

REVIEW

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Marine microplastics as vectors of major ocean pollutants and its hazards to the marine ecosystem and humans

Tan Suet May Amelia¹, Wan Mohd Afiq Wan Mohd Khalik^{1,2}, Meng Chuan Ong^{1,3}, Yi Ta Shao^{4,5*}, Hui-Juan Pan^{6,7*} and Kesaven Bhubalan^{1,2,3,8,9*} 

Abstract

Microplastic pollutes water, land, air, and groundwater environments not only visually but also ecologically for plants, animals, and humans. Microplastic has been reported to act as vectors by sorbing pollutants and contributing to the bioaccumulation of pollutants, particularly in marine ecosystems, organisms, and subsequently food webs. The inevitable exposure of microplastic to humans emphasises the need to review the potential effects, exposure pathways, and toxicity of microplastic toward human health. Therefore, this review was aimed to reveal the risks of pollutant sorption and bioaccumulation by microplastic toward humans, as well as the dominant types of pollutants sorbed by microplastic, and the types of pollutants that are bioaccumulated by microplastic in the living organisms of the marine ecosystem. The possible factors influencing the sorption and bioaccumulation of pollutants by microplastic in marine ecosystems were also reviewed. The review also revealed the prevailing types of microplastic, abundance of microplastic, and geographical distribution of microplastic in the aquatic environment globally. The literature review revealed that microplastic characteristics, chemical interactions, and water properties played a role in the sorption of pollutants by microplastic. The evidence of microplastic posing a direct medical threat to humans is still lacking albeit substantial literature has reported the health hazards of microplastic-associated monomers, additives, and pollutants. This review recommends future research on the existing knowledge gaps in microplastic research, which include the toxicity of microplastic, particularly to humans, as well as the factors influencing the sorption and bioaccumulation of pollutants by microplastic.

Keywords: Adsorption, Bioaccumulation, Contaminant, Microplastic, Pollutant, Vector

1 Introduction

The plastic industry began in the 1920s and grew rapidly since the 1940s. In 2014, the global plastic output has been 20 times that of 1964 (Neufeld et al. 2016). Globally, the annual production of 330 and 360 million metric tons was recorded for 2016 and 2018, respectively

(PlasticsEurope 2017; PlasticsEurope 2019). The annual plastic production is still increasing despite the increased awareness of plastic pollution and efforts to mitigate its pollution. An estimated 275 million metric tons of land-based plastic waste from 192 coastal countries resulted in 4.8 to 12.7 million metric tons entering the ocean in 2010 (Jambeck et al. 2015). Moreover, the degradation and the fragmentation of marine plastic litter lead to the formation of hazardous secondary microplastic in the ocean. In terms of microplastic waste, 60 to 99 million metric tons was generated in 2015 (Lebreton and Andrady 2019). In terms of stray waste alone, roughly six and three million metric tons of macroplastic and

* Correspondence: itshao@mail.ntou.edu.tw; hjpan_siraya@mail.ntou.edu.tw; kesaven@umt.edu.my

⁴Institute of Marine Biology, National Taiwan Ocean University, Keelung 20224, Taiwan

⁶Institute of Earth Sciences, College of Ocean Science and Resource, National Taiwan Ocean University, Keelung 20224, Taiwan

¹Faculty of Science and Marine Environment, Universiti Malaysia Terengganu, 21030 Kuala Nerus, Terengganu, Malaysia

Full list of author information is available at the end of the article



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microplastic, respectively, were lost to the environment in 2015 (Ryberg et al. 2019). The microplastic pollution reported in the literature chiefly refers to the visible or observable presence of microplastic in the food sources, the above-significant levels of microplastic concentrations in the ecosystems, and the risks of microplastic to the environment and public health (Abbasi et al. 2018; Akarsu et al. 2020; Alimi et al. 2018). The presence of most manmade substances in the environment, though inevitable, requires critical attention when their over-presence presents as pollution, with threats and negative implications. In order to challenge the problem of microplastic pollution, it is imperative to recognise the source, transportation, degradation, sink, and consequences of microplastic pollution.

Microplastic is a ubiquitous occurrence in terrestrial, freshwater, and marine environments (Yu et al. 2019a). The sources of microplastic include wastewater treatment plants, landfills, irrigation, agricultural fields, industrial effluent, and domestic runoff (Conley et al. 2019; Corradini et al. 2019; Gündoğdu et al. 2018; He et al. 2019; Li et al. 2018a; Rochman 2018; Steinmetz et al. 2016). Microplastic is released from atmospheric deposition, land-based sources, fertilisers, artificial turf, road and air transportation, textiles, coastal and tourism activities, commercial fishing, marine vessels, aquaculture, and oil rigs (Belzagui et al. 2019; Gieré et al. 2018; Kole et al. 2017; Liu et al. 2019b; Li et al. 2018a; Weithmann et al. 2018). Subsequently, microplastic accumulates in the deep sea, pristine polar regions, ice sheet, living organisms, and contaminants such as polycyclic aromatic hydrocarbons (PAH), heavy metals, and antibiotics (Obbard 2018; Woodall et al. 2014; Yu et al. 2018). Microplastic waste is further fragmented into nanoplastic and then accumulatively ingested by living organisms, affecting their feeding, digestion, excretion, and reproduction processes.

To curb microplastic pollution, it is important to reach an understanding of not only the source of microplastic but also its transportation, degradation, and the possible solutions of microplastic pollution. The complex transportation and distribution processes of microplastic include the ocean dynamics (i.e. surface drifting, vertical mixing, beaching, settling, and entrainment) and the physical characteristics of microplastic (i.e. size, shape, and density) (Enders et al. 2015; Guo et al. 2020; Kanhai et al. 2018; Li et al. 2020; Woodall et al. 2014). Ocean dynamics have also caused large areas of surface convergence, naturally accumulating up to 580,000 plastic pieces per square kilometre, such as the garbage patches near the Kuroshio Extension and in the western North Atlantic Ocean and Caribbean Sea (Howell et al. 2012; Law et al. 2010; Maximenko et al. 2012; Yamashita and Tanimura 2007). Organic aggregates or microorganisms,

such as diatoms, can rapidly accumulate on the surface of plastic debris and form a biofilm, which then increases the density and causes the sinking of the floating or suspended low-density microplastic, hence redistributing the latter (Galgani et al. 2015; Zhang 2017; Zhao et al. 2018; Zhao et al. 2017).

Furthermore, marine microplastic can be ingested and introduced to the biological systems of a wide range of organisms from herbivores and secondary consumers to the predators of higher trophic levels, such as microorganisms, planktons, benthic invertebrates, (Naidu et al. 2018), fish (Savoca et al. 2019), deep ocean biota (Courtenes-Jones et al. 2017), and larger mammals (Barboza et al. 2018a; Besseling et al. 2015; Wang et al. 2020b), causing neurotoxicity and genotoxicity, as well as reduced feeding, filtration, survival, and reproductive abilities (Zhang et al. 2019a). These effects decrease the quantity and quality of the food supply to humans and other aquatic organisms (Wong et al. 2020). Moreover, microorganisms including pathogens were discovered to be associated with microplastic (Brandon et al. 2018; Syranidou et al. 2017). Among the bacteria groups detected in microplastic were *Bacillus* sp., *Paenibacillus* sp., Actinobacteria, and Firmicutes (Lwanga et al. 2018; Park and Kim 2019). Although microplastic can persist in the environment and resist degradation, some microplastic-associated bacteria can degrade microplastic (Alimi et al. 2018; Kim et al. 2017b; Sen and Raut 2015; Yuan et al. 2020). Different bacteria consortia have different microplastic degradation abilities depending on the types of bacteria and enzymes (Tsiota et al. 2018). Reports have also shown that bacteria not exclusively associated with plastic have been colonising microplastic, which included the families of Rhodobacteraceae, Hyphomonadaceae, and Sphingomonadaceae (Mata et al. 2017; Moura et al. 2018; Oberbeckmann et al. 2018).

The use of microplastic-degrading microbes can enhance the biodegradation of marine microplastic already subjected to weathering and external physicochemical factors (Oberbeckmann and Labrenz 2020; Qi et al. 2017). Although microplastic is difficult to biodegrade, it offers support for microbial colonisation and growth (Rujnic-Sokele and Pilipovic 2017). Pores and irregularities that were observed on the surface of microplastic, such as polypropylene (PP), polyethylene (PE), polyethylene terephthalate (PET), polystyrene (PS), and polylactic acid (PLA), showed adhesion, colonisation, and damage by the associated bacterial and fungal strains, for example *Aspergillus flavus* (Auta et al. 2018; Kim et al. 2017a; Mohan et al. 2016; Paco et al. 2017; Devi et al. 2015; Uscategui et al. 2016; Yoshida et al. 2016). Additionally, worms have been reported to degrade plastics (Yang et al. 2015b). Although microplastic can be biodegraded gradually, ageing primary or secondary

microplastic is constantly structurally altered or fragmented through biological, mechanical, or chemical degradation in the environment into nanoplastic, which further increases its bioavailability to living organisms (Hernandez et al. 2017; Liu et al. 2019a; Wang et al. 2019a). Furthermore, microplastic is a vector of harmful pollutants such as persistent organic pollutants (POP) and heavy metals, capable of transporting contaminants to the ecosystem via the food chain (Kwon et al. 2017; Wang et al. 2020a; Zhang et al. 2020a). Microplastic can hence increase the bioavailability of pollutants to ecosystems and organisms through sorption and bioaccumulation (Guzzetti et al. 2018; Horton et al. 2017).

The strategies and solutions to mitigate microplastic pollution include source control, remediation, clean up, regional involvement, and research (Agamuthu et al. 2019; Ma et al. 2020b; Wu et al. 2017). For example, the European Union has banned primary microplastic and disposable plastics and encourages the controlled release of marine litter. The United Nations has proposed individual measures that can help to lighten microplastic pollution, such as recycling and consumption control (Barceló and Picó 2020). Non-governmental organisations and international and national authorities, such as Surfisher Foundation Europe and United Nations Environment Program Mediterranean Action Plan, were also involved in the practice or implementation of circular economy systems, and coastal and marine debris management (Agamuthu et al. 2019; Camins et al. 2020). Other efforts to alleviate microplastic pollution also include research on the application of water treatment systems and microbial degradation or biosynthesis of plastic-like material (Amelia et al. 2019; Bolto and Xie 2019; Hu et al. 2019; Ngo et al. 2019; Raju et al. 2018; Sun et al. 2019; Talvitie et al. 2017; Zhang et al. 2020a; Zhang and Chen 2020). Additionally, the fundamental research on plastic cannot be ignored as an in-depth understanding of microplastic characterisation based on its polymer type and morphology is needed to identify the most to the least environmentally harmful types of microplastic.

