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Impact of microplastic pollution on the ocean and marine animals: A comprehensive review

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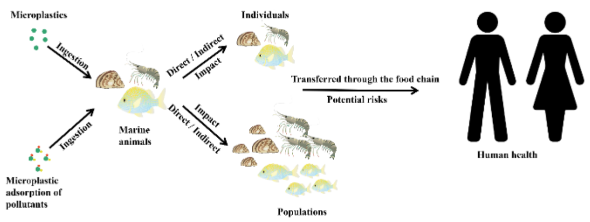
Graphical abstract



Abstract

This study aims to review the effects of microplastics (MPs) with and without adsorbed organic pollutants or heavy metals on marine animals and the associated risks. First, the sources, composition, migration, and distribution of MPs in the marine environment is presented. Second, the effects of the abovementioned MPs were summarized, revealing that MPs alone affect the behavior and physiology of marine animals, whereas MPs adsorbed with pollutants exhibit either synergistic or antagonistic effects (increasing or decreasing bioaccumulation and toxicity, respectively). The impacts pose risks to marine animals at both the individual and population levels. Moreover, the migration of MPs within the food chain introduces additional threats to marine ecosystems. Meanwhile, by consuming contaminated marine animals, MPs can enter and accumulate in the human body, damaging human health. This review provides a reference for similar studies investigating the marine environment and a scientific basis for the protection and management of marine ecosystems.

Keywords: microplastics, organic pollutant, heavy metals, marine animals, marine environment, risks

1. Preface

1.1. Introduction

Concerns regarding plastic pollution first emerged in the ocean. Specifically, Carpenter and Smith (1972) reported the widespread presence of granular plastic debris measuring approximately 0.25–0.5 cm in diameter in the western Sargasso Sea; however, this finding did not attract the attention of researchers at the time. Moore *et al.* (2001) reported that the mass of plastic litter in the North Pacific was six times higher than that of zooplankton. Thompson *et al.* (2004) first introduced the concept of “microplastics (MPs),” defining it as plastic debris sized <5 mm. With MPs being increasingly discovered, people are becoming aware of the harmful effects of MPs on the environment. Rands *et al.* (2010) reported that MPs are widespread in the marine environment and may cause harm to marine life. The Yearbook of Emerging Issues in the Global Environment published by the United Nations Environmental Programme (UNEP) listed plastic litter in the oceans as one of the top 10 emerging environmental issues worldwide (UNEP, 2014). Since then, marine MPs have attracted widespread attention worldwide, and research on this topic has entered a phase of rapid development.

Plastic are widely and globally used because of their excellent properties, such as low cost, durability, light weight, and ductility (Jacques and Prosser, 2021). Since the use of plastic became widespread, the exponential growth of plastic production has led to an increase in the amount of waste entering aquatic ecosystems. More than 300 million tons of plastics are currently produced annually, which far exceeds the production of only 1.5 million tons of plastic in 1950 (Boucher and Friot, 2017). The UNEP reported that MPs, which account for at least 85% of the total marine litter, are a growing threat to ocean ecosystems. Without effective interventions, the amount of plastic waste entering aquatic ecosystems is expected to nearly triple from 23 to 37 million tons per year by 2040 (UNEP, 2021). To understand the impact of MPs on marine animals and the associated risks, it is necessary to first explore their source, abundance, composition, migration, and distribution statuses in the marine environment.

The widespread distribution of plastic debris in marine ecosystems, including beaches, surface waters, deep-sea sediments (Moore *et al.* 2001; Thompson, 2004), and even remote environments (e.g., Equatorial Western Atlantic) (Ivar *et al.* 2009), poses a significant threat to marine animals, humans, the environment, and the economy (Napper and Thompson, 2019). Plastic debris can injure or kill marine macrofauna by entangling or clogging their intestines (Laist *et al.* 1997). Additionally, plastic litter fragments into small particles that are easily ingested by marine animals at different trophic levels (Guzzetti *et al.*

2018). The entry of MPs into their bodies can affect their growth and development, reproductive capacity, behavioral characteristics, as well as other aspects (Wang *et al.* 2020b). Furthermore, marine animals may suffer pollutant-related damages caused by the enrichment of MPs with toxic additives and the accumulation of persistent organic pollutants and heavy metals in the surrounding environment (Fred-Ahmadu *et al.* 2020). This enrichment may be transferred along the food chain, causing harmful effects on marine organisms as well as human.

Most of the previously published reviews have summarized the toxicological studies of MPs for a single trophic level or a single species. This review summarizes more comprehensively the effects of MPs with and without adsorbed contaminants on a wide range of marine animals. On this basis, the risks posed by MPs to marine animals, marine ecosystems, and even human health are discussed, and the remaining shortcomings in current research are prospected. Hopefully, this review will benefit future investigations in related fields.

1.2. Data sources

The Elsevier and Web of science databases were employed to search the literature for this review. Relevant information was searched individually or jointly using keywords such as “microplastics,” “marine microplastics,” “marine animals,” “sorption,” and other related phrases. As of 2023, upon searching the keyword “microplastics,” the number of articles retrieved from Elsevier and Web of science was 8340 and 12,822, respectively. When the keyword “marine” was added, 2514 and 6891 articles were retrieved, respectively. When the last keyword “animal” was included, the relevant publications were 179 and 480 in number, respectively. The literature was screened based on the abovementioned searches, and the articles published between 2008 and 2023 (up to January) were assimilated and analyzed.

2. MPs in the marine environment

2.1. Sources

Based on their sources, MPs in the environment can be divided into two categories: primary and secondary. Primary MPs refer to tiny plastic particles that are already present in nature (Cole *et al.* 2011) and are predominant in the marine environment, accounting for approximately 15%–31% of the total marine plastics (Boucher and Friot, 2017). Secondary MPs are fragments that are produced after the gradual breaking of larger plastic items present in the environment via physical action, chemical action, and biodegradation (Cole *et al.* 2011).

The ocean receives a large amount of MPs from a wide range of sources (Figure 1), which can be divided into two categories: terrestrial and marine. Terrestrial sources include wastewater and waste derived from daily activities (Cheung and Fok, 2017), agriculture (Steinmetz *et al.* 2016), industry (Alimi *et al.* 2018), and other activities, which account for ~80% of marine plastic litter and are the main pathways of the entry of MPs into the marine environment (Andrady, 2011). The main types of discharged items are bags, garbage bags, footwear, and household products (Barnes *et al.* 2009). The main inputs derived from marine sources are aquaculture (Sui *et al.* 2020), coastal tourism (Chen *et al.* 2020a), and ship transport (Chen *et al.* 2021). In addition, atmospheric wet deposition is an important source of marine MPs (Long *et al.* 2022). A modeling study estimated that 30% of the MPs produced by the global road traffic are transported by the atmosphere to the oceans (Evangeliou *et al.* 2020). A large number of complex MPs compounds have been detected in the marine environment. Among them, polypropylene (PP), low-density polyethylene (LDPE), high-density polyethylene (HDPE), polyvinyl chloride (PVC), polystyrene (PS), and nylon (PA) represent ~24%, 21%, 17%, 19%, and 7% of the total marine MPs, respectively (Andrady, 2011).

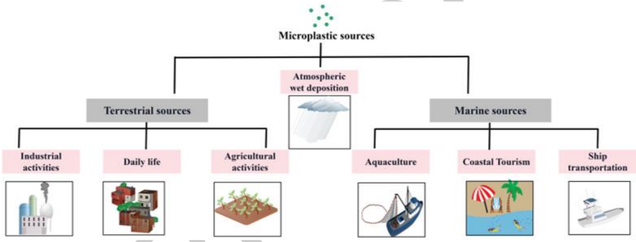


Figure 1. Sources of marine MPs

2.2. Migration

The migration of MPs in the marine environment can be divided into two types: one is the migration between environmental media, where fragments are transferred under the influence of external conditions, such as currents, wind, and gravity; the other is the migration between marine food chains, where MPs are taken up and absorbed by organisms at lower trophic levels and then enter the food chain and migrate toward higher levels.

2.2.1. Migration of MPs between environmental media

MPs present different environmental migration behaviors between different environmental media (Figure 2). Particles derived from terrestrial sources can enter the ocean through point sources (e.g., sewage treatment plants and sewer overflows) or diffusion pathways (e.g., rainwater, surface runoff, and wind) (Siegfried *et al.* 2017). For example, MPs derived from personal care products (e.g., toothpaste, scrubs, and makeup), plastic materials from households as well as laundry wear and tire wear particles can enter the marine environment through rivers (Sundt *et al.* 2014), which are therefore considered as carriers allowing the migration of MPs from terrestrial sources to the ocean.

In addition, the migration of MPs is affected by currents and meteorological factors as particles enter the ocean. Under the influence of currents and wind, MPs can be rapidly and widely dispersed and transported over long distances from their sources (Van *et al.* 2012). Kim *et al.* (2015) investigated the factors affecting the spatial variation of MPs on the high-tidal coastal beaches of Korea and reported that monsoons and currents considerably impacted particle size distribution and spatial homogeneity. Moreover, owing to the influence of rainfall and the northeast monsoon, the abundance of MPs was substantially higher on the western coast than on the eastern coast of Hong Kong (Cheung *et al.* 2016). Another mode of MPs migration is sea ice, which serves as a transport vehicle. The formation of sea ice can capture microplastic particles in the water column and carry them over long distances as it drifts away (Kanhai *et al.* 2020). Additionally, some MPs floating in offshore areas can be transported from the ocean to tidal flats via tidal action (Iwasaki *et al.* 2017).

External factors (e.g., wind (Kim *et al.* 2015), rainfall (Cheung *et al.* 2016), and flow velocity (Breivik *et al.* 2011)) can affect the migration of MPs. Specific characteristics of MPs, such as density and size, also affect the migration process. Reportedly, during atmospheric transport, smaller the particle size, easier is the transport via wind; conversely, MPs with diameters >1000µm usually cannot be easily transported over long distances via wind (Bullard *et al.* 2021). Medium-sized MPs are more likely to migrate from freshwater environments to the ocean than larger MPs (Moore, 2008). Another point of interest is that particles can migrate vertically depending on their characteristics. Those with higher density can directly reach the deep sea or seafloor, while those with lower density (lower density than seawater) can produce polymerization with microorganisms, increase their specific gravity, sinking slowly in the water column and finally depositing on the sea floor (Li *et al.* 2017). Moreover, biofouling and the adsorption of other contaminants onto microplastic particles increase their density, thereby facilitating their settling (Kim *et al.* 2009).

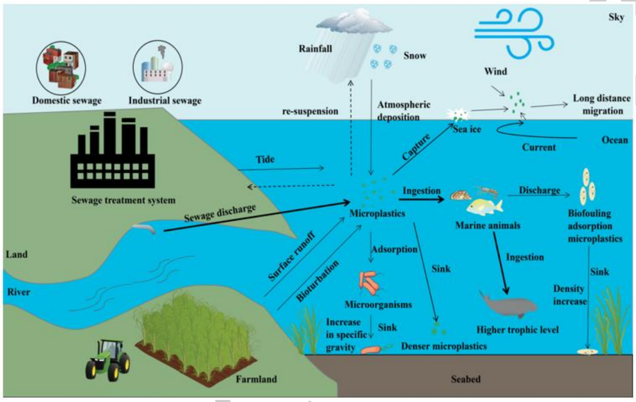


Figure 2. Migration of MPs between environmental media; bolded arrows indicate the focus on the transport of MPs in environmental media and the food chain

2.2.2. Migration of MPs through food chains

As MPs are similar in size to plankton, marine animals have difficulty in distinguishing them from food items during feeding and can easily ingest them by mistake (Moore *et al.* 2001; Moore, 2008). Moreover, some marine animals can incorporate MPs into their bodies through respiration (e.g., crabs inhale MPs through their gill aperture (Watts *et al.* 2014)). As a result, MPs in the ocean can easily enter the food chain.

