellanicus	15, 10, 16, 18, 20 µm PS	1.05 mL^{-1}	Ingestion, retention, egestion	Brillant and MacDonald, (2002)
Crassostrea virginica	10 µm PS	1,000 mL^{-1}	Ingestion, egestion	Ward et al. (2003)
Daphnia magna	Polycarbonate (PC), PVC, PU, PE, LDPE, PMMA, PET, HDPE, PTFE, acrylonitrile butadiene styrene (ABS), PP, MDPE	24 and 48 h 70e100 g L^{-1}	Mortality	Lithner et al. (2009)

Table 3 (continued)

Species name	Size of ingested material	Exposure concentration	Effects	References
Nephrops norvegicus	5 mm PP fibers	10 fibers per 1 cm^3 fish	Ingestion	Murray and Cowie (2011)
Nephrops norvegicus	500–600 µm PE loaded with 10 µg of PCBs	150 mg microplastics in gelatin food	Ingestion, 100% egestion Increased PCB levels in tissues. Same increase for positive control. No direct effect of MPs	Devriese et al. (2014)
Pomatoschistus microps	1–5 µm PE	18.4, 184 µg L^{-1}	Ingestion, modulation bioavailability or biotransformation of pyrene, decreased energy, inhibited AChE activity	Oliveira et al. (2013)
Oryzias latipes	PE pellets	Two months chronic exposure	Altered gene expression, reduced choriogenin regulation in males and decreased vitellogenin and choriogenin in females	Rochman et al. (2014)
Dicentrarchus labrax	10–45 µm PE	0–105 g^{-1} incorporated with food	Ingestion, no significant increase in growth of effect on survival of larvae Possible gastric obstruction	Oliveira et al. (2013)
Oryzias latipes	3 mm LDPE	Ground-up as 10% of diet	Liver toxicity, pathology, hepatic stress	Rochman et al. (2013)

ABS, acrylonitrile butadiene styrene; *PC*, polycarbonate; *PE*, polyethylene; *LDPE*, low-density polyethylene; *MDPE*, medium-density polyethylene; *HDPE*, high-density polyethylene; *PET*, polyethylene terephthalate; *PMMA*, poly(methyl methacrylate); *PP*, polypropylene; *PTFE*, polytetrafluoroethylene; *PS*, polystyrene; *PVC*, polyvinyl chloride; *PU*, polyurethane

Many studies have reported on the occurrence of MPs in the gastrointestinal tracts and of negative impacts of MP ingestion by pelagic fish species (e.g., Jovanovic, 2017). For example, Nadal et al. (2016) found MPs of various sizes in gastrointestinal tracts of *Boops boops* from the Mediterranean Sea. Karbalaei et al. (2019) reported finding MPs (mean = 2600 μm ±7.0 SD) in nine species of commercial fish from Mayalsia, including *Eleutheronema tridactylum* and *Clarias gariepinus.* Uptake of MPs consisting of blue nylon fragments in three species of *Gerreidae* fish (e.g., *Eugerres brasilianus*, *Eucinostomus melanopterus*, and *Diapterus rhombeus*) from a tropical estuary in Northeast Brazil was documented by Ramos et al. (2012). For example, between 4.9 and 33.4% of fish sampled contained blue nylon MP fragments (Ramos et al., 2012). Another study reported that 12% of *Gobio gobio* from French streams had ingested MPs (Sanchez et al., 2014). Visual sorting of digestive tracts showed higher MP concentrations in *Gobio gobio* from urban streams compared to rural streams. While most studies have focused on MP prevalence in gastrointestinal tracts of fish, few have investigated the potential for MP transfer to other organs (e.g., liver).

MP uptake, transfer, and deposition observed in the filter feeder (*Mytilus edulis*) were reported by Van moos et al. (2012), along with ecotoxicological effects including inflammatory cell response and destabilization of lysosome membranes. Harmful impacts of MPs on cells of *Mytilus galloprovincialis* were observed by Paul-Pont et al. (2016), including modifications of immunological reactions, lysosome alteration, peroxisomal expansion, irritation of antioxidant system, and genetic toxicity.