Overall, systematic management of and innovative research on reducing plastic waste materials at the source, removing microplastic in wastewater treatment facilities or increasing the use of bioplastic or easily biodegradable plastic, is needed to solve the issue of microplastic contamination. However, the understanding of the severity and impact of microplastic is important to mitigate microplastic pollution. With the increasing quantity of literature concerning the abundance and ubiquitous distribution of microplastic globally, research attention has been shifted toward the direct and indirect consequences of microplastic. Therefore, this review mainly focused on the possible factors that influenced pollutant sorption or

bioaccumulation by microplastic, the health risks of microplastic to humans and other living organisms of the marine ecosystem, and the abundance and distribution of microplastic.

2 Review

2.1 Types, abundance, and distribution of microplastic

Microplastic can be categorised on the basis of its original manufactured size, morphology, density, and material. Microplastic is defined as a plastic material with an average size of less than 5 mm. The fragmentation of larger plastic in the environment is caused by chemical (e.g. photolysis, hydrolysis, and thermal), mechanical (e.g. abrasion), and biological (e.g. bacteria and fungi) degradation (Andrady 2011; Ivleva et al. 2017). The classification of microplastic is subjected to two categories, namely primary microplastic (i.e. microplastic produced in micro-sized particles such as microbeads) and secondary microplastic (i.e. microplastic from the breakdown of larger plastic materials) (Duis and Coors 2016). Primary microplastic is originally manufactured as microplastic that is usually applied in cosmeceutical products, drug vectors, and engineering or industrial applications (Auta et al. 2017). In contrast, secondary microplastic is originally manufactured as relatively large plastic that is degraded and fragmented into smaller pieces by complex weathering interactions (Rocha-Santos and Duarte 2015). Based on the morphotypes or types of morphologies, microplastic consists of fibres, fragments, films, pellets, beads, and Styrofoam. The abundance of microplastic in a water column depends on its density, wherein low-density microplastic such as PE and PP is buoyant in the water, while high-density microplastic tends to sink in the sediment (Sul and Costa 2014).

2.2 Fibres

Numerous research work has generally revealed that the heterogeneity of microplastic abundance has been mostly influenced by human activities such as wastewater discharge, industrial discharge, mariculture, and settlement. For example, research work in the Yangtze estuary and coastal area of the East China Sea revealed fibre as the dominant morphotype collected up to 79.1% and 83.2%, respectively (Zhao et al. 2014). The influx from the river to the sea is believed to play a significant role in introducing microplastic to the estuary and coastal areas. Anthropogenic activities such as land-based and fisheries activities trigger the abundance of microplastic.

For Xiangshan Bay, China, the source of microplastic pollution is dominated by the presence of mariculture activities. Chen et al. (2018) revealed more than 50% of microplastic of the fibre type derived from mariculture activities. Seawater samples were collected with plankton

net tows involving nine sites. Microplastic derived from PE, PP, and PS was detected during the period of study. At another mariculture site in Maowei Sea, China, a similar pattern of a majority of fibre made from the PET, PP, and PE polymers was observed (Zhu et al. 2019). The riverine influx nearby thus enhanced the abundance of microplastic in the Maowei Sea.

A comprehensive study was carried out by Dai et al. (2018) to understand the impact of human intervention on microplastic pollution in Bohai Sea waters. In all, 75% of the microplastic collected by the researchers fell under the morphotype of fibre. The fraction of fibres in the surface layer was lower than that in the deeper levels of the water column. Sample collection using a stainless-steel bucket was believed to have easily trapped fibre as compared to other methods such as those using mantraweb nets or neustonic trawls. In the surface water of the North Yellow Sea, film and fibre were the dominant morphotypes reported by Zhu et al. (2018b). Microplastic abundance was high with an average value of 545 ± 282 items/m³. PE was identified as a major contributor of microplastic composition in surface water. The Chinese coastal current and the Korean coastal current were speculated to introduce floating microplastic into the North Yellow Sea. Domestic sewage and fishing activities were the main sources of plastic pollution in this area.

Microplastic abundance and distribution research has been focused on regions that are highly influenced by human activities, such as the closed ecosystem of Jiaozhou Bay, China. The morphotype of black or blue fibre is the main characteristic of microplastic found in the collected samples. Polyamide (PA), polyvinyl acetate (PVA), PP, PE, and PET were the main groups of polymers found in this work. The Haibo river estuary was discovered to have strong riverine input to enhance microplastic flux into the bay. In addition, residual current and sediment mobility were other factors with a significant impact on the microplastic distribution and composition (Zheng et al. 2019).

Microplastic pollution was also observed in the west region of Asia. For example, a low abundance of microplastic was recorded in Qatar marine waters with an average concentration of 0.71 particle/m³. The morphotypes of fibre and granule were spotted as the dominant types in the study (Castillo et al. 2016). The source of pollution was speculated to be derived from human anthropogenic activities, including oil-rig installation and ship operations. In the Arabic region, Aliabad et al. (2019) conducted a study on the microplastic abundance in Chabahar Bay, Oman, involving surface water from the seven selected stations. The primary colours of the microplastics collected were white, blue, and red. A low concentration of microplastic abundance was recorded in this area with fibre as the dominant morphotype,

made up of secondary microplastic. More than 60% of the analysed samples were detected to be PE and PP.

In the region of the Mediterranean Sea, the first study on microplastic abundance was reported by Kazour et al. (2019), which focused on a Lebanese coastal area, namely the Levantine Basin. This area was recognised as a polluted area with a high abundance of microplastic. More than 50% of the microplastics detected in the water samples were fibres, and PE was the most abundant material. On the other side of the Mediterranean Sea, Akarsu et al. (2020) explicated the abundance of microplastic in Mersin Bay, Turkey. The study area was reported to receive high discharge from wastewater treatment plants. Therefore, the fibre material type was expected to contribute a large portion of microplastic.

American and European continents also recorded varied microplastic abundance in the water column. For example, Sutton et al. (2016) found the occurrence of microplastic in San Francisco Bay, CA, USA. Fibres were dominant in the sample containing wastewater discharge. Furthermore, average microplastic abundance (4.5 particles/m³) was found in Tampa Bay, Florida, in the water and sediment samples. In all, 88% of the microplastic detected in the water samples was fibres, followed by fragments and flakes (McEachern et al. 2019). The source of microplastic introduced into the aquatic ecosystem was believed to be wastewater treatment discharge. High microplastic abundance was found in the vicinity of the industrial area.

Studies on microplastic abundance in Swedish marine waters included Skagerrak/Kattegat, the Baltic Sea, and Gulf of Bothnia. Two different approaches, namely manta trawl and in situ pump filtering, were used to collect the microplastic from the seawater. The pump filtering method detected a higher microplastic concentration, 0.10 particles/m³, than the manta trawl method, 0.04 particles/m³. Fibre was the most frequently found morphotype in the water samples consisting of PE and PP (Schönlau et al. 2020). This study only took into account fibres longer than 1 mm, and the transparent ones were excluded. Most of the synthetic fibres found in all the samples were black or blue, which might suggest ropes as a possible source of these fibres, as these colours are very popular for ropes and fishing gear in the studied area.

Microplastic was also detected in South Africa marine waters. Nel and Froneman (2015) reported a quantitative microplastic concentration in the water column ranging from 257 to 1215 particles/m³ and the dominant proportion of the morphotype of fibre. The main source was believed to be land-based sewage effluent. Overall, the microplastic typically found to pollute the environment was generally of the morphotype of fibre or fragment (Fig. 1), PE, or PP material, and contributed by

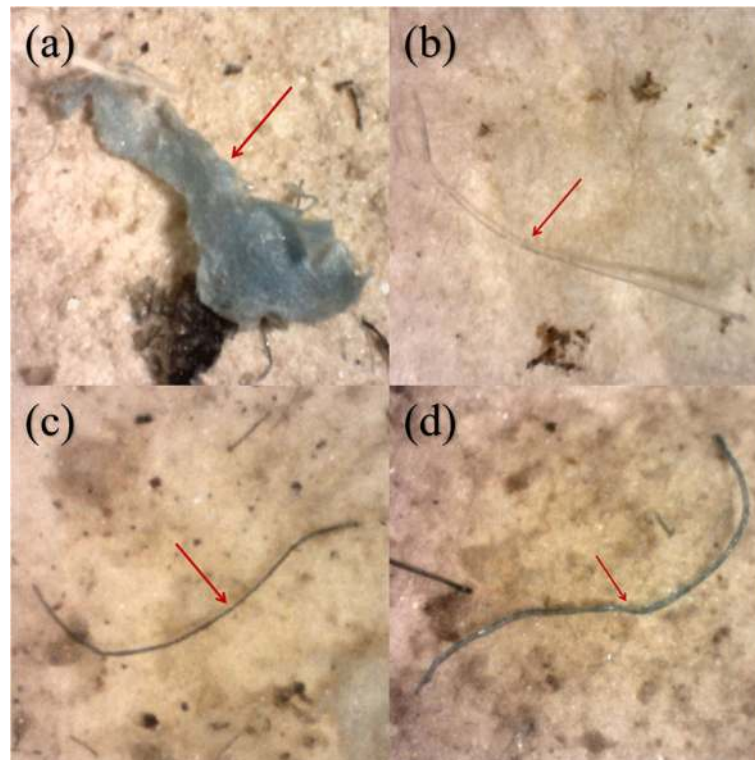


Fig. 1 Examples of microplastic morphotypes found in water samples. **a** Blue fragment, **b** transparent fibre, **c** black fibre, and **d** blue fibre

anthropogenic, urban, fishery, or marine activities. The dominant colours were blue and black. Fibres were either tangled or untangled, as well as straight or curved.

In the Oceania region, low microplastic abundance in the water, sediment, and fish samples was recorded in the coastal area of Suva, Republic of Fiji (Ferreira et al. 2020). The microplastic in the seawater samples was obtained using planktonic circular tows. The microplastic in the water samples of Laucala Bay, Suva Harbour, and *vanua* Navakavu was less than 0.24 item/m³. MPs were categorised as fibres (60.2% ± 6.9%) and fragments (26.9% ± 3.5%), whereas the lowest percentage was found to be films (9.4% ± 2.0%) and microbeads (3.5% ± 1.7%). The microplastic material analysis revealed 15 polymer types, with PE, PP, latex, and nylon as the major contributors. Interestingly, the source appointment of microplastic pollution was believed to be derived from anthropogenic activities despite the marine protected area, *vanua* Navakavu, being the most affected area.