After entering the food chain at lower trophic levels, animals at higher trophic levels ingest MPs indirectly while feeding on contaminated organisms, resulting in the transport of MPs from low to high trophic levels. For example, crabs (*Carcinus maenas*) that ingested microplastic-contaminated mussels (*Mytilus edulis*) reportedly accumulated ~167, 1007, and 68 particles in the gills, stomach, and ovaries, respectively (Farrell and Nelson, 2013). Another study showed that mysid shrimp (*Neomysis integer*) fed with plastic-contaminated copepods (*Eurytemora affinis*) exhibited MPs accumulation in the intestines 3 h after ingestion (Setälä *et al.* 2014). Studies on the tertiary transport of MPs along the marine food chain have found that particles can be transported along the algae-*Daphnia*-fish food chain (Cedervall *et al.* 2012; Mattsson *et al.* 2014) (Figure 3), resulting in accumulation at higher trophic levels. Moreover, organic pollutants (POPs) can be desorbed from MPs into *Artemia* nauplii and then transferred to zebrafish (*Danio rerio*) (Batel *et al.* 2016). This suggests that MPs ingested by marine animals can also potentially act as carriers of contaminants as they migrate through the food chain.

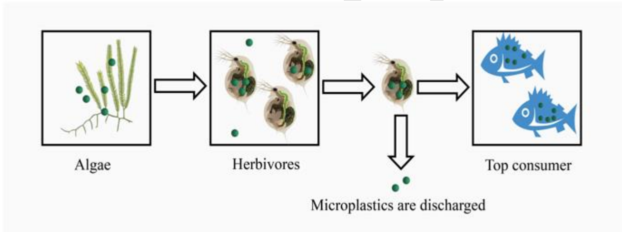


Figure 3. Migration of microplastics along the food chain.

2.3. Distribution status

MPs are widely distributed in the marine environment owing to their diverse modes of migration and are found along the water column from surface waters to deep-sea sediments as well as in numerous areas ranging from densely populated coastal waters to the sparsely populated polar areas (Table 1). Offshore regions are densely populated and frequently disturbed by human activities and are the location of numerous terrestrial sources of river pollution; therefore, microplastic pollution in these areas is more serious. MPs in offshore areas are mainly distributed in surface seawater, beaches, and sediments (Bagaev *et al.* 2018; Wang *et al.* 2018; Hengstmann *et al.* 2018; Lo *et al.* 2018; Taha *et al.* 2021). The analysis of the distribution of plastics in different areas, such as coast, bays, estuary, and around islands, reveals that offshore areas microplastic pollution has become quite common (Kim *et al.* 2015; Cheung *et al.* 2016; Zheng *et al.* 2019; Kang *et al.* 2015; Aytan *et al.* 2016; Collicutt*et al.* 2018).

Marine MPs are capable of long-range migration into distant oceans and polar regions under the influence of factors such as currents and monsoons. MPs mainly migrate with currents and have been found in the Northeast Atlantic (Lusher *et al.* 2014), Western Pacific (Liu *et al.* 2021), East Indian Ocean (Li *et al.* 2022), and South Pacific (Bakir *et al.* 2020). Currents are also responsible for the collection of floating plastic litter in five main global ocean circulation centers of comparable density (Cozar *et al.* 2014). At the same time, MPs are widespread in polar regions and, in particular, they have been detected in surface and subsurface seawater around the Svalbard archipelago, Arctic Sea (Lusher *et al.* 2015), sediments of the Canadian Arctic Sea (Adams *et al.* 2021), Weddell Sea (Cunningham *et al.* 2022), and surface seawater (Cincinelli *et al.* 2017) and sediments (Munari *et al.* 2015) in the Ross Sea, Antarctica.

Table 1. Status of microplastic pollution in some marine areas of the world

Region	**Research Location**	**Environmental Media**	**MPs**		**References**
			Abundance	**Type**	
Offshore regions	West Coast of Hong Kong, China	Surface water	528 ± 193–9067 ± 7009 items/m^2	PET, PVC	(Cheung *et al.* 2016)
	East Coast of Hong Kong, China	Surface water	1177 ± 570–1348 ± 755 items/m^2	PET, PVC	
	Jiaozhou Bay, China	Surface water	46 ± 28 items/m^3	PET, PP, PE	(Zheng *et al.* 2019)
	Baltic Sea	Surface water	0.40 ± 0.58 × 103 items/m^3	/	(Bagaev *et al.* 2018)
	Black Sea	Surface water	1.2 ± 1.1 × 103 items/m^3	/	(Aytan *et al.* 2016)
	Korean southeastern coastal	Surface water	0.64–860 items/m^3	PLY, PE, alkyd, PS	(Kang *et al.* 2015)
	Maowei Sea, China	Surface water	4.5 ± 0.1 × 103 items/m^3	RN, PLY	(Zhu *et al.* 2018a)
	Bohai Sea, China	Surface water	0.33 ± 0.34 items/m^3	PE, PP, PS	(Zhang *et al.* 2017)
	Yellow Sea, China	Surface water	0.330 ± 0.278 items/m^3	/	(Wang *et al.* 2018)
	North Yellow Sea, China	Surface water	545 ± 282 items/m^3	PE, PP	(Zhu *et al.* 2018b)
	Malaysia	Surface water	211.2 ± 104 items/m^3	PA, PE, PP	(Taha *et al.* 2021)
	Soya Island, Korea	Beach	56–285,673 items/m^2	EPS, PP, PE	(Kim *et al.* 2015)
	European	Beach	72 ± 24–151 ± 187 items/kg	PET, PP, PE	(Lots *et al.* 2017)
	Vancouver Island, British Columbia	Beach	60.2 ± 63.4 items/kg	/	(Collicutt *et al.* 2018)
	Baltic Sea	Beach	88.10 items/kg	PET, PVC	(Hengstmann *et al.* 2018)
	Hang Kong	Beach	0.58–2116 items/kg	PE, PP, PET	(Lo *et al.* 2018)
	Jiaozhou Bay, China	Sediments	15 ± 6 items/kg	PET, PP, PE	(Zheng *et al.* 2019)
	Yellow Sea, China	Sediments	2.58 ± 1.14 items/g	/	(Wang *et al.* 2018)
	North Yellow Sea, China	Sediments	37.1 ± 42.7 items/kg	PE, PP	(Zhu *et al.* 2018)
Ocean	Northeast Atlantic Ocean	Surface water	2.46 ± 2.43 items/m^3	/	(Lusher *et al.* 2014)
	Western Pacific	Surface water	0.02–0.10 items/m^3	PP, PE, PES, PMMA, PR, ER	(Liu *et al.* 2021)
	Eastern Indian Ocean	Surface water	0.40 ± 0.62 items/m^3	PP, PET, PE	(Li *et al.* 2022)
	Vanuatu, South Pacific	Surface water	0.09–0.57 items/m^3	PS, PE, PP, PEVA	(Bakir *et al.* 2020)
	Vanuatu, South Pacific	Sediments	333 ± 115–33,300 ± 7300 items/kg	/	
Polar Region	Svalbard, Arctic	Surface water	0.34 ± 0.31 items/m^3	PET, PA, PE, AC, PVC, CE	(Lusher *et al.* 2015)
		Subsurface water	2.68 ± 2.95 items/m^3		
	Ross Sea, Antarctica	Subsurface water	0.17 ± 0.34 items/m^3	PE, PP	(Cincinelli *et al.* 2017)
	Canadian Arctic-wide	Sediments	0.6–4.7 items/g	PVC, PAM, PS, PU, PE	(Adams *et al.* 2021)
	Ross Sea, Antarctica	Sediments	676.5 ± 536.4 items/ m^2（max）	PE, PP, PA, SBS, PVC, PS, TPU, PVA, EPR	(Munari *et al.* 2017)
Deep Sea	Research Location	Environmental Media	Microplastic Abundance	Depth(m)	Reference
	Mariana Trench	Seawater	2.06–13.51 items/dm3	2673–10908	(Peng *et al.* 2018)
	Mariana Trench	Sediment	200–2200 items/kg	5108–10908	
	Arctic	Sediment	42–6595 items/kg	2340–5570	(Bergmann *et al.* 2017)
	SW Indian Ocean	Sediment	28–80 items/kg	900–1000	(Woodall *et al.* 2014)
	NE Atlantic	Sediment	120–800 items/kg	1400–2200	
	Mediterranean	Sediment	200–700 items/kg	300–1300	
	Polar Front of the Southern Ocean	Sediment	0–40 items/kg	2419–4881	(Van Cauwenberghe *et al.* 2013)

Note: "/" indicates that the information is not specifically mentioned in the article.

MPs can also reach the sea floor or abyssal areas through vertical migration. A study of MPs in the Mariana Trench conducted by Peng *et al.* (2018) at the Chinese Academy of Sciences showed that the microplastic content of seawater at depths of 2,673–10,908 m was several times higher than that in the surface and subsurface layers of oceans. Sediment samples obtained from the Mariana Trench at depths of 5,108–10,908 m also contained considerably higher amounts of MPs compared with that in most deep-sea sediments collected from other seas (Table 1). In addition, a study involving MPs at three depths in the waters of Santa Monica Bay and California reported that the density of surface and mesopelagic MPs was less offshore than that on the seafloor (Lattin *et al.* 2004). Consequently, MPs abundance may be greater on the seafloor than in the surface layer offshore due to vertical migration.

3. Impacts of MPs on marine animals

3.1. Enrichment of MPs in marine animals

MPs in the ocean are abundant and poorly biodegradable and can therefore be easily ingested by marine animals with different feeding patterns and at different trophic levels (Figure 4). Typically, zooplankton, which are at the bottom of the food chain, feed on items that are within the size range of MPs, unintentionally ingesting particles and favoring their transport through the food chain (Telesh and Khlebovich, 2010). A study that evaluated the presence of MPs in 29 species of commercial fish collected from the Bohai Sea, China, found particles in about 85.4% of them (Wang *et al.* 2021). In addition, some studies have pointed out that in sea urchins (*Echinoidea*), which occupy an important position in the food chain and are key to its material cycle and energy flow, the MPs detection rate was as high as 89.52% (Feng *et al.* 2020; Dethier *et al.* 2019). It is clear that MPs are present in most marine animals.

3.2. Behavioral changes in marine animals affected by MPs

3.2.1. Ingestion behavior

Most marine animals feeding on MPs undergo changes in their ingestion behavior. For example, changes have been reported in the European perch (*Perca fluviatilis*) larvae (Lönnstedt and Eklöv, 2016), blue mussel (*Mytilus edulis*) (Wegner *et al.* 2012), mysid shrimp (*Mytilus japonica*) (Wang *et al.* 2019), and brine shrimp (*Artemia franciscana*) (Bergami *et al.* 2016). In most species, the altered behavior manifests as reduced feeding efficiency and predation capacity, resulting in reduced intake, increased mortality, and suppressed growth (Wegner *et al.* 2012; Bergami *et al.* 2016). For example, the exposure of mysid shrimp to PS MPs resulted in a 1–4-fold increase in mortality and a 4.38%–9.57% inhibition in growth, while exposure to PS–COOH increased mortality by 122%–293% and inhibited growth by 2.07%–16.6% (Wegner *et al.* 2012). In particular, *P. fluviatilis* larvae exposed to different PS concentrations exhibited altered feeding preferences; they no longer preferred to feed on prey but rather on MPs (Lönnstedt and Eklöv, 2016). Such alteration can negatively impact this species, reducing its ability to feed and increasing the mortality rate.

3.2.2. Motile behavior

MPs can also affect the motile behavior of marine animals. Numerous studies have reported that MPs affect the swimming activity of a wide range of organisms, including the mysid shrimp (*Neomysis japonica*) (Wang *et al.* 2019), brine shrimp (*Artemia franciscana*) (Bergami *et al.* 2016), Pacific oysters (*Crassostrea gigas*) (Bringer *et al.* 2020), and sheepshead minnow (*Cyprinodon variegatus*) (Choi *et al.* 2018). Among these, mysid shrimps exhibited a significant reduction by 25.2% in their maximum swimming speed when exposed to PS at a concentration of 250 nL^{-1} (Wang *et al.* 2019), and Pacific oysters exposed to microplastic microbeads demonstrated a decrease in maximum swimming speed (Bringer *et al.* 2020). A study that tested the effects of microplastic shapes on *Cyprinodon variegatus* revealed that the total swimming distance was smaller and maximum velocity was lower after exposure to irregularly shaped MPs than after exposure to spherical MPs (Choi *et al.* 2018).

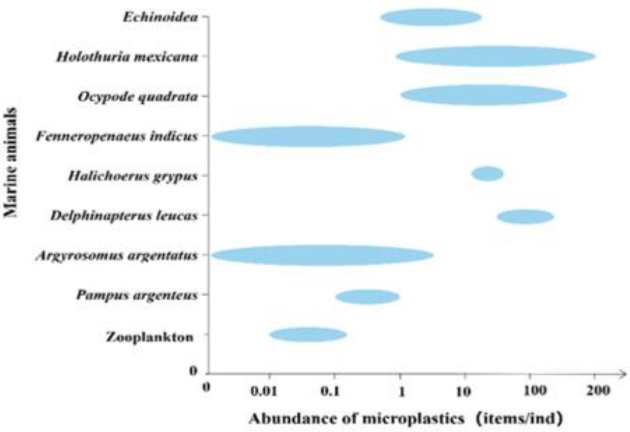


Figure 4. Microplastic contamination in some marine animals (detailed data are shown in Table S1).