4.3 MP Toxicity and Potential Trophic Transfer Across Food Webs

Bioaccumulation of MPs along with toxic organic pollutants has previously been documented by Rochman et al. (2013). For example, trophic transfer of bio-accumulated MP may cause food web toxicity including human health hazards (Wright et al., 2013). The biochemical mechanism of MP toxicity may include alteration of lipid metabolism, interaction of ROS in the bloodstream, formation of gut barriers, and dysbiosis of gut-inhabiting microbiota (e.g., Deng et al., 2017; Lu et al., 2018; Jin et al., 2019). Recent observations by Jin et al. (2020) revealed PS-MP toxicity in male mammal reproductive systems. For example, adsorption and translocation of 4–10-μm–sized PS-MP particles within testicular and spermatogenic cells caused a drop of testosterone and dysfunctions of the male reproduction system through reduction of sperm quality in male mice (Jin et al., 2020). Similarly, dietary exposure of PS-MPs was evaluated for toxicity in Japanese medaka (*Oryzias latipes*) by Zhu et al. (2020). Although low doses did not show potential toxicity, higher doses caused alteration of spleen, kidney, thickening of epithelium tissues, decrease in female fecundities, and abnormal mucosal secretions in *Oryzias latipes* (Zhu et al., 2020). Thus, there is a critical need to better understand the function of MPs in bioaccumulation of plastic-related contaminants and associated biological toxicity within aquatic ecosystems at environmentally relevant concentrations. Table 4 summarizes ecotoxicology assessment and mechanisms of MP toxicity.

Trophic transfer of microfiber MPs in indoor and outdoor environments may affect human food webs and can be deposited and translocated into the human tissues (Dris et al., 2017). Lung aggravation and severe breathing impairment can be prompted by microfiber exposure. For example, genetic toxicity and lethal malignant diseases may be linked to chronic exposure of human microfibers (Gasperi et al., 2018). The generation of ROS has been documented as a cause of deteriorating health conditions due to accidental microfiber inhalation (e.g., Wang et al., 2019).

MP-related food web toxicity in the aquatic food chain is likely influenced by prey-predator relationships (Dris et al., 2017). Accidental ingestion of MPs by predatory aquatic species has been identified as the primary entry point for MP trophic transfer. For example, prey-predator trophic transfer of MPs in the Antarctic has been documented with MP bioaccumulation in fur seals, *Arctocephalus tropicalis* and *Arctocephalus gazella*, after consumption of pelagic fish *Electrona subaspera* containing MPs (Eriksson and Burton, 2003). Additionally, MPs were reported in Gentoo penguin scats collected from remote sub-Antarctic and Antarctic islands providing further evidence of MP transfer through remote marine food webs (Bessa et al., 2019). Trophic transfer of MPs through other aquatic species such as mussels (*Mytilus edulis*), *Mysis mixta*, and crustaceans *Daphnia magna* has previously been reported (Farrell and

Table 4 Ecotoxicological profiles of MPs on multiple aquatic species and mechanism of toxicity