2.3 Fragments

An analysis of Bohai Sea waters revealed that low-density polymers, such as PP, PE, and PS, were commonly found from all the 11 stations, where manta net was towed horizontally at the surface for 15 min at approximately 1.5 to 2.0 knots for each transect (Table 1). The most abundant morphotype was fragment in this

work, which accounted for 46% of the total plastic collected. The source of microplastic was linked to fishing activities and marine farming in the Bohai Sea (Zhang et al. 2017).

The first observation in the Yellow Sea revealed a low abundance of microplastic dominated with the morphotype of fragment by 42%. The main polymers identified using micro-Fourier-transform infrared spectroscopy (micro-FTIR) were PP and PE, accounting for 55.93% and 32.20% of the total microplastic, respectively. High concentrations were observed close to coastal cities, thus indicating that the Yellow Sea coastal current played a significant role in the microplastic distribution (Sun et al. 2018).

The heterogeneity of microplastic was also found in Malaysian marine waters. Research work by Khalik et al. (2018) explicated two geological regions with different human activities, which exhibited a contradicting pattern of microplastic abundance. The first region located in Kuantan, Malaysia, is a port, while the second location situated in Kuala Nerus, Terengganu, is a non-urban area. Kuala Nerus focused on commercial fishing and tourism, while Kuantan port is one of the major multi-cargo ports in Malaysia. Kuala Nerus had higher microplastic abundance, 0.13–0.69 pieces/L, than Kuantan port with 0.14–0.15 pieces/L. The uniform microplastic abundance was believed to be caused by a closed system,

Table 1 Heterogeneity of MPs abundances in different region worldwide

Region	Location	MPs concentration range	Polymer	Reference
Asian	Yangtze Estuary, China	500–10,200 n/m ³	Not Available	Zhao et al. (2014)
	Yellow Sea, China	ND–0.81 item/m ³	PP, PE	Sun et al. (2018)
	Jiaozhou Bay, China	20–120 item/m ³	PP, PE, PET	Zheng et al. (2019)
	Bohai Sea, China	0.01–1.5 particle/m ³	PE, PP, PS, PET	Zhang et al. (2017)
	Xiangshan Bay, China	4.6–20.1 item/m ³	PE, PP, PS	Chen et al. (2018)
	Qatar marine waters	0–3 item/m ³	PE, PP, PS, PA, PMMA	Castillo et al. (2016)
	Chabahar Bay, Oman	0.07–1.14 particle/m ³	PE, PP, PS, PET, PVA	Aliabad et al. (2019)
	Levantine Basin, Lebanon	0.17–0.62 item/m ³	PE, PP, PS, PA, PET, PUR, PVC	Kazour et al. (2019)
	Mersin Bay, Turkey	0.2–5.1 pcs/L	PE, PP, PS	Akarsu et al. (2020)
	North Coast of Surabaya, Indonesia	0.38–0.61 N/L	PS, PE, PP, PU, PET, PB, PES	Cordova et al. (2019)
	Kuala Nerus and Kuantan Port, Malaysia	0.13–0.69 pcs/L	PES, PE, PP, PS, PA, PVC	Khalik et al. (2018)
	Southern Sri Lanka	0–29 item/m ³	PE, PP, PS and Mix PE+PP	Koongolla et al. (2018)
	America	San Francisco Bay, USA	15,000–2,000,000 particle/km ²	Not available
Tampa Bay, USA		1.2–18.1 particles/m ³	Not available	McEachern et al. (2019)
Europe	Sweden marine waters	0–70.3 item/m ³	PE, PP	Schönlaue et al. (2020)
	Tuscan coastal waters, Italy	0.16–0.27 item/m ³	PE, PP, EVA, SBR	Baini et al. (2018)
Africa	King Harbour, Jamaica	0–5.73 particle/m ³	PE, PP	Rose and Webber (2019)
	South Africa marine waters	257–1215 particle/m ³	Not available	Nel and Froneman (2015)

such as a port, that limited the intrusion of plastic pollutants from the open sea. Six polymers were identified from both the locations, namely PET, PS, PA, PVC, PP, and PE.

Saliu et al. (2018) proved the occurrence of microplastic in Faafu Atoll, Maldives. Surface seawater was taken using neuston net involving 12 locations. This area had low population density and influx from tourism activities. However, the average microplastic abundance in Faafu Atoll waters was 0.32 ± 0.15 particles/m³. Microplastics of the fragment morphotype found in a high fraction contributed up to 52% of the total particles in the analyses.

In addition, Savoca et al. (2019) discussed the occurrence of microplastic in Tyrrhenian waters, the Mediterranean Sea. Water samples were taken using a Bongo net with a diameter of 60 cm. The microplastic found in the seawater was dominated by the fragment type and originated from polyvinyl chloride (PVC) and low-density polyethylene (LDPE) materials. The blue colour of the plastic fragments was the most dominant colour, reaching 56.25%. In South Asia, the abundance of microplastic varied between counties. A preliminary work carried out in Southern Sri Lanka by Koongolla et al. (2018) revealed that 70% of the collected water samples contained microplastic. The morphotype of fragment was dominant, and the microplastic composition was largely contributed by PE and PP. Countries with tourism-driven economies, such as the Maldives, also experienced microplastic pollution in the water column.

In Spanish marine waters, Ruiz-Orejón et al. (2018) reported the occurrence of microplastic in the coastal waters of the Balearic Islands. Water samples were taken using a mantra trawl net. More than 90% of the plastic materials collected were identified as the morphotype of fragment with a dominant size range of 0.5 to 1.3 mm. Plastic with a micro size showed a stronger prevalence than that with the meso and macro sizes. A high density of microplastic was recorded in the water column, close to the coastal line, particularly in the first 10 km. Italian marine waters also experienced microplastic pollution as reported by Baini et al. (2018) in the Tuscan coastal waters. Surface samples were collected using a manta trawl, involving four transects located along the Tuscan coast. The average microplastic abundance was low, 0.26 item/m³, with PE and PP identified as the main source of microplastic in this area. Fragments were the most prevalent type (62%) found in the water samples, followed by filaments (29%); the highest abundance was of particles of the size of < 1 mm (52%).

For the African region, microplastic abundance was also reported at different locations. For example, microplastic abundance was studied in King Harbour, Jamaica. Samples were taken using a mantra trawl net. The microplastic concentrations varied from 0 to 5.73 particles/m³. The morphotype of fragment was the most frequently found from the collected samples with sizes ranging from 1 to 2.5 mm. Microplastics have also been characterised by a wide variety of colours, most of which are transparent, accounting for 35%. The FTIR analysis

indicated the PE and PP composition for the morphotype of fragment (Rose and Webber 2019).

Saeed et al. (2020) conducted the first survey to understand the occurrence of microplastic in Kuwait marine waters. Their findings revealed lower microplastic abundance in the water samples than the other matrices such as sediment and fish. Neuston nets were used to trap the microplastics from the seawater samples. In all, 37 microplastics were found in only 15 stations out of the 44 sampling stations, where each trawl was approximately 1 km. The PP and PE polymers were identified using Raman spectroscopy. Six of the microplastics were described as the fragment morphotype, while the remaining were labelled as the filament morphotype.

2.4 Other types

In the South China Sea region, researchers reported microplastic abundance in the water samples of Nansha Island, China. The microplastic concentration analysed from 15 water samples was high at 1733 item/m³. Microbead was reported to be the dominant morphotype and accounted for 76.5% of all the detected microplastic (Nie et al. 2019). However, in Hong Kong marine waters, Tsang et al. (2017) found a relatively low density of microplastic, such as PP and PE. The morphotype of pellet was identified as the main fraction of the microplastic, accounting for 96.8%. Discharge from a nearby river, Pearl River or untreated sewage, storm runoff, and other land-based activities were believed to contribute as the sources of the plastic pollution.

In Southeast Asia, a similar trend was reported by many researchers, for example, the microplastic abundance in the north coast of Surabaya. Microplastic concentrations ranged from 0.38 to 0.61 item/L, in three selected areas namely the Lamong Bay, Kenjeran Beach, and Wonorejo coastal area. A high contribution was derived from the close proximity to human settlements and shipping activities. The foam type was reported to be the dominant morphotype, which accounted for 58.4% of all the detected microplastic. Seven polymers were identified from the studied location, and polystyrene was the most frequently detected in the samples (Cordova et al. 2019).

2.5 Pollutant sorption of microplastic and its factors

With the global severity of microplastic pollution, at least 80 countries have publications about microplastic sorbing pollutants, among which the major countries are China, the USA, the UK, Italy, and South Korea (Yu et al. 2019a). Research on microplastic sorption has been increasing as microplastic is gaining attention as a vector or carrier of harmful contaminants, such as POP and heavy metals (Brennecke et al. 2016; Kwon et al. 2017).

Recently, in the field of microplastic, the terms 'absorb' and 'sorb' have been used in addition to the more commonly used 'adsorb' to distinctly mean adsorption, absorption, both adsorption and absorption, or either one of the former two situations (Huffer et al. 2018; Katsnelson 2015; Munier and Bendell 2018; Xia et al. 2020). According to [Dictionary.com](https://www.dictionary.com) and Oxford University Press, 'sorb' is defined as 'to gather on a surface either by absorption, or adsorption, or a combination of the two processes' ("Sorb" 2020). The terms 'adsorb' and 'absorb' are defined as 'hold molecules as a thin film on the outside surface or on internal surfaces within the material' and 'take in or soak up substance by chemical or physical action', respectively ("Adsorb" 2020; "Absorb" 2020). In the recent publications in the microplastic field, 'adsorb' and 'absorb' are referred to as 'sorbed onto the surface of microplastic' and 'sorbed into the bulk phase of microplastic', respectively (Endo et al. 2008; Huffer et al. 2018; Katsnelson 2015). The distinction of surface adsorption and bulk absorption could shed new insights behind the mechanism of pollutant sorption by microplastics, the behaviour of microplastic-pollutant complexes, or the complete vector capacity of multiple microplastic types (Huffer et al. 2018; Katsnelson 2015). For example, certain substances such as minerals and rigid carbonaceous geosorbents (charcoal and soot) tend to adsorb compounds, whereas organic contaminants tend to absorb into organic matter (Endo et al. 2008).