3.3. Effects of MPs on the physiological activities of marine animals

3.3.1. Digestive system

The digestive tract is the main organ where marine animals accumulate MPs after ingesting them. MPs can harm fish by blocking their digestive system and enzyme production and affecting the intestinal flora (Wright *et al.* 2013). Reportedly, the most direct effect of MPs on the digestive tract of fish is intestinal injury, and the degree of damage varies from site to site, with the distal intestine being more damaged than the front and middle sections (Pedà *et al.* 2016). Moreover, the ingestion of MPs can have affect the relevant digestive enzymes and intestinal flora. For example, the exposure of juvenile large yellow croaker (*Larimichthys crocea*) to PS considerably altered the proportion of the three dominant bacterial phyla present in the intestinal tract. In addition, there was a substantial increase in the proportion of potentially pathogenic bacteria and decrease in lysozyme activity and specific growth rate, resulting in a higher total mortality of juvenile fish (Gu *et al.* 2020). In addition to damaging the intestine of fish, MPs affect the digestive system of Mediterranean mussels (*Mytilus galloprovincialis*) (Bråte *et al.* 2018) and whiteleg shrimp (*Litopenaeus vannamei*) (Chae *et al.* 2019).

Table 2. Effects of MPs on selected marine animals and the associated risks.

Effect Type	Subject Animal	MPs		Exposure Duration	Single-action Effects	Risks	References
		Type	Size				
Ingestion behavior	*Perca fluviatilis*	PS	90 µm	2 d	Change in ingestion preference	Predatory ability ↓, mortality ↑	(Lönnstedt and Eklöv, 2016)
	Mytilus edulis	PS	30 nm	8 h	Filtration rate decreased, leading to starvation	Negative effects	(Wegner *et al.* 2012)
	Neomysis japonica	PS, PS–COOH	5 µm	96 h	Reduced feeding efficiency and predatory ability	PS, PS–COOH: mortality rate ↑ (1–4 times), (122%–293%), growth inhibition rate ↑ (4.38%–9.57%), (2.07%–16.6%)	(Wang *et al.* 2019)
	Artemia franciscana	PS–COOH	40 nm	48 h	Reduced food intake	Affects survival under prolonged exposure	(Bergami *et al.* 2016)
Motor behavior	*Neomysis japonica*	PS, PS–COOH	5 µm	96 h	Decreased speed and frequency, reduced ability to hunt and explore	Mortality rate ↑, growth inhibition rate ↑	(Wang *et al.* 2019)
	Artemia franciscana	PS-NH2	50 nm	48 h	Reduced motor ability	Affects survival under prolonged exposure	(Bergami *et al.* 2016)
	Crassostrea gigas	/	1–5 µm	24 h	Changing swimming behavior (speed and trajectory)	No significant mortality, significant sublethal effects	(Bringer *et al.* 2020)
	Cyprinodon variegatus	PE	150–180 µm, 6–350 µm	4 d	Total distance swam and maximum speed reduced		(Choi *et al.* 2018)
Digestive system	*Dicentrarchus labrax*	PVC	<0.3 mm	90 d	Impaired bowel function		(Pedà *et al.* 2016)
	Larimichthys crocea	PS	100 nm	14 d	Decrease in enzyme activity, change in the proportion of dominant phyla, increase in potential pathogenic bacteria	Total mortality rate ↑ (up to 42.8% at high concentrations)	(Gu *et al.* 2020)
	Mytilus galloprovincialis	PE	50–570 µm	21 d	Digestive tube epithelial cell thinning, digestive tract structural changes, blood cell tissue necrosis	Negative effects	(Bråte *et al.* 2018)
	Litopenaeus vannamei	PS	44 nm	21 d	Altered microbial and enzymatic activity		(Chae *et al.* 2019)
Metabolic activity	*Epinephelus moara*	PS	22.3 µm, 20–100	/	Lipid deposition, liver lesions, growth inhibition	No significant mortality, significant sublethal effects	(Wang *et al.* 2020b)

			μm, −38.6μm				
	Carassius	PS (nanoparticle)	24 nm	/	Liver damage and changes in fat metabolism	Negative effects	(Cedervall *et al.* 2012)
	Cherax quadricarinatus	PS (microsphere)	200 nm	21 d	Affected lipid metabolism		(Chen *et al.* 2020b)
Respiratory effect	*Gasterosteus aculeatus*	PE, PES	27–32 μm, 500-μm long	2 h	Long retention time of microplastics in gills		(Pratte *et al.* 2018)
	Oryzias latipes	PES, PP	10–20 μm, 50–60 μm	21 d	Changes in gill morphology, affecting respiration	Affects survival under prolonged exposure	(Hu *et al.* 2020)
	Ostrea edulis	PVC, HDPE	0.6–363 μm 0.48–316 μm	2 mo	Increased respiration rate	Decrease in biomass	(Green, 2016)
	Carcinus maenas	PS	8 μm	1 h	Affects gill function and reduces oxygen consumption	No significant mortality, significant sublethal effects	(Watts *et al.* 2016)
	Perna viridis	PVC	1–50 μm	44 d	Reduced respiratory rate	Mortality rate ↑	(Rist *et al.* 2016)
	Mytilus edulis	PS	40 mm	14 d	Increased respiration	Affects survival under prolonged exposure	(Van Cauwenberghe *et al.* 2015)
Reproductive development	*Oryzias latipes*	PE	<0.5 mm	2 mo	Downregulation of reproductive gene expression, abnormal germ cell proliferation in male fish	Reproduction rate ↓, affecting individuals and populations	(Rochman *et al.* 2014)
	Crassostrea gigas	PS	2, 6 μm	2 mo	Decrease in oocyte number and diameter, sperm velocity	Larval production↓(41%), larval development rate↓ (18%)	(Sussarellu *et al.* 2016)
	Emerita analoga	PP	< 0.1mm	71 d	Decrease in egg hatching rate and change in embryo development rate	Mortality↑	(Horn *et al.* 2019)
	Lytechinus variegatus	PE	/	24 h	Increased rate of developmental abnormalities	Abnormal larval development↑ (17.2%–53.3%) Abnormal development of embryos↑ (55.2%–61.9%)	(Nobre *et al.* 2015)

Note: "/" indicates that the information is not specifically mentioned in the article.

3.3.2. Metabolic activity

The ingestion of MPs by marine animals can also disrupt metabolic activities. In fish, the main source of energy is lipid metabolism, which mainly occurs in the liver (Greene, 1987). MPs exposure damages fish through liver injury. For example, juvenile groupers (*Epinephelus moara*) that were fed with three different PS microbeads (i.e., C-PS, P-PS, and PD-PS), and it was found that bioaccumulation of all three types of MPs in the liver increases and releases endogenous toxins, which led to lipidosis-driven hepatic lesions of groupers (Wang *et al.* 2020b). Cedervall *et al.* (2012) studied the effects of nano-PS MPs on crucian carp (*Carassius carassius*) after transmission through the tertiary food chain and detected liver damage and changes in lipid metabolism due to nanoparticles binding to apolipoprotein A-I in serum and inhibiting the utilization of body fat reserves regulated by this protein. In addition, the ingestion of MPs by the crustacean redclaw crayfish

(*Cherax quadricarinatus*) resulted in disruption of hepatopancreatic lipid metabolism and inhibition of hepatopancreatic cells' ability to utilize fatty acids (Chen *et al.* 2020b).

3.3.3. Respiratory effects

Exposure to MPs can have effects on the respiratory system of marine animals. The gills are the main site of respiration and waste exchange in fish and are one of the organs directly exposed to microplastics (Pratte *et al.* 2018). Three-spined sticklebacks (*Gasterosteus aculeatus*) stored more MPs in the gills and retained them for a longer period of time on average after exposure to MPs compared to the gut (Bour *et al.* 2020). This suggested that the gills are the organ susceptible to MPs exposure. In addition, MPs can affect the normal function of the gills, with repercussions on fish respiration. For example, adult Japanese medaka (*Oryzias latipes*) exposed to different concentrations of PES and PP were found to develop an exfoliated gill arch epithelium, fused primary gill lamellae, increased mucus in the gill lumen, and an altered morphology of most of the gill lamellae on the mouth side (Hu *et al.* 2020).

Effects on respiration have also been reported in shellfish and crustaceans. Both shore crab (*Carcinus maenas*) (Watts *et al.* 2016) and Asian green mussel (*Perna viridis*) (Rist *et al.* 2016) were found to have reduced respiration rates after exposure to MPs. European flat oysters (*Ostrea edulis*) exposed to high concentrations of PLA (polylactic acid) had increased respiration rates compared to the control group and the group exposed to high concentrations of HDPE (conventional high-density polyethylene), and the respiration rate was 2.6 times higher than that of the group exposed to high concentrations of HDPE (Green, 2016). Another study found that respiration increased in blue mussel (*Mytilus edulis*) exposed to MPs compared to controls and suggested that it may be due to the organisms' attempts to maintain physiological homeostasis while dealing with increased stress (Van Cauwenberghe *et al.* 2015).

3.3.4. Reproductive development

Exposure to MPs can have effects on reproductive development in marine animals, particularly on reproductive gene expression, embryo development, and reproductive rates. In Japanese medaka (*Oryzias latipes*), PE exposure for 2 months considerably downregulated reproductive gene expression in males and females and abnormal germ cell proliferation in males (Rochman *et al.* 2014). In adult Pacific oyster (*Crassostrea gigas*), exposure to PS microspheres of different sizes at certain concentrations for 2 months substantially decreased oocyte number (−38%) and diameter (−5%) and sperm velocity (−23%), thereby decreasing larval production and development rate (Sussarellu *et al.* 2016). Microplastic toxicity studies reported that MPs induced abnormal embryonic development in Pacific mole crabs (*Emerita analoga*) (Horn *et al.* 2019) and sea urchins (*Lytechinus variegatus*) (Nobre *et al.* 2015). Specifically, in sea urchin embryos exposed to virgin and beach-stranded plastic particles, the proportion of developmental abnormalities increased by 58.1% and 66.5%, respectively. Other studies have reported that MPs may increase spawning rates by acting as spawning vectors for certain marine animals. For example, the spawning density of the pelagic insect *Halobates sericeus* increased with increasing microplastic content (Goldstein *et al.* 2012).

4. Impacts of MPs with adsorbed pollutants on marine animals

4.1. Pollutant sorption by MPs in the marine environment

With the rapid development of the global economy, a large number of pollutants (such as organic pollutants, heavy metals, and MPs) continue to enter the marine environment. MPs formed under natural conditions have special surface characteristics, such as high porosity and a large specific surface area, which confer them a strong adsorption capacity (Hirai *et al.* 2011), allowing them to easily adsorb organic pollutants and heavy metals from the surrounding environment, as shown in *Figure* 5 (The data contained in are only some of the collected data,which does not show the reality of the totality of the studies about metals in MPs).