Studies sample/species	Eco-toxicity assessment	Reference
Chaetoceros neogracile	Decreased in cellular esterase activity and neutral lipid content	Seoane et al. (2019)
Chlorella vulgaris	Inhibitory influence on cell photosynthesis	Luo et al. (2019)
Daphnia magna	Activities of CAT, GST, and MDA levels	Zhang et al. (2019a)
Seals, sea lions, dolphins, and sea snake	Increase in morbidity and mortality	Review by Alimba and Faggio (2019)
Myriophyllum spicatum	Inhibited shoot growth	van Weert et al. (2019)
Paracentrotus lividus	Decrease of larval length	Oliviero et al. (2019)
Chlorella pyrenoidosa	Growth inhibition and reduced photosynthetic activity	Mao et al. (2018)
Skeletonema costatum	Growth inhibition and photosynthesis inhibition	Zhang et al. (2017)
Sparus aurata, *Dicentrarchus labrax*	Affect the cell viability	Espinosa et al. (2018)
Lemna minor	Inhibited root growth	Kalcikova et al. (2017)
Danio rerio	Inflammation and lipid accumulation in fish liver	Lu et al. (2016)
Chlorella pyrenoidosa and *Microcystis*	Inhibited growth.	Wu et al. (2019)
Scleractinian coral	Repress detoxification and immune system	Tang et al. (2018)
Lemna minor	Inhibited root growth	Kalcikova et al. (2017)
Carassius auratus	Accumulation in gut	Grigorakis et al. (2017)
Carcinus maenas	Adsorbed on gill surface	Watts et al. (2014)
Hyalella azteca	Decreased growth and reproduction	Au et al. (2015)
Mytilus edulis L.	Histological changes and inflammatory response	von Moos et al. (2012)
Dunaliella tertiolecta *Thalassiosira pseudonana* *Chlorella vulgaris*	Growth negatively affected by uncharged particles	Sjollema et al. (2016)
Arenicola marina	Higher susceptibility to oxidative stress	Browne et al. (2013)
Oncorhynchus mykiss	DNA damage	Pannetier et al. (2019)
Arenicola marina	Long residence time in gut, inflammation, feeding apparatus affected	Wright et al., (2013)
Arenicola marina	Accumulation of PCBs in tissue	Besseling et al. (2013)
Ostrea edulis, *Mytilus edulis*	Reduced filtration rates	Green et al. (2017)
Scrobicularia plana	Neurotoxicity, mechanical injury to gills, MPs-adsorbed BaP and PFOS exerting a negative influence over assessed biomarkers in this tissue	O'Donovan et al. (2018)
Carassius carassius, *Alburnus alburnous*, *Scardinius erythrophthalmus*, *Tinca tinca*, *Esox esox*, *Salmo salar*	Triglycerides: cholesterol ratio in blood serum, nanoparticles bind to apolipoprotein A-I in fish serum in vitro, restraining them from properly utilizing their fat reserves	Cedervall et al. (2012)
Daphnia magna, *Carassius carassius*	Direct interactions between plastic nanoparticles and brain tissue of fish	Mattsson et al. (2017)
Dunaliella tertiolecta	Significant inhibition on the algal growth with no effects on photosynthesis	Sjollema et al. (2016)
Tisochrysis lutea	Formation of hetero-aggregation for C. *neogracile*	Long et al. (2017)
Skeletonema costatum	Inhibition on algal growth; chlorophyll content and photosynthesis	Zhang et al. (2017)

Table 4 (continued)

Studies sample/species	Eco-toxicity assessment	Reference
Chlorella pyrenoidosa	Inhibition on algal growth from lag to earlier logarithmic phases	Mao et al. (2018)
Chlamydomas reinhardtii	No significant influence on growth and expression of genes involved in stress response. Enhanced expression of genes involved in sugar biosynthesis pathways	Lagarde et al. (2016)
Scenedesmus obliquus	Inhibition on algal growth. Reduction in chlorophyll-a content	Besseling et al. (2014)
Pseudokirchneriella subcapitata	Inhibition on algal growth	Casado et al. (2013)
Centropages typicus	Significant reduction in algal ingestion	Cole et al. (2013)
Nephrops norvegicus	Reduced feeding rate, body mass, metabolic rate, and catabolism of stored lipids	Welden and Cowie. (2016)
Mytilus edulis	Increase in energy consumption	Van Cauwenberghe et al. (2015)
Sparus aurata; *Dicentrarchus labrax*	No effects on cell viability of head-kidney leucocytes. Negligible effects on main cellular innate immune activities. No effects on expression of genes related to immunity, oxidative stress, cellular protection to pollutants, and cell death	Espinosa et al. (2018)
Danio rerio	No or low lethality. Intestinal damage including cracking of villi and splitting of enterocytes	Lei et al. (2018)

Nelson, 2013; Wang et al., 2019). Figure 2 shows potential translocation of MPs in aquatic ecosystems along with multi-trophic transfer focusing on bioaccumulation.