Although the biological or mechanical degradation of microplastic can lead to its fragmentation, increased surface area, enhanced bioavailability, and subsequently, its sorption of pollutants, other chemical degradation and ultraviolet (UV) light can also change the functional groups and other chemical properties of microplastic, and hence, more directly aggravate the sorption of pollutants by microplastic (Göpferich 1995; Hernandez et al. 2017; Horton et al. 2016; Jun et al. 2014; Liu et al. 2019a; Ter et al. 2016; Wang et al. 2019a). Microplastic can sorb environmental contaminants that include hydrophobic organic pollutants (i.e. PAH, polychlorinated biphenyl [PCB]), hydrophilic organic pollutants (i.e. perfluoroalkyl acids [PFAA]), drugs, and metals (Table 2). The factors known to affect the pollutant sorption capacity of microplastic are the colour, density, age, and chemical properties of the microplastic, type of pollutant, biofilm presence, and environmental conditions such as dissolved organic matter, pH, and salinity (Fisner et al. 2017; Frias et al. 2013; Fries and Zarfl 2012; Wang et al. 2015).

2.6 Physical properties of microplastic as a factor

The physical and chemical properties of microplastic have been proven to affect its pollutant sorption capacity (Fisner et al. 2017; Fries and Zarfl 2012; Huffer et al.

Table 2 Types of pollutants carried by different types of microplastics

Microplastic	Pollutant	Reference
PA	Benzene derivative	Rehse et al. (2018)
PBAT	Heavy metals	Kedzierski et al. (2018)
PE	Lubrication oil	Haghi and Banaee (2017)
	PAH	Oliveira et al. (2013)
	Paraquat	Rochman et al. (2013b)
	PBDE	Llorca et al. (2018)
	PCB	Wang et al. (2015)
	Per- and polyfluoroalkyl substances	Fisner et al. (2017)
	PFOSA	Zhan et al. (2016)
	PPCP	Hu et al. (2017)
	Triclosan	Frydkjær et al. (2017)
PET	Heavy metals	Rochman et al. (2013a)
PP	Heavy metals	Rochman et al. (2013b)
	PAH	Fisner et al. (2017)
PS	Antibiotic	Wen et al. (2018)
	Cadmium	Zhang et al. (2019b)
	Lubrication oil	Llorca et al. (2018)
	PCB	Wang et al. (2015)
	Per- and polyfluoroalkyl substances	Guo et al. (2018)
	PFOSA	Zhan et al. (2016)
	Roxithromycin	Hu et al. (2017)
PVC	17 α -Ethinylestradiol	Qu et al. (2018)
	Antibiotic	Sleight et al. (2017)
	Benzene derivatives	Kedzierski et al. (2018)
	Heavy metals	Rochman et al. (2013a)
	Odesmethyvelafaxine	Pascall et al. (2005)
	PCB	Wu et al. (2016)
	PFOSA	Guo et al. (2018)
	Phenanthrene	
	Venlafaxine	
LDPE	Heavy metals	Rochman et al. (2013a)
HDPE	Heavy metals	Holmes et al. (2014)

PBAT polybutylene adipate terephthalate, *HDPE* high-density polyethylene, *PFOSA* perfluorooctanesulfonamide, *PPCP* pharmaceuticals personal care product

2018; Llorca et al. 2018; Ma et al. 2020a). It has been reported that the microplastic of a lighter colour contained PAH of lower molecular weight and lower concentrations of PAH or PCB than the microplastic of a darker colour (Antunes et al. 2013; Fisner et al. 2017). The hypothesised relationship between colour and chemical composition against microplastic sorption can encourage research for alternative colour pigments that are environmentally friendly (Ma et al. 2020a). Furthermore, microplastics of higher density were reported to contain lower concentrations of PAH, PCB, and phenanthrene (Fries and Zarfl 2012; Karapanagioti and Klontza 2008; Mato et al. 2001).

The ageing of microplastic cannot be ignored because weathered microplastic not only increases in surface area but also generates oxygen groups, porosity, roughness, charge, and polarity (Fotopoulou and Karapanagioti 2012). Hydrogen bonds formed among the oxygen-containing functional groups of aged microplastic can substantially increase the sorption capacity (Huffer et al. 2018; Liu et al. 2018). In an experiment by Kedzierski et al. with 502 day long-term exposure, Ni and Cu were found on PVC. Si, Al, Fe, and Mg were found on polybutylene adipate terephthalate (PBAT) after 1 year, while no change was observed for PET over the entire period (Kedzierski et al. 2018). Moreover, the effect of

microplastic ageing on pollutant sorption capacity varies in the past reports. According to Ma et al. weathered or aged microplastic was reported to possess less pollutant sorption capacity (Ma et al. 2020a). However, aged microplastic was also reported to show higher pollutant sorption capacity than virgin microplastic (Huffer et al. 2018; Liu et al. 2018; Yu et al. 2019a). The colour, density, and surface morphology or weathering or microplastic play a role synergistically at influencing the sorption ability of a microplastic particle, and the sequence of factors mentioned does not reflect the importance of a factor over the other. Mixed findings from the past literature suggest that the effect of microplastic ageing on pollutant sorption capacity depends on the type of pollutant or microplastic, and the gradual changes in their chemical bonds, forces, or interactions over time.

2.7 Chemical properties of microplastic as a factor

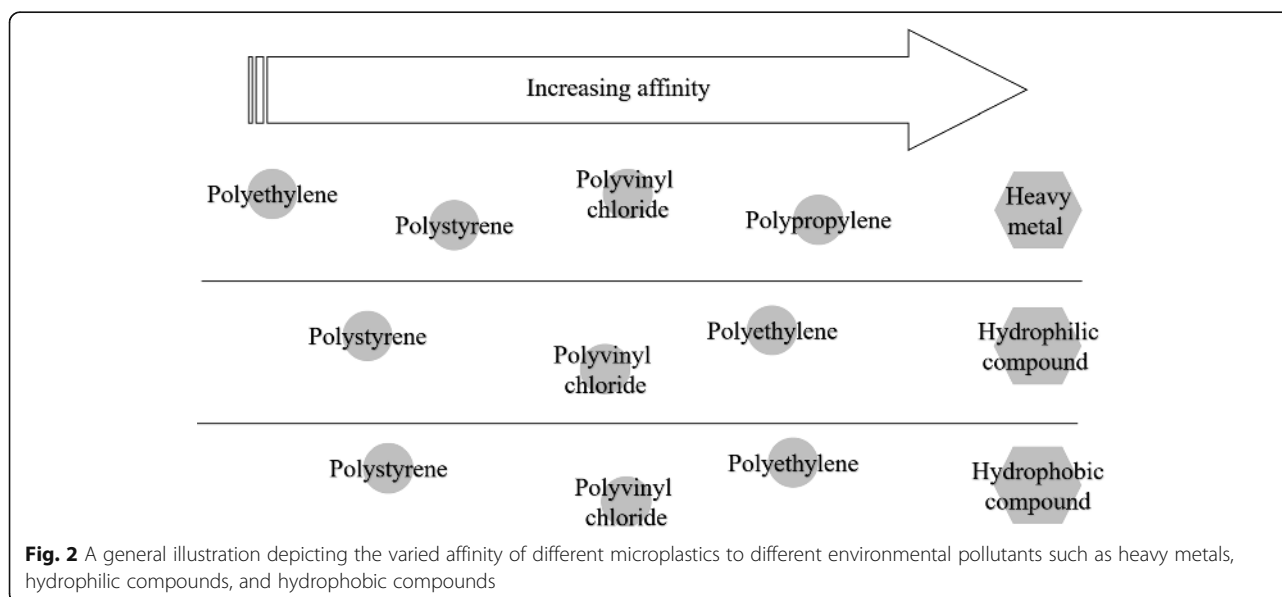
Besides physical properties, the chemical properties of microplastic and pollutants, such as composition, synthesis method, crystallinity, stability, polarity, functional group, and chemical bonding, also influence the sorption capacity of microplastic (Li et al. 2018b; Llorca et al. 2018; Wang et al. 2015). A chemical structure with a large surface area or porosity can lead to higher sorption capacity. For example, the sorption capacity of PE was the highest, but the sorption rate of PET was the highest. The former's larger surface area, greater gap between polymer chains, and free volume led to relatively higher sorption capacity and diffusion of the pollutant into PE (Karapanagioti and Klontza 2008; Rochman et al. 2013a; Teuten et al. 2007). In contrast, the faster sorption rate on PET could be attributed to its smaller surface area and glassy polymeric structure, which does not facilitate

diffusion into the material as compared to PE that absorbs more pollutants into structural gaps within the polymer (Pascall et al. 2005; Teuten et al. 2007).

The affinity of pollutants with microplastic varies as it depends on chemical interactions, including electrostatic interaction, van der Waals force, hydrophobic interaction, π - π interaction, or hydrogen bonding (Ma et al. 2020a) (Fig. 2). For example, the hydrogen bonding between PA and antibiotics caused strong affinity because of the amide groups (Li et al. 2018b). In another example, between carboxyl-functionalised polystyrene (PS-COOH) and amine-modified polystyrene (PS-NH₂) with different surface charges, the latter caused embryonic apoptosis, cell death, oxidative stress, and cell membrane disruption on the embryos of sea urchin *Paracentrotus lividus* (Vega and Epel 2004).

Hydrophobic POP are currently among the most researched microplastic-associated pollutants because of their high affinity and hydrophobicity-driven sorption with microplastic (Liu et al. 2019d; Rios et al. 2007). Hydrophobic organic pollutants include PCB, PAH, and other endocrine-disrupting chemicals (EDC) or benzene derivatives (Ma et al. 2017; Ma et al. 2018; Velzeboer et al. 2014). For example, the strong interaction between microplastic and hydrophobic organic PAH in the aquatic environment has been reported ubiquitously (Tan et al. 2019). In terms of crystallinity, hydrophobic organic pollutants typically favoured amorphous to crystalline structures, as organic contaminants harboured higher affinities for rubber plastics than glass plastics (Guo et al. 2012; Rochman et al. 2013a; Wu et al. 2001).

Hydrophilic POP, represented by perfluorooctanesulfonate (PFOS) and perfluorooctanesulfonamide



(PFOSA), interact dominantly with microplastic via partitioning rather than electrostatic force and hydrophobicity. The PFOS is mainly anionic, whereas PFOSA is non-ionic because of the sulphonamide functional group; PE is thus more attracted to PFOSA because of the presence of the functional group (Wang et al. 2015). Furthermore, PE, PS, PP, PA, and PVC microplastics were found to sorb hydrophilic antibiotics, such as trimethoprim (TMP), ciprofloxacin hydrochloride (CIP HCl), tetracycline (TC), amoxicillin (AMX), and sulfadiazine (SDZ) (Li et al. 2018b; Shen et al. 2018).