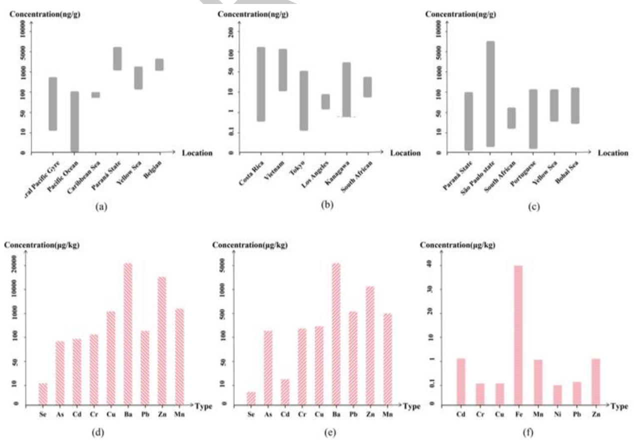


Figure 5. Schematic representations of the concentration of PAHs (a), DDT (b), and PCBs (c) in different locations worldwide, and graphs illustrating heavy metal types and concentrations at North Freemantle beach (d), Redhead beach (e), and Milna beach (f). ("---" indicates non-determined data. Detailed data are shown in Table S2)

4.2. Impact of the adsorption of organic pollutants by microplastics on marine animals

4.2.1. Synergistic effects

The adsorption of organic pollutants on MPs can synergistically affect marine animals. MPs can carry organic pollutants into tissues or organs, consequently increasing contaminant concentrations and leading to bioaccumulation and toxic effects. For example, Avio *et al.* (2015) reported that pyrene adsorbed on PE and PS particles could be transferred to the hemolymph, gills, and digestive system of mussels (*Mytilus galloprovincialis*), where it increasingly accumulated. Granby *et al.* (2018) reported that halogenated contaminants adsorbed onto MPs could be absorbed by the intestinal system of European seabass (*Dicentrarchus labrax*) and transferred to the circulatory system, leading to increased contaminant bioaccumulation and enhanced toxic effects (e.g., altered liver metabolism and immune system and oxidative stress). Bellas *et al.* (2020) reported that in *Acartia tonsa*, chlorpyrifos (CPF) was 4–25 times more toxic in combination with PE than alone. Tang *et al.* (2019) revealed that the combined toxic effects of PS with benzo[a]pyrene (B[a]P) and 17β-estradiol (E2) reduced the number and proportion of hemocytes in the ark shell *Tegillarca granosa*, limiting the recognition, phagocytosis, and degradation of foreign substances by these specialized cells, and thus reducing the immunity conferred by them. Browne *et al.* (2013) reported that in lugworms (*Arenicola marina*), the combination of PVC and nonylphenol caused at least a 60% reduction in the ability of colonocytes to remove pathogenic bacteria. In addition, the combination of PVC and triclosan reduced the ability of lugworms to modify sediment, resulting in a mortality rate of >55%.

Table 3. Effects of MPs adsorbed with pollutants on selected marine animals and associated risks

Effect	MPs		Exposure Duration	Organic /Heavy Metal Type	Subject Animal	Co-action Effects	Risks	References
	Type	Size						
Synergistic	PE PS	≤1000 µm	7 d	Pyrene	*Mytilus galloprovincialis*	Increases the absorption of pyrene and affects immune response, genotoxicity and neurotoxicity	Affects survival in long-term exposure situations	(Avio *et al.* 2015)
	PE	<400 µm	40 d	Halogenate	*Dicentrarchus labrax*	Increased bioaccumulation, enhanced toxicity		(Granby *et al.* 2018)
	PS	30 µm, 500 nm	4 d	B[a]P, E2	*Tegillarca granosa*	Increases bioaccumulation and reduces blood cell capacity (recognition, phagocytosis, degradation)		(Tang *et al.* 2019)
	PE	1.4–42 µm	24 h	CPF	*Acartia tonsa*	Increased toxicity	Survival rate ↓, egg laying rate ↓, ingestion ↓, hatching rate ↓	(Bellas *et al.* 2020)
	PVC	<330 µm	10 d	Nonylphenol, phenanthrene, Triclosan PBDE-47	*Arenicola marina*	Reduces survival and ingestion, impairs immune system and antioxidant system	Ability to remove pathogenic bacteria ↓ (60%), mortality ↑ (55%), ingestion ↓ (30%)	(Browne *et al.* 2013)
Antagonistic	PE	1–5 µm	96 h	Pyrene	*Pomatoschistus microps*	Significant delayed mortality	100% mortality of fish, toxic effects directly or indirectly affect individual and population health	(Oliveira *et al.* 2013)

	PVC	200–250 µm	96 h	Phe, EE2	*Danio rerio*	Reduced bioavailability and toxicity of Phe and EE2	-	(Sleight *et al.* 2017)
	PS	0.1 µm, 0.55 µm, 5 µm	96 h	DBP	*Tigriopus japonicus*	Reduced bioavailability and toxicity of DBP	Affects survival in long-term exposure situations	(Li *et al.* 2020b)
Synergistic	PE	1–5 µm	96 h	Hg	*Dicentrarchus labrax*	Increased bioaccumulation, neurotoxicity, oxidative stress and damage, and altered activity of energy-related enzymes	Directly or indirectly affect individual and population health	(Barboza *et al.* 2018)
	PS	32–40 µm	30 d	Cd	*Symphysodon aequifasciatus*	Oxidative stress, stimulation of innate immune response		(Wen *et al.* 2018)
	PS	5 µm	3 w	Cd	*Danio rerio*	Increased bioaccumulation, producing oxidative damage and inflammatory responses		(Lu *et al.* 2018)
	PE	1–5 µm	96 h	Cr	*Pomatoschistus microps*	Reduces predatory behavior and AChE activity in fish		(Luís *et al.* 2015)
Antagonistic	PS	32–40 µm	30 d	Cd	*Symphysodon aequifasciatus*	Reduced bioaccumulation		(Wen *et al.* 2018)
	PS	201.5–191.3 nm	48 h	Ni	*Daphnia magna*	Reduced toxicity		(Kim *et al.* 2017)
	PET	150 µm, 3–5 mm diameter 20 µm	72 h	Cd	*Danio rerio*	Reduced bioaccumulation and toxicity	Incubation rate ↓	(Cheng *et al.* 2020)

Note: "/" indicates that the information is not specifically mentioned in the article.

4.2.2. Antagonistic effects

The adsorption of organic pollutants on MPs also produces antagonistic effects in marine animals. Through adsorption, MPs can reduce the free state of these harmful substances in the environment, thus reducing their concentration, bioavailability, and toxicity (Wang *et al.* 2020a). For example, Oliveira *et al.* (2013) observed a substantial delay in mortality in juvenile common gobies (*Pomatoschistus microps*) exposed to pyrene and polyethylene MPs compared with those exposed to pyrene alone. Sleight *et al.* (2017) reported that the adsorption of phenanthrene (Phe) and 17α-ethinylestradiol (EE2) by MPs reduced their bioavailability to zebrafish larvae (*Danio rerio*) by 33% and 48%, respectively. Li *et al.* (2020b) studied the combined effects of dibutyl phthalate (DBP) and polystyrene MPs on the marine copepod *Tigriopus japonicus* using acute mortality tests and found that the MPs reduced the bioavailability of DBP, consequently reducing its toxicity.

4.3. Impact of the adsorption of heavy metals on MPs on marine animals

4.3.1. Synergistic effects

MPs can act as carriers of heavy metals in marine systems, increasing the bioaccumulation and toxic effects of these compounds in marine animals. For example, Barboza *et al.* (2018) found that MPs can adsorb Hg from the surrounding water environment, resulting in a significant increase in Hg accumulation in European seabass (*Dicentrarchus labrax*). Specifically, the individuals exposed to Hg-contaminated MPs showed a 76% increase in accumulation compared to those exposed only to Hg. Wen *et al.* (2018) showed that exposure to Cd or MPs had no effects on the survival and growth of juvenile discus fish (*Symphysodon aequifasciatus*), but the combined exposure induced oxidative stress and stimulated innate immune responses. Similarly, Lu *et al.* (2018) reported that exposure to Cd-contaminated MPs resulted in an increased bioaccumulation of Cd in zebrafish (*Danio rerio*) and that the toxic effects were higher compared to those observed under the exposure to Cd alone. Luís et al (2015) found that Cr exposure alone significantly decreased the predatory performance (≤74%) of juvenile gobies (*Pomatoschistus microps*). Whereas, the combination of MPs and Cr resulted in reduced gobies predatory performance (≤67%) and significant inhibition of AChE activity (≤31%).

4.3.2. Antagonistic effects

The interaction of MPs and heavy metals also causes antagonistic effects in marine animals. In particular, it decreases the bioaccumulation and toxic effects of heavy metals. For example, Wen *et al.* (2018) found that the accumulation of Cd in juvenile discus fish (*Symphysodon aequifasciatus)* decreased as the concentration of microplastics increased. Cheng *et al.* (2020) studied the combined effect of MPs and Cd in zebrafish embryos and detected a reduced bioavailability and significantly reduced accumulation of this heavy metal, which resulted in reduced toxicity. Another study showed that simultaneous exposure to microplastics and Ni was less toxic to *Daphnia magna* compared to the exposure to Ni alone (Kim *et al.* 2017).

5. MPs with or without adsorbed contaminants do not affect marine animals

5.1. Differential results on the impacts of MPs with or without adsorbed contaminants on marine animals

Numerous studies have been conducted to show that MPs are easily ingested by marine animals and stored in the body; specifically, smaller particles enter organisms, penetrate cell membranes, and reach tissues and cells, producing toxic effects at the cellular and molecular levels (Rist and Hartmann, 2017) and affecting the behavior and physiology of marine animals. However, other studies have reached the opposite conclusion. For example, sea cucumber (*Apostichopus japonicus*) exhibited no significant change in its swimming speed and distances covered in the short term after exposure to MPs (Mohsen *et al.* 2019). Juvenile spiny chromis (*Acanthochromis polyacanthus*), a planktivorous fish, demonstrated no significant behavioral or physiological changes after exposure to PET MPs with a diameter of 2 mm (Critchell and Hoogenboom, 2018). Exposure to 0.05-µm PS MPs may lead to mortality in the nauplii of the marine copepod *Tigriopus japonicus* due to nutritional deficiencies or digestive inhibition; however, 6-µm MPs did not have any effect on their survival (Lee *et al.* 2013).

Similarly, some studies have concluded that there are no toxicological interactions between MPs and pollutants, meaning that there are no synergistic or antagonistic effects. For example, in planktonic sea urchin larvae that ingested PE MPs, the toxicity of the hydrophobic organic chemical 4-nonylphenol (NP) did not increase (Beiras and Tato, 2019); in mussels, PVC MPs exhibited no significant effects on Cd uptake into digestive tissues, and combined exposure to PVC and Cd did not cause additional deleterious effects on mussel health compared with Cd exposure alone (Li *et al.* 2020a).

5.2. Reasons for impact or non-impact

The different conclusions present in the literature depend on many factors. In the case of fish, contrasting results may be related to the ability of fish to recognize MPs (Ory *et al.* 2018). Reportedly, in most cases, fish spit out isolated MPs but swallow particles that are floating near food. This may be because of the inability of fish to discriminate and reject MPs while feeding (e.g., the spitting behavior of zebrafish after microplastic ingestion) (Ory *et al.* 2018; Kim *et al.* 2018). Based on the above finding, it has been suggested that smaller the microplastic particles, lower the discrimination ability of fish and higher the intake of MPs (Critchell and Hoogenboom, 2018).

Moreover, the type (Horn *et al.* 2018), shape (Choi *et al.* 2018, Beiras and Tato, 2019), and size (Sussarellu *et al.* 2016) of MPs as well as the concentration (Lee *et al.* 2013; Mbedzi *et al.* 2019; Wen *et al.* 2018) and duration (Mohsen *et al.* 2019) of exposure to marine animals varied in different studies; furthermore, the affinity of different pollutants to MPs (Koelmans *et al.* (2013a), (2013b)) differed. In addition to these exposure-related factors, the fact that the affected animals have different biosensitivities, decontamination capabilities (Wang *et al.* 2019), and gastrointestinal conditions (Chua *et al.* 2014) may contribute to variable results. At the same time, some aspects related to the experiments conducted cannot be ignored. For example, some experiments are acute, at which point the possible chronic effects of MPs should not be ruled out, and the health of the subject animals may be potentially at risk under conditions of long-term, chronic exposure (Bellas and Gil, 2020; Sun *et al.* 2022); using smaller particles, laboratory experiments may overestimate the effects of MPs in nature (Burns *et al.* 2018); marine animals in their natural environments are more bioresistant to MPs than those cultured in the laboratory (Belzunce-Segarra *et al.* 2015).

6. Risks posed by MPs with and without adsorbed contaminants to marine animals

In recent years, the effects of MPs with and without adsorbed contaminants on marine animals have been increasingly studied to better assess exposure risks in the marine environment (*Figure* 6). Thus, both microplastic types may or may not impact marine animals; however, as shown in Tables 2 and 3, the effects produced lead to risks. Moreover, a recent study reported that the combined effect of MPs and pollutants considerably increased the bioaccumulation of the latter by 31% and exacerbated their toxicity by 18% (Sun *et al.* 2022).