4.4 MP Toxicity and Human Health Implications

Several studies have highlighted the pervasive role of MPs in atmospheric aerosol and have raised concerns about atmospheric MP and NP contamination and human health impacts (see Sharma and Chatterjee, 2017; Prata, 2018; Revel et al., 2018; Rist et al., 2018; Lehner et al., 2019; Allen et al., 2020; Campanale et al., 2020b). MPs and NPs can enter the human body through ingestion of contaminated foods (Rakib et al., 2021b; Wright and Kelly, 2017; Silva-Cavalcanti, 2017; Waring et al., 2018). It has been predicted that MPs present in marine organisms may cause human health impacts through consumption (Karbalaei et al., 2018, 2019; Waring et al., 2018). Previous studies have reported MP deposition through dietary intake in the human body, and particles <150 μm can cause systemic exposure by crossing the gastrointestinal epithelium (Huang et al., 2020; Zhang et al., 2020; Domenech and Marcos, 2021). It is believed that a very small amount (0.3%) is adsorbed, and a lower fraction (0.1%) of particles composed of 10-μm particle size can penetrate different organs and cellular membranes including the placenta and blood-brain barrier (Cole et al., 2011; Barboza et al., 2018b). However, there is uncertainty about MP toxicity in human organs because of technical and analytical difficulties in extracting, characterizing, and quantifying them from environmental sources (Campanale et al., 2020a, 2020b). Importantly, MPs consisting of particles <2.5 μm can enter the circulatory system after adsorption through the gastrointestinal tract (Wright and Kelly, 2017). As a result, MPs <2.5 μm can cause inflammation and dose-dependent accumulation in the human body. MPs can also enter the human body through direct inhalation (Prata, 2018; Vianello et al., 2019). Airborne MPs from atmospheric contaminants, erosion of agricultural lands, wastewater treatment products, synthetic garments, road-dust, industrial pollution, and marine aerosols may cause breathing distress, inflammatory, cytotoxic, and autoimmune disorders in human health (Barboza et al., 2018a; Prata, 2018).

Human lungs have a relatively large alveolation surface composed of a very thin tissue barrier (<1 μm) allowing penetrating nanoparticles into blood vessels and the entire human body (Wright and Kelly, 2017). Depending on individual metabolism and sensitivity, MPs can trigger asthma-like symptoms, fiber-inclusive granulomas interstitial fibrosis, chronic bronchitis, and pneumothorax, with characteristics such as chemical composition, size, shape, and hydrophobicity governing the cytotoxicity of MPs to cells and tissues (Prata, 2018). Human skin may also be a route of access for MPs (<100 nm) exposure to the human body via use of nano-scrubs and cosmetics (Revel et al., 2018).

There is limited data about particle size, shape, and chemical composition of MPs in facial scrubs, soap, and human food. The German Federal Institute for Risk Assessment reported that the particle size of MP is >1 μm in shower products and face pack fillings. The repeated use of these may cause the accumulation of PP, and PE MPs in tissues and cells that eventually harm the skin (Sharma and Chatterjee, 2017). Humans are susceptible to MPs and associated contaminants through dietary intake of MPs in seafood which presents a major threat to food safety (Onyena et al., 2021; Van Cauwenberghe and Janssen, 2014).

Therefore, to assess causative effects of contaminated marine food on public health, more studies and evaluation of possible health hazards of MPs consumption from a variety of foods must be conducted. For example, an in vitro study by Deng et al. (2017) reported toxic effects of MPs on human health. This study determined the oxidative stress by ROS, following exposure to MPs in cerebral and epithelial cells. Using mice, the distribution, accumulation, and tissue-specific health risks were observed (Deng et al., 2017). For example, intake of 0.5 mg/day of PS-MPs resulted in accumulation in the liver, kidney, and intestine. Therefore, kinetics of tissue accumulation and pattern of distribution were severely affected by MP toxicity. Moreover, analysis of biochemical characteristics and metabolic profile in mice liver stated that MPs were responsible for oxidative stress, neurotoxic effect, energy, and lipid metabolism (Revel et al., 2018). These findings indicate cellular toxicity of liver cells in humans. MPs in the lumen area of the intestine can also influence adsorption of fluids. Large proteins can interact with MPs and alter the immune system, resulting in local inflammation (Powell et al., 2007). There is scarce information on the toxic effects of NPs on humans. The intake of plastic nanoparticles may cause inflammation and cell death in the liver (Khlebtsov and Dykmana, 2011). Likewise, several recent studies have reported the adverse metabolic and cellular complexities in human health through NP toxicity (Shen et al., 2019; Chang et al., 2020; Yee et al., 2021; Walker et al., 2022). Furthermore, microbial biofilms colonizing MP surfaces in the environment—termed the "Plastisphere"—can comprise pathogenic microbial communities (McCormick et al., 2014; Wright et al., 2021). Figure 3 shows a schematic of postulated human health implications of MP toxicity.