Different types of microplastics possess varied pollutant sorption and affinity as their dissimilar structure generates different chemical interactions, such as hydrophobicity or hydrophilicity. For example, non-polar PFOSA was more attracted to the non-polar PE via hydrophobic interactions but less attracted to the benzene-containing PS caused by decreased free volume, steric hindrance, and bond rotation (Wang et al. 2015). However, Yu et al. reported that PS can generate planarity, hydrophobic, and π - π interactions, whereas PE can generate only van der Waals interactions (Yu et al. 2019b). Contradictions were also reported regarding the type of plastic as a sorption factor, where no difference in PAH sorption was observed in PP and PE (Fisner et al. 2017). In-depth investigation is required to discover additional trends and associations of physical or chemical properties that are responsible for the sorption capacity of a microplastic with a specific pollutant.

Furthermore, a stronger hydrophobic interaction causes microplastic to attract benzene derivatives rather than single benzene (Huffer and Hofmann 2016). The PVC microplastic was reported to attract five bisphenol analogues by hydrophobic, π - π , covalent hydrogen, and halogen interactions. Electrostatic forces were present but are relatively easier to be influenced by external environmental factors (Wu et al. 2016). In contrast, PCB was reported to sorb more onto PE (highest uptake diffusion and partition coefficients) than PS and PVC. However, the sorption capacity with PVC reduced at higher chlorinated congeners because of the increasing cohesive density and molar volume of PCB (Pascall et al. 2005).

Additionally, the affinity or sorption capacity of microplastic and metals was neither temporally nor spatially associated (Yu et al. 2019b). High-density polyethylene (HDPE) was reported to have relatively lower affinity to metals than PET, PVC, LDPE, and PP (Rochman et al. 2013a). Sorption between metals and virgin or beached microplastic was rapid and significant. Virgin PE, which showed less affinity, possessed charged or polar surfaces, charged contaminants or additives, and nonspecific interactions between the metal organic complexes and the hydrophobic surface (Holmes et al. 2012). Beached or

aged PE, which showed higher affinity to heavy metals, contained not only chemical precipitation and oxygen-containing groups but also biofilms, which may contribute to the relatively high partition coefficient (Brennecke et al. 2016; Mato et al. 2001; Turner and Holmes 2015). Overall, as the attraction of different pollutants to the respective plastics depends on the ever-changing environmental conditions and physicochemical properties of both the substances, the difficulty of selecting the 'best' type of plastic could likely encourage future research attention on the development of an inert material.

2.8 Water properties as a factor

Environmental conditions and water properties, such as dissolved organic matter, pH, and salinity, have been reported to affect the pollutant sorption capacity of microplastic. The role of dissolved and particulate organic matter in microplastic sorption requires attention, as it can alter the in situ sorption capacity and distribution of marine microplastic (Besseling et al. 2016; Koelmans et al. 2009; Lambert et al. 2013). According to Wu et al. dissolved organic matter can influence the sorption process by (1) interacting with pollutants via complexation or hydrophobic interactions, (2) competition with other sorbents, and (3) subsequent interaction with pollutants after sorption onto other sorbents (Wu et al. 2016). The dissolved organic matter reduced the sorption capacity of PE for three pharmaceuticals and personal care products (PPCP), namely triclosan (TCS), 17 α -ethinyl estradiol (EE2), and 4-methylbenzylidene camphor (4MBC), because of the higher affinity between pharmaceuticals and dissolved organic matter. However, dissolved organic matter did not reduce the sorption of PE microplastic for carbamazepine (CBZ) (Wu et al. 2016).

Investigating the impact of pH on microplastic sorption is important, as the marine ecosystem has been experiencing acidification. As reported by Guo et al. the sorption capacity of PS and PVC for antibiotic tylosin (TYL) increases as pH decreases, because the proportion of TYL⁺ is larger, while the sorption capacity of PP and PE has minimal changes (Guo et al. 2018). The sorption of PS and PE for PFOS also increases as pH decreases, which protonates the microplastic surface and produces more anionic PFOS. Additionally, the PE surface can be easier to protonate than the PS surface (Wang et al. 2015). However, when the pH decreases, the sorption capacity of HDPE for Pb, Ni, Co, and Cd decreases, whereas for Cr and Cu, the sorption capacity increases and remains constant, respectively (Holmes et al. 2012; Holmes et al. 2014). Overall, although the pH of the surrounding water has been shown to influence the pollutant sorption by microplastic, such as through protonation (Guo et al. 2018), the literature regarding

the impact of pH against pollutant sorption by microplastic exhibits a need for a broader comparative or statistical approach to achieve insights into the associative patterns of pH with microplastic and pollutants.

Salinity has both positive and negative effects on the sorption behaviour. Increased salinity has been reported to increase sorption capacities (Velzeboer et al. 2014; Wang et al. 2015), such as elevated PCB or lubrication oil in PE and PS (Hu et al. 2017; Zhan et al. 2016). However, increased salinity was reported to decrease sorption capacities, namely in dichlorodiphenyltrichloroethane (DDT) and ciprofloxacin because of the competition for the sorption site (Bakir et al. 2014; Li et al. 2018b; Liu et al. 2018). Additionally, the effect of salinity was not obvious for phenanthrene (Bakir et al. 2014). Nonetheless, a fixed pattern was not present in terms of the salinity effect against metal sorption by microplastic. Increased salinity reduced the sorption capacity of Cd, Co, and Ni while increasing that of Cr, but interestingly kept that of Cu and Pb constant. It is apparent from the literature that salinity influences sorption capacity by principally disturbing the free ions and the competition for sorption sites (Holmes et al. 2014).

In summary, the literature review suggested that chemical interactions could be a significant factor of pollutant sorption by microplastic, as the chemical bonds on a microplastic is not only associated with the physical properties of the polymer (i.e. material, colour, and age) but also the properties of the surrounding medium (i.e. pH and salinity). In other words, the underlying force that allows the other physical properties of the microplastic and the surrounding medium to affect its sorbability is the chemical bonding. For example, the influence of microplastic age and material against the sorption capacity of a pollutant was eventually still because of the differences in the surface functional groups, polarity, or chemical forces such as hydrophobicity over time or across various pollutant types (Huffer and Hofmann 2016; Liu et al. 2018). Research in the direction of investigating the chemical interactions between microplastic, pollutants, and the environment can disclose new indirect factors of pollutant sorption by microplastic.

2.9 Hazards of microplastic to the marine ecosystem and its organisms

The widely known impacts of large plastic debris are the loss of aesthetic impression, disrupted tourism- and marine-related economy (e.g. aquaculture, energy production, fishing, and shipping), and the death and injury of marine wildlife (Lusher et al. 2015). In contrast, although both macroplastic and microplastic can also transport pathogens and release plasticizers or additives, microplastic can sorb and desorb and, hence,

bioaccumulate quantitatively more toxic pollutants because of the larger ratio of the surface area to the volume of the microplastic (Avio et al. 2017; Zettler et al. 2013; Teuten et al. 2009).

The existence of microplastic has been found in the guts of benthic invertebrates (Naidu et al. 2018; Nakki et al. 2017), fish (Savoca et al. 2019), deep ocean biota (Courtene-Jones et al. 2017), and larger mammals (Besseling et al. 2015) from different trophic levels. Ingested microplastic is transferred throughout the food web, driving increasing concerns about the threats to aquatic biota (Betts 2008; Wright et al. 2013a). Microplastic can accumulate and pose hazards in marine ecosystems and organisms (Guzzetti et al. 2018; Horton et al. 2017). Prior to understanding the threats of microplastic against marine organisms, it is imperative that we grasp the factors that affect the bioavailability of microplastic, as it indirectly affects the extent of dangers that the organisms are subjected to. The factors of microplastic bioavailability toward organisms are the size, colour, density, morphology, and mobility of microplastic, as well as the species, morphology, and physiology of organisms (e.g. ingestion likelihood, egestion rate, and translocation potential) (Baldwin 1995; Boerger et al. 2010; Cheung et al. 2018; Kokalj et al. 2018; Ory et al. 2017; Peters et al. 2017). Additionally, as each type of organism consumes microplastic differently because of the respective organism physiology and morphology, the representative model organisms selected in microplastic toxicological studies should be specifically based on the research objectives (Ma et al. 2020a). Nonetheless, the existing research has accumulatively confirmed that the types of organisms and the physicochemical traits of microplastic influence the bioavailability of microplastic to organisms, by altering the distribution and availability of microplastic as food for organisms, which influences the microplastic-associated hazards or extent of damage faced by different species (Setala et al. 2016).

Microplastic causes dangers to marine ecosystems and organisms through several approaches. In a typical marine ecosystem, microplastic acts as a reservoir or vector, thus accumulating chemical contaminants, transporting them over long distances, and leading to the high bioavailability of pollutants toward organisms when consumed (Avio et al. 2017; Brennecke et al. 2016; Caruso 2019; Liao and Yang 2020) (Fig. 3). Microplastic by itself without pollutants also poses various indirect and direct hazards to diverse organisms, which include translocation, physiological stress, energy budget alteration, abnormal metabolism, immune response, behavioural alteration, fecundity, egestion inefficiency, severe intestinal damage, and mortality (Besseling et al. 2015; Betts 2008; Courtene-Jones et al. 2017; Gambardella et al. 2017; Greven et al. 2016; Jeong et al. 2016; Jovanovic