On the one hand, MPs with and without adsorbed contaminants pose a risk to marine animals both at the individual and population scales. For example, MPs were shown to affect the behavior and physiological activities of the mysid shrimp (Wang *et al.* 2019) and brine shrimp (Bergami *et al.* 2014), leading to growth inhibition and increased mortality. The exposure of adult Pacific oysters to microplastic-contaminated environments resulted in reduced larval production and development rates (Sussarellu *et al.* 2016). Compared to the exposure to pyrene alone, the combination of pyrene and PE MPs caused a more significant delay in mortality in common goby juveniles; however, they continued to die, and this combined effect was also shown to potentially affect their growth, reproduction, and behavioral activities (Oliveira *et al.* 2013). The combined action of MPs and Ni on *Daphnia magna* was shown to reduce the toxicity of this element, but direct or indirect effects on the health of individuals and population were still observed (Kim *et al.* 2017).

On the other hand, MPs with and without adsorbed contaminants can migrate along the marine food chain, posing a potential risk to the entire marine ecosystem. For example, the swimming behavior of marine animals is negatively affected, resulting in a reduced ability to hunt and avoid predators, which will further alter prey–predator relationships and energy flows in marine food webs (Wang *et al.* 2019). Copepods ingest phytoplankton and zooplanktonic protozoa and are in turn consumed by fish. This represents an ecological link between primary producers and the microbial cycle as well as higher trophic levels (Roman, 2000; Reeve and Walter, 1977). A study found that the combined action of CPF, an organic pollutant, and MPs led to increased toxicity in the copepod *Acartia tonsa*. This increased toxicity poses not only a direct or indirect risk to copepod individuals and populations, but also a potential risk to predators along the food chain, threatening the entire marine ecosystem (Bellas and Gil, 2020). Similarly, the increase in Hg concentrations in fish brain and muscle caused by MPs increases the health risks to fish and predators, potentially affecting entire marine ecosystems (Barboza *et al.* 2018).

7. Potential risks of MPs for human health

A study published in 2019 assessed the impact of environmental pollution on cardiovascular disease in 28 European countries and estimated that it reduced the average life expectancy by ~2.2 years (Lelieveld *et al.* 2019). Long-term exposure to high concentrations of MPs, which are one of the factors of environmental pollution, may pose risks to human health. Currently, studies have confirmed that MPs can enter and accumulate in the human body and cause damages. For example, a recent study by Antonio *et al.* (2021) detected 12 spherical- or irregular-shaped microplastic fragments ranging in size from 5 to10μm on the fetal side, maternal side, and chorionic villus of the human placenta. Dong *et al.* (2019) reported that polystyrene MPs can cause the inflammation of lung cells by inducing the production of reactive oxygen species and that exposure to high concentrations of these MPs may increase the risk of chronic obstructive pulmonary disease. MPs pollution affects a large number of marine animals, and there are risks of potential transmission to humans through the food chain with negative consequences. For example, as MPs are usually stored in the intestine after ingestion by marine animals, their consumption (for example that of fish, mussel, or crab offal) may pose a threat to human health (Farrell and Nelson, 2013; Wegner *et al.* 2012; Barboza *et al.* 2018). Therefore, concerns should be raised about the potential for MPs to reach higher trophic levels, accumulation of environmental pollutants, and health of animals, including humans.

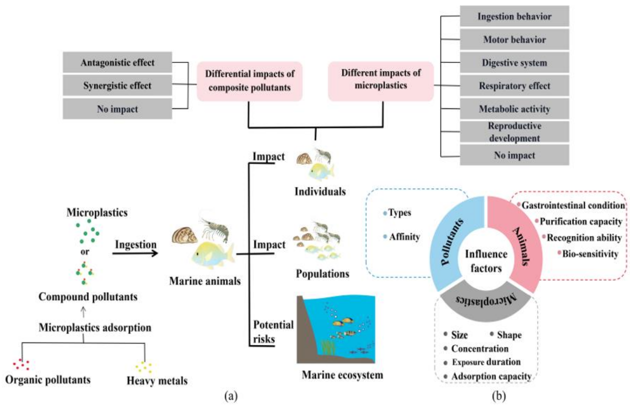


Figure 6. Schematic diagram of the impacts of MPs with and without adsorbed organic compounds or heavy metals on marine animals and associated risks (a) and influencing factors connected to different effects (b)

8. Conclusion and prospect

The behavior of MPs after entering the marine environment can be divided into three categories: physical (transport and deposition), chemical (degradation and adsorption), and biological (ingestion and migration) (Wang *et al.* 2016). A large number of studies have reported that marine MPs may be ingested by marine animals, altering their behavior and physiological activities. In addition to the toxicity of MPs, the combined toxicity of MPs with adsorbed pollutants to marine animals is at the forefront of research. Reportedly, >78% (Rochman *et al.* 2013b) of the pollutants adsorbed onto plastic debris are classified as "priority pollutants" (Rochman, 2013a) and can be harmful to marine animals.

MPs, as a new type of pollutant, not only considerably impact the marine environment but also are widely present in marine animals and may pose a risk to them. The environmental and biological effects of MPs in the ocean have been investigated; however, their abundance and physicochemical properties, effects on the marine water environment, and mechanisms underlying their single or combined effects on marine animals and humans need to be further elucidated. Therefore, future research needs to focus on the following aspects:

The effects of MPs and pollutants on marine animals remain inconsistency, and more research is warranted on a wide range of chemicals and plastic types to provide a theoretical basis for preventing and controlling marine microplastic pollution.

Most models for risk assessment have been built using data obtained from experiments regarding the toxicity of single MPs (Santos *et al.* 2021). However, in the natural environment, the combination of MPs and other pollutants is inevitable. The synergistic effects arising from such interaction deserve consideration and attention during risk assessments.

There are significant differences between laboratory and real-world conditions, and more ecological and environmentally relevant research is needed to fill the knowledge gap on how to extrapolate laboratory results that correspond to actual conditions in the natural environment.

Most research on MPs has been conducted on animals, and research involving humans remains in its initial stage. In the future, research regarding the risks of MPs for human health should be strengthened to reveal the mechanisms underlying the effects at the genetic, cellular, and tissue levels.

Based on the study of the mechanisms of microplastic toxicity, attention should be focused on the migration pattern and effects of MPs along the food chain to provide a theoretical basis for preventing and controlling potential risks to the marine ecosystem and human health.

Supplementary materials

The following supporting information can be downloaded at: www.xxx.com/xxx/s1. Table S1: Microplastic contamination in some marine animals; Table S2: Coexistence of pollutants and MPs in some marine areas.

Author contributions

Conceptualization, methodology, supervision, Y.Q.; Writing—original draft and reviewing, C.C.; investigation, visualization, Y.T.; investigation, methodology, F.W.; supervision, Y.Y.; visualization, W.C. All authors have read and agreed to the published version of the manuscript.

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Declaration of competing interest

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

Data availability statement

Data will be made available on request.

Reference

Adams J.K., Dean B.Y., Athey S.N., Jantunen L.M., Bernstein S., Stern G., Diamond M.L., Finkelstein S.A. (2021), Anthropogenic particles (including microfibers and microplastics) in marine sediments of the Canadian Arctic, *Science of The Total Environment*, **784**, 147155.,

Alimi O.S., Farner Budarz J., Hernandez L.M., Tufenkji N. (2018). Microplastics and Nanoplastics in Aquatic Environments: Aggregation, Deposition, and Enhanced Contaminant Transport, *Environmental Science & Technology*, **52**, 1704-1724.

Andrady A.L. (2011). Microplastics in the marine environment, *Marine Pollution Bulletin*, **62**, 1596-1605.

Antunes J.C., Frias J.G.L., Micaelo A.C., Sobral P. (2013). Resin pellets from beaches of the Portuguese coast and adsorbed persistent organic pollutants, *Estuarine Coastal & Shelf Science*, **130**, 62-69.

Avio C.G., Gorbi S., Milan M., Benedetti M., Fattorini D., d' Errico G., Pauletto M., Bargelloni L., Regoli F. (2015). Pollutants bioavailability and toxicological risk from microplastics to marine mussels, *Environmental Pollution*, **198**, 211-222.

Aytan U., Valente A., Senturk Y., Usta R., Sahin F.B.E., Mazlum R.E., Agirbas E. (2016). First evaluation of neustonic microplastics in Black Sea waters, *Marine Environmental Research*, **119**, 22-30.

Bagaev A., Khatmullina L., Chubarenko I. (2018). Anthropogenic microlitter in the Baltic Sea water column, *Marine Pollution Bulletin*, **129**, 918-923.

Bakir A., Desender M., Wilkinson T., Van Hoytema N., Amos R., Airahui S., Graham J., Maes T. (2020). Occurrence and abundance of meso and microplastics in sediment, surface waters, and marine biota from the South Pacific region, *Marine Pollution Bulletin,* **160**, 111572.

Barboza L.G.A., Vieira L.R., Branco V., Figueiredo N., Carvalho F., Carvalho C., Guilhermino L. (2018). Microplastics cause neurotoxicity oxidative damage and energy-related changes and interact with the bioaccumulation of mercury in the European seabass *Dicentrarchus labrax* (Linnaeus 1758). *Aquatic Toxicology*, **195**, 49-57.

Barnes D.K.A., Galgani F., Thompson R.C., Barlaz M. (2009). Accumulation and fragmentation of plastic debris in global environments, *Philosophical Transactions of the Royal Society B: Biological Sciences*, **364**, 1985-1998.

Batel A., Linti F., Scherer M., Erdinger L., Braunbeck T. (2016). Transfer of benzo[a]pyrene from microplastics to *Artemia nauplii* and further to zebrafish via a trophic food web experiment: CYP1A induction and visual tracking of persistent organic pollutants, *Environmental Toxicology and Chemistry*, **35**, 1656-1666.

Beiras R., Tato T. (2019). Microplastics do not increase toxicity of a hydrophobic organic chemical to marine plankton, *Marine Pollution Bulletin*, **138**, 58-62.

Bellas J., Gil I. (2020). Polyethylene microplastics increase the toxicity of chlorpyrifos to the marine copepod *Acartia tonsa*, *Environmental Pollution*, **260**, 114059.

Belzunce-Segarra M.J., Simpson S.L., Amato E.D., Spadaro D.A., Hamilton I.L., Jarolimek C.V., Jolley D.F. (2015). The mismatch between bioaccumulation in field and laboratory environments: Interpreting the differences for metals in benthic bivalves, *Environmental Pollution*, **204**, 48-57.

Bergami E., Bocci E., Vannuccini M.L., Monopoli M., Salvati A., Dawson K.A., Corsi I. (2016). Nano-sized polystyrene affects feeding behavior and physiology of brine shrimp *Artemia franciscana larvae*, *Ecotoxicology and Environmental Safety*, **123**, 18-25.

Bergmann M., Wirzberger V., Krumpen T., Lorenz C., Primpke S., Tekman M.B., Gerdts G. (2017). High Quantities of Microplastic in Arctic Deep-Sea Sediments from the HAUSGARTEN Observatory, *Environmental Science & Technology*, **51**, 11000-11010.

Boucher J., Friot D. (2017). *Primary microplastics in the oceans: A global evaluation of sources*. Publisher: IUCN Gland Switzerland 43pp.

Bour A., Hossain S., Taylor M., Sumner M., Carney Almroth B. (2020). Synthetic Microfiber and Microbead Exposure and Retention Time in Model Aquatic Species Under Different Exposure Scenarios, *Frontiers in Environmental Science*, **8**, 83.

Bråte I.L.N., Blázquez M., Brooks S.J., Thomas K.V. (2018). Weathering impacts the uptake of polyethylene microparticles from toothpaste in Mediterranean mussels (*M. galloprovincialis*), *Science of The Total Environment*, **626**, 1310-1318.

Breivik Ø., Allen A.A., Maisondieu C., Roth J.C. (2011). Wind-induced drift of objects at sea: The leeway field method, *Applied Ocean Research*, **33**, 100-109.

Bringer A., Cachot J., Prunier G., Dubillot E., Clérandeau C., Thomas H. (2020). Experimental ingestion of fluorescent microplastics by pacific oysters, *Crassostrea gigas* and their effects on the behaviour and development at early stages, *Chemosphere*, **254**, 126793.

Browne M.A., Niven S.J., Galloway T.S., Rowland S.J., Thompson R.C. (2013). Microplastic Moves Pollutants and Additives to Worms Reducing Functions Linked to Health and Biodiversity, *Current Biology*, **23**, 2388-2392.

Bullard J.E., Ockelford A., O'Brien P., McKenna Neuman C. (2021). Preferential transport of microplastics by wind, *Atmospheric Environment*, **245**, 118038.