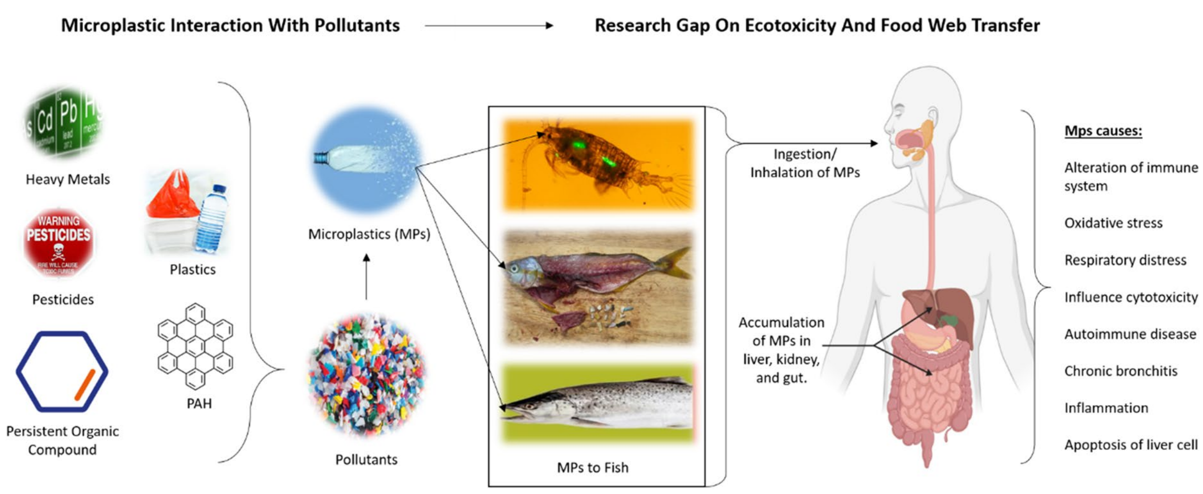

Fig. 3 Schematic interactions of toxic pollutants with MPs and toxicity profiles in ingested species and human food chain focusing on contemporary research gaps

MPs pose threats to food webs through trophic transfer in aquatic ecosystems, although uncertainty and current research gaps on MP translocation through food webs to humans along with exposure risks and toxicity exist. Several studies reported limited data and accuracy in gut barrier, tissue metabolism, and alteration of gut microbiota diversity in mice due to MP toxicity. Additionally, exploration of the complex mechanism of ecotoxicological profiles, mediated by MPs, is poorly understood, and requires more research. MP microfibers and MPs derived from seafood also pose potential threats to human health. However, there remains uncertainty for ecotoxicological profiles of MPs with respect to potential entry routes (e.g., via aquatic food webs or atmospheric microfibers) and subsequent human health implications.

5 Conclusions

This non-exhaustive review has highlighted many studies that have identified MP pollution in aquatic environments as a research priority. Although some studies reported in this review describe the fate and toxicological effects of MPs in laboratory or ad hoc regional case studies, detailed studies of fate, transport, and toxicological effects of MPs through global environments are yet to be fully explored. Even after documenting a plethora of studies in this review, there remains research uncertainties concerning the toxicity of MPs in aquatic species, interactions of MPs with hazardous pollutants, and toxicity effect mechanisms.

This review has demonstrated that MPs can act as vectors for toxins such as POPs, pesticides, metals, PAHs, and other organic pollutants. However, there is still a lack of scientific understanding on the sorption potential (either hydrophilic or hydrophobic) of MPs which may arise due to diverse chemical properties and size of MPs. In fact, many of the thousands of chemicals used in plastic production are known to have toxic effects on organisms and humans. Thus, the chemical and physical association (e.g., adsorption or desorption) of pollutants with MPs and the impact of polymer size and type should be investigated in greater detail. Future studies should focus on the sorption potential of the diverse range of organic and inorganic pollutants that now exist in the environment. This review has shown that the toxic effects of organic and inorganic pollutants sorbed onto MPs are complex even when contaminants are studied in isolation or when combined with other contaminants in the environment (e.g., synergistic effects). This is only exacerbated in the environment, so it is suggested that further studies are required on complex MP polymer types and sizes and in combinations with suite of different contaminants at environmentally relevant concentrations. Thus, this review has shown that there is a need to expand existing information regarding food web toxicity of MPs and assessing the role of MPs as "vectors" to transport contaminants, such as POPs, metals, hydrophobic/hydrophilic contaminants, pesticides, and their potential impact on food safety.

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Declarations

Conflict of interest The authors declare no competing interests.

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