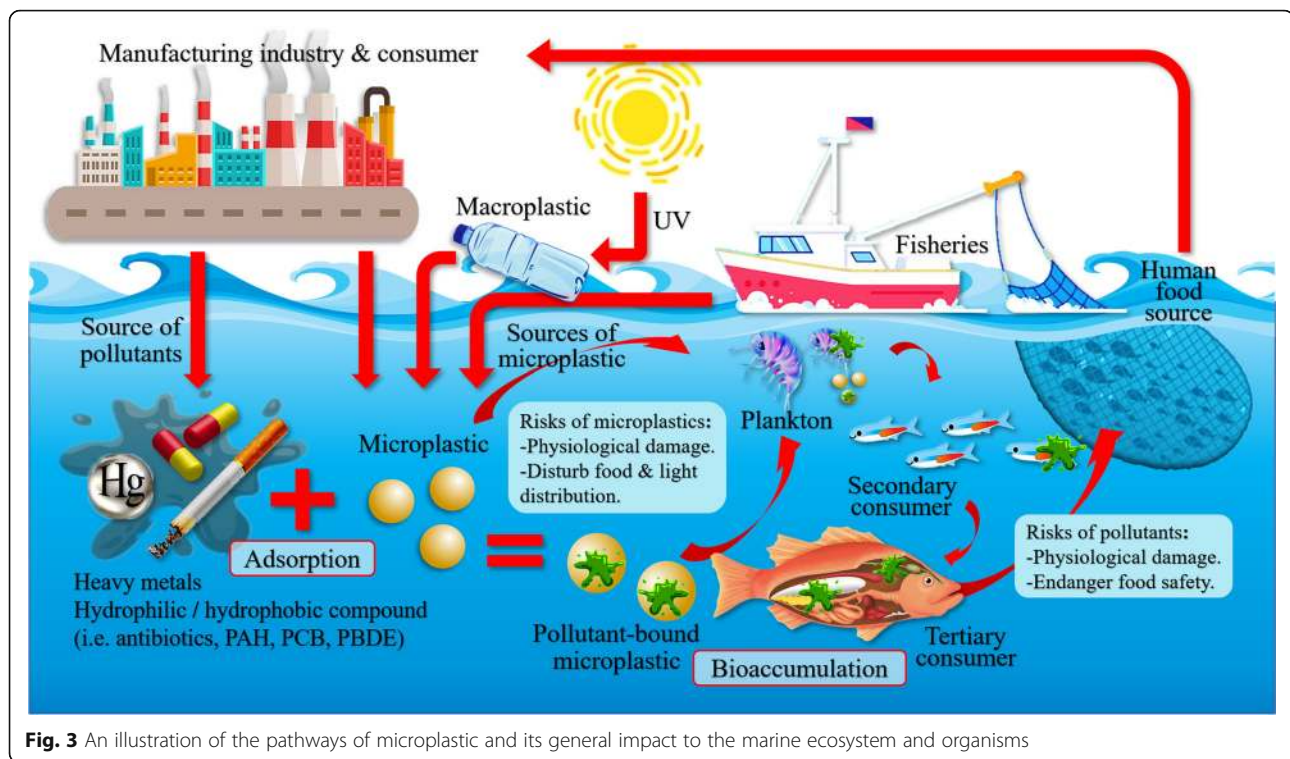


Fig. 3 An illustration of the pathways of microplastic and its general impact to the marine ecosystem and organisms

2017; Lei et al. 2018; Leroueil et al. 2008; Naidu et al. 2018; Nakki et al. 2017; Prokic et al. 2019; Savoca et al. 2019; Setala et al. 2014; Song et al. 2014; Vajargah et al. 2018; Vajargah et al. 2019; Veneman et al. 2017; Wright et al. 2013b). Other than having physiological effects, microplastic causes external effects to the ecosystem by shifting the distribution of food and light for organisms (Cole et al. 2016; Song et al. 2014).

The translocation of microplastic in marine organisms has been reported (Scanes et al. 2019; Windsor et al. 2019). The movement of microplastic through biological membranes can have varying impacts depending on the species and location that the microplastic is translocated to. The translocation routes can begin from the ingestion or inhalation of microplastic to the gastrointestinal tracts, gills, or lungs, and then to the circulatory system, haemolymph, liver, or kidney (Abbasi et al. 2018; Leroueil et al. 2008; Su et al. 2018; Vajargah et al. 2018; Vajargah et al. 2019). Moreover, microplastic causes gastrointestinal blockage, reduced feeding or satiation (Lusher et al. 2013), structural deterioration, nutritional or growth complications (Jabeen et al. 2018; Peda et al. 2016; Yin et al. 2018), inflammation (Lu et al. 2016), and alteration of metabolic profile (Mattsson et al. 2014).

Furthermore, microplastic contributed to egestion inefficiency and intestinal damage. For example, microplastic produced higher retention times in the monogonont rotifer, *Brachionus koreanus* (Jeong et al. 2016). In another example, microplastic in

Caenorhabditis elegans caused mortality, accumulation, lowered intestinal Ca^{2+} level, and elevated glutathione S-transferase expression, which led to chronic intestinal damage (Lei et al. 2018). Additionally, microplastic was reported to externally affect marine organisms and ecosystems. A water surface that is polluted with microplastic impedes the respiration and photosynthesis of marine plankton (Amin et al. 2020; Song et al. 2014). Other than this, ingested microplastic that is excreted along with faecal matter can alter the density and sinking velocity of the latter, which is an important food source, hence interrupting the normal pattern of food distribution for other marine organisms (Cole et al. 2011, 2016).

However, contradictions were also reported. For example, limited acute toxicity and no mortality were observed from the ingestion of microplastic by the Antarctic krill *Euphausia superba* (Dawson et al. 2018). In summary, the sorption, aggregation, ingestion, retention, egestion, reingestion, and release of chemicals present potential mechanisms for the transport of POP and metals. The transfer and bioaccumulation of organic pollutants, such as PAH, PCB, EDC, and polybrominated diphenyl ether (PBDE), in living organisms, such as lugworms, marine amphipods, mussels, rainbow fish, and microalgae, via sorption on microplastic have been detected (Fisner et al. 2017; Guo et al. 2020; Llorca et al. 2018; Rehse et al. 2018; Rochman et al. 2013b; Wang et al. 2020b). Although some studies have not reported significant negative effects of microplastic, reports on

the hazards of microplastic toward the environment should not be ignored.

2.10 Types of microplastic-bound pollutants bioaccumulated in organisms

Microplastic pollution in the ocean is one of the most serious recent environmental problems. The ingestion of microplastic has been reported for a wide range of organisms, from herbivores and secondary consumers to the top predators (e.g. Barboza et al. 2018a; Clark et al. 2016; Nelms et al. 2018; Vroom et al. 2017; Wang et al. 2020b). Previous research has indicated the potential adverse and long-term environmental impacts of microplastic in the ocean. For example, the ingestion of larger microscopic particles may cause harmful physical effects to living organisms (Wright et al. 2013b). Moreover, other hypotheses suggested the deleterious effects of microplastic, regarding its role in the delivery of harmful chemicals to the ecosystem via the food chain (Avio et al. 2017; Hartmann et al. 2017; Lohmann 2017; Wang et al. 2018a; Wang et al. 2020b; Zhang et al. 2020a). It is well known that microplastic in seawater may sorb chemical pollutants, e.g. hydrophobic organic chemicals. These chemicals may originate from the chemical additives that were incorporated into plastics during manufacture (Kwon et al. 2017) or accumulated from the surrounding polluted water because of its high sorption capacity (Bakir et al. 2012; Liu et al. 2016; Velzeboer et al. 2014).

Similarly, when conditions move in the opposite direction that discourages sorption, contaminants are at risk of being desorbed from the ingested microplastic into the biological systems of living organisms. This situation makes microplastic both a source and a sink of pollutants (Liu et al. 2019c). Additionally, the reports of the bioaccumulation of toxic pollutants in other living organisms suggest a similar situation for humans, whereby microplastic–pollutant complexes can release the pollutants into the surrounding biological tissue. Therefore, this raises an important question of whether microplastic significantly contributes to the uptake and accumulation of environmental pollutants through the marine food web than the other sources.

Thus far, several types of environmental pollutants that can be found in living organisms along with plastic debris have been documented, which suggests that microplastic can act as vectors for the transfer of chemical contaminants from seawater to organisms (Table 3). Because of the hydrophobic surface of the plastics, a hydrophobic organic compound is the most noticeable category that transfers via this pathway. Plastic materials, such as PS, PE, and PP, may sorb hydrophobic organic compounds, namely PCB, PBDE, PAH, and hexabromocyclododecane (HBCD), and then, carry them to

organisms. Furthermore, recent studies have shown that microplastic can be a vector for heavy metal contamination in the marine environment (Brennecke et al. 2016; Davarpanah and Guilhermino 2015). The model animal, zebrafish (*Danio rerio*), exposed to Ag-incubated PE microbeads significantly increased the proportion of intestinal Ag levels, which suggested that the microplastic altered the uptake route of a metal contaminant in a model fish species (Khan et al. 2015). Such ecotoxicological responses by microplastic-bound pollutants have been observed in other cases as well. The exposure of lugworms (*Arenicola marina*) to PVC particles containing TCS or nonylphenol (NP) resulted in lower survivorship, feeding rate, and phagocytic activity than that of the lugworms to clean PVC particles (Browne et al. 2013). Using Japanese medaka (*Oryzias latipes*) as model animals, the researchers indicated that the ingestion of LDPE microplastic with the environmental pollutants sorbed from San Diego Bay may cause hepatic toxicity and pathological responses (Rochman et al. 2013a). Pollutant exposure that was transferred by microplastic resulted in alterations in behaviour, wherein beachhoppers (*Platorchestia smithi*) that ingested PAH-contaminated PE beads displayed reduced jump height and survival rate (Tosetto et al. 2016). Moreover, microplastic could be a vector for the transport of pathogens. Viršek et al. (2017) demonstrated 28 bacterial species on the microplastic collected from North Adriatic, including *Aeromonas salmonicida*, which is a pathogenic bacterium that severely impacts fish aquaculture.