Burns E.E., Boxall A.B.A. (2018). Microplastics in the aquatic environment: Evidence for or against adverse impacts and major knowledge gaps, *Environmental Toxicology and Chemistry*, **37**, 2776-2796.

Carbery M., MacFarlane G.R., O'Connor W., Afrose S., Taylor H., Palanisami T. (2020). Baseline analysis of metal(loid)s on microplastics collected from the Australian shoreline using citizen science, *Marine Pollution Bulletin*, **152**, 110914.

Carpenter E.J., Smith K.L. (1972). Plastics on the Sargasso Sea Surface, *Science*, **175**, 1240-1241.

Cedervall T., Hansson L.A., Lard M., Frohm B., Linse S. (2012). Food Chain Transport of Nanoparticles Affects Behaviour and Fat Metabolism in Fish, *PLoS ONE*, **7(2)**, e32254.

Chae Y., Kim D., Choi M.J., Cho Y., An Y.J. (2019). Impact of nano-sized plastic on the nutritional value and gut microbiota of whiteleg shrimp *Litopenaeus vannamei* via dietary exposure, *Environment International*, **130**, 104848.

Chen C.F., Ju Y.R., Lim Y.C., Chen C.W., Dong C.D. (2021). Seasonal variation of diversity weathering and inventory of microplastics in coast and harbor sediments, *Science of The Total Environment*, **781**, 146610.

Chen M.C., Chen T.H. (2020a). Spatial and seasonal distribution of microplastics on sandy beaches along the coast of the Hengchun Peninsula, Taiwan, *Marine Pollution Bulletin*, **151**, 110861.

Chen Q., Lv W., Jiao Y., Liu Z., Li Y., Cai M., Wu D., Zhao Y. (2020b). Effects of exposure to waterborne polystyrene microspheres on lipid metabolism in the hepatopancreas of juvenile redclaw crayfish, *Cherax quadricarinatus*, *Aquatic Toxicology*, **224**, 105497.

Cheng H., Feng Y., Duan Z., Duan X., Zhao S., Wang Y., Gong Z., Wang L. (2020). Toxicities of microplastic fibers and granules on the development of zebrafish embryos and their combined effects with cadmium, *Chemosphere*, **269**, 128677.

Cheung P.K., Cheung L.T.O., Fok L. (2016). Seasonal variation in the abundance of marine plastic debris in the estuary of a subtropical macro-scale drainage basin in South China, *Science of The Total Environment*, **562**, 658-665.

Cheung P.K., Fok L. (2017). Characterisation of plastic microbeads in facial scrubs and their estimated emissions in Mainland China, *Water Research*, **122**, 53-61.

Choi J.S., Jung Y.J., Hong N.H., Hong S.H., and Park J.W. (2018). Toxicological effects of irregularly shaped and spherical microplastics in a marine teleost, the sheepshead minnow (*Cyprinodon variegatus*), *Marine Pollution Bulletin*, **129**, 231-240.

Chua E.M., Shimeta J., Nugegoda D., Morrison P.D., Clarke B.O. (2014). Assimilation of Polybrominated Diphenyl Ethers from Microplastics by the Marine Amphipod, *Allorchestes Compressa*, *Environmental Science & Technology*, **48**, 8127-8134.

Cincinelli A., Scopetani C., Chelazzi D., Lombardini E., Martellini T., Katsoyiannis A., Fossi M.C., Corsolini S. (2017). Microplastic in the surface waters of the Ross Sea (Antarctica): Occurrence, distribution and characterization by FTIR, *Chemosphere*, **175**, 391-400.

Cole M., Lindeque P., Halsband C., Galloway T.S. (2011). Microplastics as contaminants in the marine environment: A review, *Marine Pollution Bulletin*, **62**, 2588-2597.

Collicutt B., Juanes F., Dudas S.E. (2018). Microplastics in juvenile Chinook salmon and their nearshore environments on the east coast of Vancouver Island, *Environmental Pollution*, **244**, 135-142.

Costa L.L., Arueira V.F., da Costa M.F., Di Beneditto A.P.M., Zalmon I.R. (2019). Can the Atlantic ghost crab be a potential biomonitor of microplastic pollution of sandy beaches sediment? *Marine Pollution Bulletin*, **145**, 5-13.

Cózar A., Echevarría F., González-Gordillo J.I., Irigoien X., Úbeda B., Hernández-León S., Palma Á.T., Navarro S., García-de-Lomas J., Ruiz A., Fernández-de-Puelles M.L., Duarte C.M. (2014). Plastic debris in the open ocean, *Proceedings of the National Academy of Sciences*, **111**, 10239-10244.

Critchell K., Hoogenboom M.O. (2018). Effects of microplastic exposure on the body condition and behaviour of planktivorous reef fish *(Acanthochromis polyacanthus*), *PLOS ONE*, **13(3)**, e0193308.

Cunningham E.M., Rico S.N.A.K.E., Audh R.R., Burger J.M., Bornman T.G., Fawcett S.G.C.M.B., Osborne A.O., Woodall L.C. (2022). The transport and fate of microplastic fibres in the Antarctic: The role of multiple global processes, *Frontiers in Marine Science*, **9**, 2296-7745.

Daniel D.B., Ashraf P.M., Thomas S.N. (2020). Abundance characteristics and seasonal variation of microplastics in Indian white shrimps (*Fenneropenaeus indicus*) from coastal waters off Cochin, Kerala, India, *Science of The Total Environment*, **737**, 139839.

Dethier M.N., Hoins G., Kobelt J., Lowe A.T., Galloway A.W.E., Schram J.B., Raymore M., Duggins D.O. (2019). Feces as food: The nutritional value of urchin feces and implications for benthic food webs, *Journal of Experimental Marine Biology and Ecology*, **514-515**, 95-102.

Dong C.D., Chen C.W., Chen Y.C., Chen H.H., Lee J.S., Lin C.H. (2019). Polystyrene microplastic particles: In vitro pulmonary toxicity assessment, *Journal of Hazardous Materials*, **385**, 121575.

Evangeliou N., Grythe H., Klimont Z., Heyes C., Eckhardt S., Lopez-Aparicio S., Stohl A. (2020). Atmospheric transport is a major pathway of microplastics to remote regions, *Nature Communications*, **11**, 3381.

Farrell P., Nelson K. (2013). Trophic level transfer of microplastic: *Mytilus edulis* (L.) to *Carcinus maenas* (L.), *Environmental Pollution*, **177**, 1-3.

Feng Z., Wang R., Zhang T., Wang J., Huang W., Li J., Xu J., Gao G. (2020). Microplastics in specific tissues of wild sea urchins along the coastal areas of northern China, *Science of The Total Environment*, **728**, 138660.

Fred-Ahmadu O.H., Bhagwat G., Oluyoye I., Benson N.U., Ayejuyo O.O., Palanisami T. (2020). Interaction of chemical contaminants with microplastics: Principles and perspectives, *Science of The Total Environment*, **706**, 135978.

Gauquie J., Devriese L., Robbens J., De Witte B. (2015). A qualitative screening and quantitative measurement of organic contaminants on different types of marine plastic debris, *Chemosphere*, **138**, 348-356.

Goldstein M.C., Rosenberg M., Cheng L. (2012). Increased oceanic microplastic debris enhances oviposition in an endemic pelagic insect, *Biology letters*, **8**, 817-820.

Gorman D., Moreira F.T., Turra A., Fontenelle F.R., Combi T., Bícego M.C., de Castro Martins C. (2019). Organic contamination of beached plastic pellets in the South Atlantic: Risk assessments can benefit by considering spatial gradients, *Chemosphere*, **223**, 608-615.

Granby K., Rainieri S., Rasmussen R.R., Kotterman M.J.J., Sloth J.J., Cederberg T.L., Barranco A., Marques A., Larsen B.K. (2018). The influence of microplastics and halogenated contaminants in feed on toxicokinetics and gene expression in European seabass (*Dicentrarchus labrax*), *Environmental Research*, **164**, 430-443.

Green D.S. (2016). Effects of microplastics on European flat oysters, *Ostrea edulis* and their associated benthic communities, *Environmental Pollution*, **216**, 95-103.

Greene D.H.S., Selivonchick D.P. (1987). Lipid metabolism in fish, *Progress in Lipid Research*, **26**, 53-85.

Gu H., Wang S., Wang X., Yu X., Hu M., Huang W., Wang Y. (2020). Nanoplastics impair the intestinal health of the juvenile large yellow croaker *Larimichthys crocea*, *Journal of Hazardous Materials*, **397**, 122773.

Guzzetti E., Sureda A., Tejada S., Faggio C. (2018). Microplastic in Marine Organism: Environmental and Toxicological Effects, *Environmental Toxicology and Pharmacology*, **64**, 164-171.

Hämer J., Gutow L., Köhler A., Saborowski R. (2014). Fate of Microplastics in the Marine Isopod *Idotea emarginata*, *Environmental Science & Technology*, **48**, 13451-13458.

Hengstmann E., Tamminga M., vom Bruch C., Fischer E.K. (2018). Microplastic in beach sediments of the Isle of Rügen (Baltic Sea) - Implementing a novel glass elutriation column, *Marine Pollution Bulletin*, **126**, 263-274.

Hernandez-Milian G., Lusher A., MacGabban S., Rogan E. (2019). Microplastics in grey seal (*Halichoerus grypus*) intestines: Are they associated with parasite aggregations? *Marine Pollution Bulletin*, **146**, 349-354.

Hirai H., Takada H., Ogata Y., Yamashita R., Mizukawa K., Saha M., Kwan C., Moore C., Gray H., Laursen D., Zettler E.R., Farrington J.W., Reddy C.M., Peacock E.E., Ward M.W. (2011). Organic micropollutants in marine plastics debris from the open ocean and remote and urban beaches, *Marine Pollution Bulletin*, **62**, 1683-1692.

Horn D., Granek E.F., Steele C.L. (2019). Effects of environmentally relevant concentrations of microplastic fibers on Pacific mole crab (*Emerita analoga*) mortality and reproduction, *Limnology and Oceanography Letters*, **5**, 74-83.

Hu L., Chernick M., Lewis A.M., Ferguson P.L., Hinton D.E. (2020). Chronic microfiber exposure in adult Japanese medaka (*Oryzias latipes*), *PLOS ONE*, **15**, e0229962.

Ivar do Sul J.A., Spengler Â., Costa M.F. (2009). Here, there and everywhere. Small plastic fragments and pellets on beaches of Fernando de Noronha (Equatorial Western Atlantic), *Marine Pollution Bulletin*, **58**, 1236-1238.

Iwasaki S., Isobe A., Kako S., Uchida K., Tokai T. (2017). Fate of microplastics and mesoplastics carried by surface currents and wind waves: A numerical model approach in the Sea of Japan, *Marine Pollution Bulletin*, **121**, 85-96.

Jacques O., Prosser R.,S. (2021). A probabilistic risk assessment of microplastics in soil ecosystems, *Science of The Total Environment*, **757**, 143987.

Kang J.-H., Kwon O.Y., Lee K.-W., Song Y.K., Shim W.J. (2015). Marine neustonic microplastics around the southeastern coast of Korea, *Marine Pollution Bulletin*, **96**, 304-312.

Kanhai L.D.K., Gardfeldt K., Krumpen T., Thompson R,C., O'Connor I. (2020). Microplastics in sea ice and seawater beneath ice floes from the Arctic Ocean, *Scientific Reports*, **10**, 5004.

Kim D., Chae Y., An Y.-J. (2017). Mixture Toxicity of Nickel and Microplastics with Different Functional Groups on *Daphnia magna*, *Environmental Science & Technology*, **51**. 12852-12858.

Kim I.S., Chae D.H., Kim S.K., Choi S., Woo S.B. (2015). Factors Influencing the Spatial Variation of Microplastics on High-Tidal Coastal Beaches in Korea, *Archives of Environmental Contamination and Toxicology*, **69**, 299-309.

Kim S.K., Kang D.J., Kim K.R., Lee D.S. (2009). Distribution of Organochlorine Pesticides in Intertidal and Subtidal Sediments in Coastal Wetland with High Tidal Ranges, *Archives of Environmental Contamination and Toxicology*, **58**, 514-522.

Kim S.W., Chae Y., Kim D., An Y.J. (2018). Zebrafish can recognize microplastics as inedible materials: Quantitative evidence of ingestion behavior, *Science of The Total Environment*, **649**, 156-162.

Koelmans A.A., Besseling E., Wegner A., Foekema E.M. (2013a). Plastic as a Carrier of POPs to Aquatic Organisms: A Model Analysis, *Environmental Science & Technology*, **47**, 7812-7820.

Koelmans A.A., Besseling E., Wegner A., Foekema E.M, (2013b). Correction to Plastic As a Carrier of POPs to Aquatic Organisms: A Model Analysis, *Environmental Science & Technology*, **47**, 8992-8993.

Laist D.W. (1997). Impacts of Marine Debris: Entanglement of Marine Life in Marine Debris Including a Comprehensive List of Species with Entanglement and Ingestion Records, *Springer Series on Environmental Management*, 99-139.

Lattin G.L., Moore C.J., Zellers A.F., Moore S.L., Weisberg S.B. (2004). A comparison of neustonic plastic and zooplankton at different depths near the southern California shore, *Marine Pollution Bulletin*, **49**, 291-294.

Lee K.-W., Shim W.J., Kwon O.Y., Kang J.-H. (2013). Size-Dependent Effects of Micro Polystyrene Particles in the Marine Copepod *Tigriopus japonicus*, Environmental Science & *Technology*, **47**, 11278-11283.

Lelieveld J., Klingmüller K., Pozzer A., Pöschl U., Fnais M., Daiber A., Münzel T. (2019). Cardiovascular disease burden from ambient air pollution in Europe reassessed using novel hazard ratio functions, *European Heart Journal*, **40**, 1590-1596.

Li C., Zhu L., Wang X., Liu K., Li D. (2022). Cross-oceanic distribution and origin of microplastics in the subsurface water of the South China Sea and Eastern Indian Ocean, *Science of The Total Environment*, **805**, 150243.

Li F., Jia F., Tu H., Sun C., Li F. (2017). Environmental Behavior and Ecological Effects of Microplastics in the Ocean, *Asian Journal of Ecotoxicology*, **12**, 11-18 (in Chinese).

Li J., Chapman E.C., Shi H., Rotchell J.M. (2020a). PVC Does Not Influence Cadmium Uptake or Effects in the Mussel (*Mytilus edulis*), *Bulletin of Environmental Contamination and Toxicology*, **104**, 315-320.

Li Z., Zhou H., Liu Y., Zhan J., Li W., Yang K., Yi X. (2020b). Acute and chronic combined effect of polystyrene microplastics and dibutyl phthalate on the marine copepod *Tigriopus japonicus*, *Chemosphere*, **261**, 127711.

Liu M., Ding Y., Huang P., Zheng H., Wang W., Ke H., Chen F., Liu L., Cai M. (2021). Microplastics in the western Pacific and South China Sea: Spatial variations reveal the impact of Kuroshio intrusion, *Environmental Pollution*, **288**, 117745.

Lo H.S., Xu X., Wong C.Y., Cheung S.G. (2018). Comparisons of microplastic pollution between mudflats and sandy beaches in Hong Kong, *Environmental Pollution*, **236**, 208-217.

Long X., Fu T.M., Yang X., Tang Y., Zheng Y., Zhu L., Shen H., Ye J., Wang C., Wang T., Li B. (2022). Efficient Atmospheric Transport of Microplastics over Asia and Adjacent Oceans, *Environmental Science & Technology*, **56**, 6243-6252.

Lönnstedt O.M., Eklöv P. (2016). Environmentally relevant concentrations of microplastic particles influence larval fish ecology, *Science*, **352**, 1213-1216.

Lots F.A.E., Behrens P., Vijver M.G., Horton A.A., Bosker T. (2017). A large-scale investigation of microplastic contamination: Abundance and characteristics of microplastics in European beach sediment, *Marine Pollution Bulletin*, **123**, 219-226.

Lu K., Qiao R., An H., Zhang Y. (2018). Influence of microplastics on the accumulation and chronic toxic effects of cadmium in zebrafish (*Danio rerio*), *Chemosphere*, **202**, 514-520.

Luís L.G., Ferreira P., Fonte E., Oliveira M., Guilhermino L. (2015). Does the presence of microplastics influence the acute toxicity of chromium(VI) to early juveniles of the common goby (*Pomatoschistus microps*)? A study with juveniles from two wild estuarine populations, *Aquatic Toxicology*, **164**, 163-174.

Lusher A.L., Burke A., O'Connor I., Officer R. (2014). Microplastic pollution in the Northeast Atlantic Ocean: Validated and opportunistic sampling, *Marine Pollution Bulletin*, **88**, 325-333.

Lusher A.L., Tirelli V., O'Connor I., Officer R. (2015). Microplastics in Arctic polar waters: the first reported values of particles in surface and sub-surface samples, *Scientific Reports*, **5**, 14947.

Maršić-Lučić J., Lušić J., Tutman P., Bojanić Varezić D., Šiljić J., Pribudić J. (2018). Levels of trace metals on microplastic particles in beach sediments of the island of Vis, Adriatic Sea, Croatia, *Marine Pollution Bulletin*, **137**, 231-236.

Massos A., Turner A. (2017). Cadmium, lead and bromine in beached microplastics, *Environmental Pollution*, **227**, 139-145.

Mattsson K., Ekvall M.T., Hansson L.A., Linse S., Malmendal A., Cedervall T. (2014). Altered Behavior, Physiology, and Metabolism in Fish Exposed to Polystyrene Nanoparticles, *Environmental Science & Technology*, **49**, 553-561.

Mbedzi R., Dalu T., Wasserman R.J., Murungweni F., Cuthbert R.N. (2019). Functional response quantifies microplastic uptake by a widespread African fish species, *Science of The Total Environment*, **700**, 134522.

Mohsen M., Zhang L., Sun L., Lin C., Wang Q., Yang H. (2019). Microplastic fibers transfer from the water to the internal fluid of the sea cucumber *Apostichopus japonicus*, *Environmental Pollution*, **257**, 113606.

Moore C.J. (2008). Synthetic polymers in the marine environment: A rapidly increasing, long-term threat, *Environmental Research*, **108**, 131-139.

Moore C.J., Moore S.L., Leecaster M.K., Weisberg S.B. (2001). A Comparison of Plastic and Plankton in the North Pacific Central Gyre, *Marine Pollution Bulletin*, **42**, 1297-1300.

Moore R.C., Loseto L., Noel M., Etemadifar A., Brewster J.D., MacPhee S., Bendell L., Ross P.S. (2019). Microplastics in beluga whales (*Delphinapterus leucas*) from the Eastern Beaufort Sea, *Marine Pollution Bulletin*, **150**, 110723.

Munari C., Infantini V., Scoponi M., Rastelli E., Corinaldesi C., Mistri M. (2017). Microplastics in the sediments of Terra Nova Bay (Ross Sea, Antarctica), *Marine Pollution Bulletin*, **122**, 161-165.

Napper I.E., Thompson R.C. (2019). Environmental Deterioration of Biodegradable, Oxo-biodegradable, Compostable, and Conventional Plastic Carrier Bags in the Sea, Soil, and Open-Air Over a 3-Year Period, *Environmental Science & Technology*, **53**, 4775-4783.

Nobre C.R., Santana M.F.M., Maluf A., Cortez F.S., Cesar A., Pereira C.D.S., Turra A. (2015). Assessment of microplastic toxicity to embryonic development of the sea urchin *Lytechinus variegatus* (Echinodermata: Echinoidea), *Marine Pollution Bulletin*, **92**, 99-104.

Oliveira M., Ribeiro A., Hylland K., Guilhermino L. (2013). Single and combined effects of microplastics and pyrene on juveniles (0+ group) of the common goby *Pomatoschistus microps* (Teleostei, Gobiidae), *Ecological Indicators*, **34**, 641-647.

Ory N.C., Gallardo C., Lenz M., Thiel M. (2018). Capture, swallowing, and egestion of microplastics by a planktivorous juvenile fish, *Environmental Pollution*, **240**, 566-573.

Pedà C., Caccamo L., Fossi M.C., Gai F., Andaloro F., Genovese L., Perdichizzi A., Romeo T., Maricchiolo G. (2016). Intestinal alterations in European sea bass *Dicentrarchus labrax* (Linnaeus, 1758) exposed to microplastics: Preliminary results, *Environmental Pollution*, **212**, 251-256.

Peng X., Chen M., Chen S., Dasgupta S., Xu H., Ta K., Du M., Li J., Guo Z., Bai S. (2018). Microplastics contaminate the deepest part of the world's ocean, *Geochemical Perspectives Letters*, **9**, 1-5.

Plee T.A., Pomory C.M. (2020). Microplastics in sandy environments in the Florida Keys and the panhandle of Florida, and the ingestion by sea cucumbers (Echinodermata: Holothuroidea) and sand dollars (Echinodermata: Echinoidea), *Marine Pollution Bulletin*, **158**, 111437.

Pratte Z.A., Besson M., Hollman R.D., Stewart F.J. (2018). The Gills of Reef Fish Support a Distinct Microbiome Influenced by Host-Specific Factors, *Applied and Environmental Microbiology*, **84**, e00063-18.

Ragusa A., Svelato A., Santacroce C., Catalano P., Notarstefano V., Carnevali O., Papa F., Rongioletti M.C.A., Baiocco F., Draghi S., D'Amore E., Rinaldo D., Matta M., Giorgini E., (2021). Plasticenta: First evidence of microplastics in human placenta, *Environment International*, **146**, 106274.

Rands M.R.W., Adams W.M., Bennun L., Butchart S.H.M., Clements A., Coomes D., Entwistle A., Hodge I., Kapos V., Scharlemann J.P.W., Sutherland W.J., Vira B. (2010). Biodiversity Conservation: Challenges Beyond 2010, *Science*, **329**, 1298-1303.

Reeve M.R., Walter M.A. (1977). Observations on the existence of lower threshold and upper critical food concentrations for the copepod *Acartia tonsa* Dana, *Journal of Experimental Marine Biology and Ecology*, **29**, 211-221.

Rist S., Hartmann N.B. (2017). Aquatic Ecotoxicity of Microplastics and Nanoplastics: Lessons Learned from Engineered Nanomaterials, *Freshwater Microplastics*, **58**, 25-49.

Rist S.E., Assidqi K., Zamani N.P., Appel D., Perschke M., Huhn M., Lenz M. (2016). Suspended micro-sized PVC particles impair the performance and decrease survival in the Asian green mussel *Perna viridis*, *Marine Pollution Bulletin*, **111**, 213-220.

Rochman C.M. (2013a). Plastics and Priority Pollutants: A Multiple Stressor in Aquatic Habitats, *Environmental Science & Technology*, **47**, 2439-2440.

Rochman C.M., Browne M.A., Halpern B.S., Hentschel B.T., Hoh E., Karapanagioti H.K., Rios-Mendoza L.M., Takada H., Teh S., Thompson R.C. (2013b). Policy: Classify plastic waste as hazardous, *Nature*, **494**, 169-171.

Rochman C.M., Kurobe T., Flores I., Teh S.J. (2014). Early warning signs of endocrine disruption in adult fish from the ingestion of polyethylene with and without sorbed chemical pollutants from the marine environment, *Science of The Total Environment*, **493**, 656-661.

Roman M.R. (2000). *The Biology of Calanoid Copepods* J. Mauchline, *The Quarterly Review of Biology*, **75(2)**, 198-198.

Ryan P.G., Bouwman H., Moloney C.L., Yuyama M., Takada H. (2012). Long-term decreases in persistent organic pollutants in South African coastal waters detected from beached polyethylene pellets, *Marine Pollution Bulletin*, **64**, 2756-2760.

Santos D., Luzio A., Matos C., Bellas J., Monteiro S.M., Félix L. (2021). Microplastics alone or co-exposed with copper induce neurotoxicity and behavioral alterations on zebrafish larvae after a subchronic exposure, *Aquatic Toxicology*, **235**, 105814.

Setälä O., Fleming-Lehtinen V., Lehtiniemi M. (2014). Ingestion and transfer of microplastics in the planktonic food web, *Environmental Pollution*, **185**, 77-83.

Siegfried M., Koelmans A.A., Besseling E., Kroeze C. (2017). Export of microplastics from land to sea. A modelling approach, *Water Research*, **127**, 249-257.

Sleight V.A., Bakir A., Thompson R.C., Henry T.B. (2017). Assessment of microplastic-sorbed contaminant bioavailability through analysis of biomarker gene expression in larval zebrafish, *Marine Pollution Bulletin*, **116**, 291-297.