As a vector, the microplastic itself can be transferred through the trophic levels. Research has shown that microplastic is readily ingested by zooplanktons (Botterell et al. 2019), which is the main food source for many secondary consumers. Ingested microplastic could either be transferred up the trophic levels via predation (Farrell and Nelson 2013) or be egested from the food web and then very likely be re-ingested (Mazurais et al. 2015). There are relatively few studies on the estimation of the potential role of microplastic in transferring chemicals through the trophic structure. Lugworms that were exposed to PCB-contaminated PE showed an increase in bioaccumulation than the control group exposed to PCB alone (Besseling et al. 2013), which was in agreement with an earlier estimation of the bioavailability model discussed by Teuten et al. (2007). For the top predators, namely great shearwaters (*Puffinus gravis*) (Ryan 1988) and short-tailed shearwaters (*Puffinus tenuirostris*) (Tanaka et al. 2013), the concentrations of PCB and PBDE in the fat tissue and eggs showed a positive correlation to the amount of ingested plastic particles. However, note that the pollutants accumulated in the trophic structure may be transferred through the food item or prey itself rather than the microplastic vector. A

Table 3 Types of microplastic-carried pollutants bioaccumulated in marine organisms

Organism	Class	Pollutant	Microplastic	Reference
Marine unicellular alga (<i>Tetraselmis chuii</i>)	Chlorodendrophyceae	Copper	PE	Davarpanah and Guilhermino (2015)
Lugworm (<i>Arenicola marina</i>)	Polychaeta	PCBs	PS	Besseling et al. (2013)
Lugworms (<i>Arenicola marina</i>)	Polychaeta	Triclosan or nonylphenol	PVC	Browne et al. (2013)
Mediterranean mussel (<i>Mytilus galloprovincialis</i>)	Bivalvia	HBCDs	PS	Jang et al. (2016)
Peppery furrow shell (<i>Scrobicularia plana</i>)	Bivalvia	BaP	LDPE	O'Donovan et al. (2018)
Zooplankton ^a	^a	PBDEs and PCBs	PE and PP	Yeo et al. (2020)
Marine amphipod (<i>Allorchestes Compressa</i>)	Malacostraca	PBDEs	PE	Chua et al. (2014)
Daphnia (<i>Daphnia magna</i>)	Branchiopoda	PCB	PS	Lin et al. (2019)
Marine copepod (<i>Acartia tonsa</i>)	Maxillopoda	PAHs	PS	Sørensen et al. (2020)
Marine copepod (<i>Acartia tonsa</i>)	Maxillopoda	Triclosan	PE	Syberg et al. (2017)
Marine copepod (<i>Calanus finmarchicus</i>)	Hexanauplia	PAHs	PS	Sørensen et al. (2020)
Beachhopper (<i>Platorchestia smithi</i>)	Malacostraca	PAHs	PE	Tosetto et al. (2016)
North Pacific lanternfish (<i>Tarletonbeania taylori</i>)	Actinopterygii	Lower chlorinated PCBs/pesticides	Plastic ^a	Gassel and Rochman (2019)
Japanese medaka (<i>Oryzias latipes</i>)	Actinopterygii	PHAs, PCBs, and PBDEs	LDPE	Rochman et al. (2013a)
zebrafish (<i>Danio rerio</i>)	Actinopterygii	Ag	PE	Khan et al. (2015)
Rainbow trout (<i>Oncorhynchus mykiss</i>)	Actinopterygii	Ag	PE	Khan et al. (2017)
Rainbow fish (<i>Melanotaenia fluviatilis</i>)	Actinopterygii	PBDEs	PP and PS	Wardrop et al. (2016)
Rabbitfish (<i>Siganus rivulatus</i>)	Actinopterygii	PCB	DPE and PP	van der Hal et al. (2020)
Streaked shearwater (<i>Calonectris leucomelas</i>)	Aves	PCB	PE	Tanaka et al. (2018)
Streaked shearwater (<i>Calonectris leucomelas</i>)	Aves	PBDEs	PE	Tanaka et al. (2015)
Laysan albatross (<i>Phoebastria immutabilis</i>)	Aves	PCBs, PAHs, and pesticides	PP and PE	Rios et al. (2007)
Northern fulmar (<i>Fulmarus glacialis</i>)	Aves	PCBs	Plastic ^a	Herzke et al. (2016)
Short-tailed shearwaters (<i>Puffinus tenuirostris</i>)	Aves	PBDEs	Plastic ^a	Tanaka et al. (2013)
Great Shearwaters (<i>Puffinus gravis</i>)	Aves	PCBs	Plastic ^a	Ryan (1988)

^aNo information

comparison with the concentrations of stomach plastic debris revealed that high PCB levels in northern fulmar (*Fulmarus glacialis*) are more likely to be contributed by their prey, instead of the plastics debris (Herzke et al. 2016). Moreover, in contrast to the vector role, microplastic could play a scavenger role that removes contaminants from individuals. Using ¹³C-labelled PBDE, the researchers showed that microplastic ingestion could both transfer and remove contaminants from sandhopper (*Talitrus saltator*), indicating a partial balance among the positive and the negative effects (Scopetani et al. 2018). Such conclusion is supported by the correlation between the concentrations of the hydrophobic organic compounds in the muscle tissue and the abundance of plastic particles in the stomachs of herring (*Clupea harengus membras*) collected along the Baltic Sea (Ogonowski et al. 2017).

However, most of the model analyses indicated that the ingestion of microplastic may not be an important pathway for the transfer of sorbed chemicals from the seawater to the ecosystem (Bakir et al. 2016; Gouin et al. 2011; Koelmans et al. 2014; Koelmans et al. 2016). Nevertheless, the extent of pollutants to be accumulated in the food web via the pathway of microplastic as a vector is still inconclusive. Furthermore, the chemical affinities of the plastics may influence their vector role (Koelmans et al. 2013). By measuring the desorption rates in artificial gut fluid, researchers have found that the intake of microplastic-bound hydrophobic organic compounds by organisms with similar gut retention times as fish varied according to the type of chemical. Moreover, the uptake of hexachlorocyclohexane (HCH) via PE microplastic ingestion can be negligible as compared to the uptake via other exposure routes, such as

water ventilation and food ingestion. However, microplastic ingestion can still increase the total uptake rate of pentachlorobenzene (PeCB) and hexachlorobenzene (HeCB) (Lee et al. 2019). Hence, future studies need to focus on the plastic debris that may have higher chemical affinities than the others. For example, PBAT has a greater vector effect than traditional materials because of its higher sorption and desorption capacities as an organic pollutant (Zuo et al. 2019).

2.11 Pollutant bioaccumulation of microplastic and its factors

The factors of bioaccumulation are desorption, organism internal environment, and retention time (Bakir et al. 2014; Barboza et al. 2018a; Ma et al. 2020a). Other bioaccumulation factors were similar to those that affect desorption, such as the size of microplastic and hydrophobicity (Ma et al. 2016). Bioaccumulation has been investigated and reported by several studies (Barboza et al. 2018a; Besseling et al. 2017; Chua et al. 2014; Ma et al. 2016; Rehse et al. 2018; Rochman et al. 2013b).

Bioaccumulation was prominently exhibited in a recent work by Barboza et al. wherein mercury concentrations in the gills and livers of juvenile European seabass *Dicentrarchus labrax* were doubled in the microplastic/mercury mixture as compared to in the mercury-only setup (Barboza et al. 2018c). The presence of PE microplastic was also reported to increase the bioaccumulation of PBDE, PCB, and PAH (Rochman et al. 2013a). In another study, PBDE was bioaccumulated in the marine amphipod *Allorchestes compressa* in the presence of PE microplastic (Chua et al. 2014). Moreover, the congener hydrophobicity of PCB congeners showed varied bioaccumulation capacity in the marine lugworm *Arenicola marina* in the presence of PS (Besseling et al. 2013, 2017). Additionally, the presence of smaller PS microplastic (50 nm) increased the bioaccumulation of phenanthrene in planktonic crustacean *Daphnia magna*.

However, a contradiction was reported in a study when the microplastic PA/bisphenol mixture and the bisphenol-only setup did not exhibit dissimilar bioaccumulation or EC50 for *Daphnia magna* immobility (Rehse et al. 2018). The PA material could be a negligible vector of bisphenol. A similar finding was reported where the microplastic PE/phenanthrene mixture and the phenanthrene-only setup did not exhibit dissimilar EC50 for the immobilisation of *Daphnia magna* (Frydkjaer et al. 2017). Batel et al. (2016) reported biomagnification when microplastic was found to transfer benzo[a]pyrene (BaP) from shrimp nauplii to zebrafish in an artificial aquatic food chain of zebrafish *Danio rerio* and brine shrimp nauplii *Artemia* sp.

Tourinho et al. suggested four possible scenarios of pollutant bioaccumulation introduced by microplastic into

organisms, which were (1) strong sorption and strong desorption, (2) strong sorption and low desorption, (3) low sorption and strong desorption, and (4) low sorption and low desorption (Coffin et al. 2019; Magara et al. 2018; Paul-Pont et al. 2016; Tourinho et al. 2019; Ziajahromi et al. 2019). Firstly, microplastic with strong sorption and strong desorption acts as a vector for bioaccumulation. Pollutant uptake at high sorption levels can lead to high chemical desorption and increased bioaccumulation in an organism. A fast desorption rate was reported under gut conditions. For example, the 50-fold increment of microplastic (high sorption) led to higher venlafaxine concentrations (high desorption and bioaccumulation) in the liver of pond fish *Misgurnus anguillicaudatus* (Qu et al. 2018). In another example, 30-fold pollutant desorption was observed in the gut than in the seawater by Bakir et al. (Bakir et al. 2014; Coffin et al. 2019; Nor and Koelmans, 2019). Secondly, the scenario with strong sorption and low desorption may decrease bioavailability despite the pre-sorbing of microplastic with the pollutant. Examples that resembled this scenario include the low bioaccumulation of pre-sorbed fluoranthene and Ag by PE in mussels and zebrafish, respectively (Khan et al. 2015; Magara et al. 2018; Paul-Pont et al. 2016). Thirdly, the scenario with low sorption and strong desorption has a low sorbed fraction to desorb a high quantity of pollutants. This scenario resembles the low sorption of PBDE and bifenthrin on PE that are bioaccumulated equally or nearly completely in amphipods and midge *Chironomus tepperi*, respectively (Chua et al. 2014; Ziajahromi et al. 2019). Fourthly, the scenario with low sorption and low desorption typically ends with proportionally low bioaccumulation. For example, the low sorption of fluoranthene by microplastic was shown to produce low fluoranthene bioaccumulation in mussels *Mytilus* spp. (Paul-Pont et al. 2016). In this scenario, the main source of pollutants is typically through dietary means; hence, the distribution or bioavailability of pollutants would be the same with or without microplastic (Tourinho et al. 2019).

However, the ingestion of low-contaminated microplastic can lead to low bioaccumulation. As shown in Nor and Koelmans (2019), PCB were transferred from contaminated food to clean PE microplastic under a simulated gut condition. This suggests a fifth scenario, whereby instead of pre-sorbed microplastic, unsorbed or low-contaminated microplastic is ingested and then egested in a sorbed state caused by the strong sorption of pollutants in the organism's gut.