Steinmetz Z., Wollmann C., Schaefer M., Buchmann C., David J., Tröger J., Muñoz K., Frör O., Schaumann G.E. (2016). Plastic mulching in agriculture. Trading short-term agronomic benefits for long-term soil degradation? *Science of The Total Environment*, **550**, 690-705.

Sui Q., Zhang L., Xia B., Chen B., Sun X., Zhu L., Wang R., Qu K. (2020). Spatiotemporal distribution, source identification and inventory of microplastics in surface sediments from Sanggou Bay, China, *Science of The Total Environment*, **723**, 138064.

Sun T., Wang S., Ji C., Li F., Wu H. (2022). Microplastics aggravate the bioaccumulation and toxicity of coexisting contaminants in aquatic organisms: A synergistic health hazard, *Journal of Hazardous Materials*, **424**, 127533.

Sundt P., Schulze P.E., Syversen F. (2014). Sources of Microplastic Pollution to the Marine Environment, *Mepex for the Norwegian Environment Agency* (Miljødirektoratet 86).

Sussarellu R., Suquet M., Thomas Y., Lambert C., Fabioux C., Pernet M.E.J., Goïc N.L., Quillien V., Mingant C., Epelboin Y., Corporeau C., Guyomarch J., Robbens J., Paul-Pont I., Soudant P., Huvet A. (2016). Oyster reproduction is affected by exposure to polystyrene microplastics, *Proceedings of the National Academy of Sciences*, **113**, 2430-2435.

Taha Z.D., Md Amin R., Anuar S.T., Nasser A.A.A., Sohaimi E.S. (2021). Microplastics in seawater and zooplankton: A case study from Terengganu estuary and offshore waters, Malaysia, *Science of The Total Environment*, **786**, 147466.

Tang Y., Rong J., Guan X., Zha S., Shi W., Han Y., Han Y., Du X., Wu F., Huang W., Liu G., (2019). Immunotoxicity of microplastics and two persistent organic pollutants alone or in combination to a bivalve species, *Environmental Pollution*, **258**, 113845.

Taniguchi S., Colabuono F.I., Dias P.S., Oliveira R., Fisner M., Turra A., Izar G.M., Abessa D.M.S., Saha M., Hosoda J., Yamashita R., Takada H., Lourenço R.A., Magalhães C.A., Bícego M.C., Montone R.C. (2016). Spatial variability in persistent organic pollutants and polycyclic aromatic hydrocarbons found in beach-stranded pellets along the coast of the state of São Paulo, southeastern Brazil, *Marine Pollution Bulletin*, **106**, 87-94.

Telesh I.V., Khlebovich V.V. (2010). Principal processes within the estuarine salinity gradient: A review, *Marine Pollution Bulletin*, **61**, 149-155.

Thompson R.C., Olsen Y., Mitchell R.P., Davis A., Rowland S.J., John A.W.G., McGonigle D., Russell A.E. (2004). Lost at Sea: Where Is All the Plastic? *Science*, **304**, 838-838.

United Nations Environmental Programme. (2014). UNEP Year Book 2014: emerging issues in our global environment.

United Nations Environmental Programme. (2021). From Pollution to Solution: a global assessment of marine litter and plastic pollution.

Van Cauwenberghe L., Claessens M., Vandegehuchte M.B., Janssen C.R. (2015). Microplastics are taken up by mussels (*Mytilus edulis*) and lugworms (*Arenicola marina*) living in natural habitats, *Environmental Pollution*, **199**, 10-17.

Van Cauwenberghe L., Vanreusel A., Mees J., Janssen C.R. (2013). Microplastic pollution in deep-sea sediments, *Environmental Pollution*, **182**, 495-499.

Van Sebille E., England M.H., Froyland G. (2012). Origin dynamics and evolution of ocean garbage patches from observed surface drifters, *Environmental Research Letters*, **7**, 044040.

Wang J., Tan Z., Peng J., Qiu Q., Li M. (2016). The behaviors of microplastics in the marine environment, *Marain Environmental Research*, **113**, 7-17.

Wang Q., Zhu X., Hou C., Wu Y., Teng J., Zhang C., Tan H., Shan E., Zhang W., Zhao J. (2021). Microplastic uptake in commercial fishes from the Bohai Sea, China, *Chemosphere*, **263**, 127962.

Wang T., Wang L., Chen Q., Kalogerakis N., Ji R., Ma Y. (2020a). Interactions between microplastics and organic pollutants: Effects on toxicity, bioaccumulation, degradation, and transport, *Science of The Total Environment*, **748**, 142427.

Wang T., Zou X., Li B., Yao Y., Li J., Hui H., Yu W., Wang C. (2018). Microplastics in a wind farm area: A case study at the Rudong Offshore Wind Farm, Yellow Sea, China, *Marine Pollution Bulletin*, **128**, 466-474.

Wang X., Liu L., Zheng H., Wang M., Fu Y., Luo X., Li F., Wang Z. (2019). Polystyrene microplastics impaired the feeding and swimming behavior of mysid shrimp *Neomysis japonica*, *Marine Pollution Bulletin*, **150**, 110660.

Wang X., Zheng H., Zhao J., Luo X., Wang Z., Xing B. (2020b). Photodegradation Elevated the Toxicity of Polystyrene

Microplastics to Grouper (*Epinephelus moara*) through Disrupting Hepatic Lipid Homeostasis, *Environmental Science & Technology*, **54**, 6202-6212.

Watts A.J.R., Lewis C., Goodhead R.M., Beckett S.J., Moger J., Tyler C.R., Galloway T.S. (2014). Uptake and Retention of Microplastics by the Shore Crab *Carcinus maenas*, *Environmental Science & Technology*, **48**, 8823-8830.

Watts A.J.R., Urbina M.A., Goodhead R., Moger J., Lewis C., Galloway T.S. (2016). Effect of Microplastic on the Gills of the Shore Crab *Carcinus maenas*, *Environmental Science & Technology*, **50**, 5364-5369.

Wegner A., Besseling E., Foekema E.M., Kamermans P., Koelmans A.A. (2012). Effects of nanopolystyrene on the feeding behavior of the blue mussel (*Mytilus edulis L.*), *Environmental Toxicology and Chemistry*, **31**, 2490-2497.

Wen B., Jin S.R., Chen Z.Z., Gao J.Z., Liu Y.N., Liu J.H., Feng X.S. (2018). Single and combined effects of microplastics and cadmium on the cadmium accumulation, antioxidant defence and innate immunity of the discus fish (*Symphysodon aequifasciatus*), *Environmental Pollution*, **243**, 462-471.

Woodall L.C., Sanchez-Vidal A., Canals M., Paterson G.L.J., Coppock R., Sleight V., Calafat A., Rogers A.D., Narayanaswamy B.E., Thompson R.C. (2014). The deep sea is a major sink for microplastic debris, *Royal Society Open Science*, **1**, 140317.

Wright S.L., Thompson R.C., Galloway T.S. (2013). The physical impacts of microplastics on marine organisms: A review, *Environmental Pollution*, **178**, 483-492.

Zhang W., Ma X., Zhang Z., Wang Y., Wang J., Wang J., Ma D. (2015). Persistent organic pollutants carried on plastic resin pellets from two beaches in China, *Marine Pollution Bulletin*, **99**, 28-34.

Zhang W., Zhang S., Wang J., Wang Y., Mu J., Wang P., Lin X., Ma D. (2017). Microplastic pollution in the surface waters of the Bohai Sea, China, *Environmental Pollution*, **231**, 541-548.

Zheng Y., Li J., Cao W., Liu X., Jiang F., Ding J., Yin X., Sun C. (2019). Distribution characteristics of microplastics in the seawater and sediment: A case study in Jiaozhou Bay, China, *Science of The Total Environment*, **674**, 27-35.

Zhu J., Zhang Q., Li Y., Tan S., Kang Z., Yu X., Lan W., Cai L., Wang J., Shi H. (2018a). Microplastic pollution in the Maowei Sea, a typical mariculture bay of China, *Science of The Total Environment*, **658**, 62-68.

Zhu L., Bai H., Chen B., Sun X., Qu K., Xia B. (2018b). Microplastic pollution in North Yellow Sea, China: Observations on occurrence, distribution and identification, *Science of The Total Environment*, **636**, 20-29.

Supplementary materials

Table S1. Microplastic contamination in some marine animals.

Animal Category	Location	Abundance (items/ind)	Shapes	References
Zooplankton	Terengganu Estuary, Malaysia	0.01 ± 0.002–0.20 ± 0.14	Fibers, fragments, pellets	(Lusher *et al.* 2014)
Pampus argenteus	Bohai Sea, China	0.89 ± 0.77	Fibers, fragments, pellets, films	(Wang *et al.* 2021)
Argyrosomus argentatus		2.11 ± 2.36		
Delphinapterus leucas	Eastern Beaufort Sea, Canada	97 ± 42	Fibers, fragments	(Moore *et al.* 2019)
Halichoerus grypus	South coast, Ireland	27.9 ± 14.7	Fibers, fragments, film	(Hernandez-Milian *et al.* 2019)
Fenneropenaeus indicus	Cochin, India	0.39 ± 0.6	Fibers, others	(Daniel *et al.* 2020)
Ocypode quadrata	Grussaí Beach Arch, Brazil	1–158	Fibers, fragments, foam	(Costa *et al.* 2019)
Holothuria mexicana	Florida Keys, USA	>1	Fibers, fragments	(Plee and Pomory, 2020)
Echinoidea	Coastal areas, China	2.20 ± 1.50–10.04 ± 8.46	Fibers, fragments, film	(Feng *et al.* 2020)

Table S2. Coexistence of pollutants and MPs in some marine areas of the world.

Location	Type of MPs	Organic pollutant/Heavy metal type and concentration	Reference
Central Pacific Gyre	PP, PE	PAHs (12–868 ng/g)	(Hirai *et al.* 2011)
Pacific Ocean		PAHs (112 ng/g)	
Caribbean Sea		PAHs (88–105 ng/g)	
Costa Rica, beach		DDTs (0.6–124.4 ng/g)	
Vietnam, beach		DDTs (11–118 ng/g)	
Tokyo, urban beach		DDTs (0.2–52 ng/g)	
Los Angeles, urban beach		DDTs (2.2–8.4 ng/g)	
Kanagawa, urban beach		DDTs (n.d–76 ng/g)	
Paraná State, southern coastline	/	PAHs (1,454–6,020 ng/g)	(Gorman *et al.* 2019)
		PCBs (0.8–104.6 ng/g)	
São Paulo state, coastline state	/	PCBs (3.41–7554 ng/g)	(Taniguchi *et al.* 2016)
South African, beach	PE, PP, other	PCBs (25–61 ng/g)	(Ryan *et al.* 2012)
		HCHs (2–5 ng/g)	
		DDTs (8–31 ng/g)	
Portuguese, coastline	/	PCBs (2–223 ng/g)	(Antunes *et al.* 2013)
Yellow Sea, beach	Plastic resin pellet	PAHs (136.3–1586.9 ng/g)	(Zhang *et al.* 2015)
Yellow Sea, beach		PCBs (34.7–213.7 ng/g)	

Bohai Sea, beach		PCBs (21.5–323.2 ng/g)	
Belgian, coast	PE, PS	PAHs (1076–3007 ng/g)	(Gauquie *et al.* 2015)
Australian, North Freemantle beach	PE, PP, PS, PET	Se (12.74 µg/kg), As (89.58 µg/kg), Cd (94.73 µg/kg), Cr (165.38 µg/kg), Cu (1.16 mg/kg), Ba (22.35 mg/kg), Pb (0.24 mg/kg), Zn (15.12 mg/kg), Mn (1.25 mg/kg)	(Carbery *et al.* 2020)
Australian, Redhead beach		Se (6.14 µg/kg), As (204.85 µg/kg), Cd (24.33 µg/kg), Cr (208.27 µg/kg), Cu (0.28 mg/kg), Ba (5.23 mg/kg), Pb (0.57 mg/kg), Zn (2.27 mg/kg), Mn (0.54 mg/kg)	
Vis, Croatia, Milna beach	/	Cd (2.90 ng/g), Cr (0.21 µg/g), Cu (0.21 µg/g), Fe (40.3 µg/g), Mn (1.78 µg/g), Ni (0.14 µg/g), Pb(0.26µg/g), Zn (2.08 µg/g)	(Maršić-Lučić *et al.* 2018)
England, beach	PE, PP, PS, PVC	Cd (30–50 µg/g), Pb (5–20 µg/g)	(Massos and Turner, 2018)