Furthermore, a delayed effect may occur, such as during or after depuration, and hence, it can be an important experimental step. For instance, excluding or including the remaining microplastic in the organism's gut at the end of an interval or experiment can cause disparities, particularly in comparative research (Paul-Pont et al. 2016; Tourinho et al. 2019). In a study on bioaccumulation in mussel

Mytilus spp., the fluoranthene-only setup showed more toxicity before depuration, while higher fluoranthene concentration and histopathological damage were observed in the fluoranthene/microplastic mixture setup after depuration (Paul-Pont et al. 2016). Moreover, the combined setup of the pyrene/PE microplastic mixture showed the delayed mortality of fish *Pomatoschistus microps* (Oliveira et al. 2013). Nevertheless, the existing literature indicates that most studies have not reported a prominent bioaccumulation of pollutants by microplastic. According to Koelmans et al. the dietary pathway had a more direct bioaccumulation effect than microplastic (Koelmans et al. 2016).

The literature review revealed the impact of the differences in methodology toward the data. Researchers should consider the presence or absence of pre-incubation or pre-sorption, depuration period, combined co-exposure effect of a microplastic/pollutant mixture analysis, single effect of a pollutant-only analysis, and a dose-response analysis (Khan et al. 2015; Zhu et al. 2018a). Questions and challenges still exist concerning the bioaccumulation factors of microplastic-bound pollutants in organisms, as there are still gaps in this field that is yet to be thoroughly studied partially because of the varied internal environments of different organisms.

2.12 Risks of pollutant bioaccumulation to the marine ecosystem and humans

2.12.1 Risks toward the marine ecosystem

Other than the dangers of microplastic on its own, the risks of contaminants that are bound to the microplastic

add to the combined effects of microplastic-pollutant complexes (Fig. 4). The contaminants were reported to desorb in the intestine of organisms followed by translocation to the circulatory system, intestinal epithelia, tissues, and organs such as liver. The combined effects of microplastic and the bioaccumulation of pollutants caused by microplastic-pollutant complexes include physiological stress, morphological abnormalities, immobilisation, neurotoxicity, oxidative damage, enzymatic dysregulation, reduced growth rate, reduced chlorophyll concentration, altered immune response, genotoxicity, and mortality (Barboza et al. 2018b; Batel et al. 2016; Khan et al. 2017; Kim et al. 2017a; Ma et al. 2016; Prata et al. 2018; Qiao et al. 2019; Qu et al. 2018; Syberg et al. 2017; Zhu et al. 2019). Heavy metals accumulate in brain and muscle tissues, causing the abovementioned neurotoxicity, oxidative damage, and enzymatic dysregulation, for example, the Ag accumulated in the intestinal mucous, mucosal epithelium, muscle layer, and serosa (Khan et al. 2017). In another example, bioaccumulated phenanthrene and its by-products caused joint system toxicity and immobilisation and were translocated into the daphnid tissue, gut, and carapace (Ma et al. 2016). The combined effects of microplastic and pollutants, such as metals, were also detected on marine fish and microalgae (Kim et al. 2017a).

However, there are conflicting findings about the biological toxicity of microplastic. For example, Rochman and co-workers reported the nil toxic effect of virgin microplastic against a fish cell line (Rochman et al.

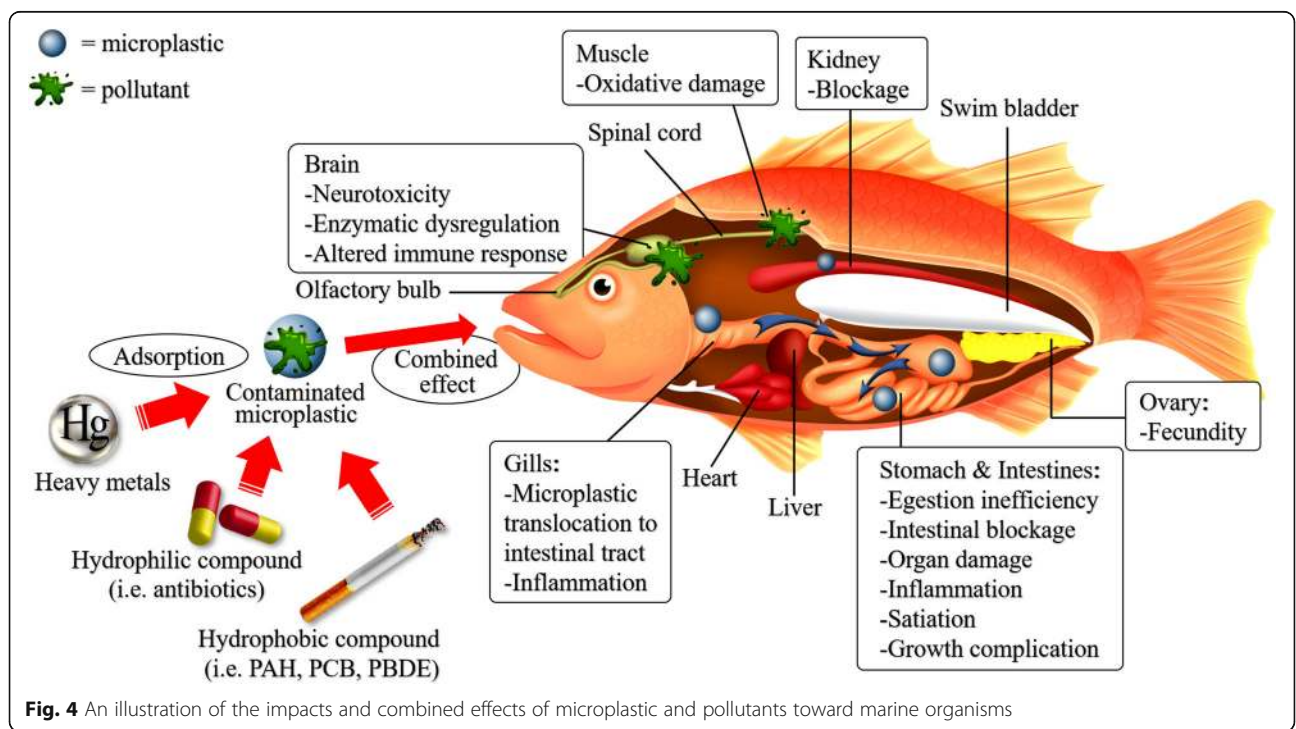


Fig. 4 An illustration of the impacts and combined effects of microplastic and pollutants toward marine organisms

2014). In another study, the data of microplastic ingestion by the Antarctic krill *Euphausia superba* showed no mortality, toxicity, weight loss, or bioaccumulation (Dawson et al. 2018). Moreover, some research reported lowered concentration exposure and reduced short-term effects of bisphenol A (BPA) and PAH toward *Daphnia magna* (Kleinteich et al. 2018; Rehse et al. 2018).

2.12.2 Risks toward humans

Microplastic can be introduced to humans via inhalation or dietary routes through the consumption of seafood, such as fish and shellfish. Based on the abundances observed, it was estimated that Chinese shellfish consumers could be exposed to 100,000 microplastics each year (Wright and Kelly 2017). Furthermore, approximately 700 microplastics/kg of microplastic, mainly PET and PE, were found in 15 brands of commercial sea salt (Yang et al. 2015a). A recent study reported the presence of mainly PET and PP microplastics in all the eight human stool samples considered (Schwabl et al. 2019). The existing literature shows that microplastic is endangering our food safety and security as it contaminates the food meant for human consumption, but there is a lack of more recent studies regarding its direct clinical impacts on the human body.

The effect of microplastic against the health of organisms was investigated more than three decades ago. The literature indicates the possibility of microplastic to avoid the clearance mechanism, penetrate human or mammalian airways and deep lungs, become embedded or lodged, and induce chronic or acute inflammation (Pauly et al. 1998; Porter et al. 1999; Wright and Kelly 2017). An older report described microfibrils as toxic to pulmonary cells and carcinogenic (Omenn et al. 1986). Investigations of microplastic in model mammalian systems indicated possible translocation from living cells to human organ systems, such as the lymphatic and circulatory systems and, thus, the ability to indirectly impact the immune system (Brown et al. 2001; Eldridge et al. 1989; Frohlich et al. 2009; Hodges et al. 1995; Jani et al. 1992; Rieux et al. 2005; Volkheimer 1975).

More recently, the health risks of microplastic have also been reported but are limited to the use of model organisms and relatively focused on the intoxication of organisms by the additives and pollutants leached from the microplastic than the microplastic itself (Costa et al. 2016). For example, a study analysed the *in vitro* gastrointestinal uptake of microplastic by using the human intestinal epithelial cell line and co-cultures mimicking intestinal M-cells and goblet cells, which showed no interference by the microplastic in terms of differentiation and activation (Stock et al. 2019). In another example, Cr-contaminated PLA microplastic in the human digestive system was studied using the whole digestive system

in vitro method, a systematic model that included the mouth, gastric, small intestine, and large intestine digestive phases (Liao and Yang 2020). The Cr levels were high in the gastric, small intestinal, and large intestinal phases. Furthermore, some pollutants, monomers, and additives in microplastic are POP and EDC that have been detected in the human body in a higher amount than expected by gut exposure, indicating substance introduction via other possible routes (Galloway 2015; Koelmans et al. 2016; Thompson et al. 2009; Vethaak and Leslie 2016). Additionally, long-term consumption of water contaminated with a plasticizer, phthalate, at levels above the US Environmental Protection Agency (USEPA) threshold (<6 µg/L) could cause liver and reproductive complications (Lambert et al. 2014; Martin and Voulvoulis 2009; USEPA 2012). Other health-threatening plastic additives (i.e. monomers, initiators, catalysts, emulsifiers, and stabilisers) include styrene, benzoyl peroxide, zeolites, and azobisisobutyronitrile (Todd et al. 2003). Carcinogenic PAH is among the environmental contaminants sorbed by microplastic in the ocean. These chemicals, now considered to be environmental contaminants, bioaccumulate in the fatty tissues of animals, which are eventually consumed by humans (Hwang et al. 2020). The literature review also demonstrated that the leaching of sorbed contaminants and the use of some additives and monomers contribute to the toxicity of microplastic–pollutant complexes toward living organisms, which include humans.

Although there are older studies describing the effect or existence of microplastic in humans and some relatively newer studies on the effect of microplastic conducted using model organisms or human cell lines as the reference, there is an urgent need for a more recent investigation on the possibility of microplastic causing any medical condition directly against human patients (Pauly et al. 1998; Porter et al. 1999; Stock et al. 2019). The lack of strong evidence of plastic leading to any direct detrimental effect on humans may allow continuous plastic production, consumption, and leaking of plastic debris into the oceans.

2.13 Gaps in microplastic research and future outlook

Microplastic is compromising our food safety because it pollutes the food meant for human consumption. Nonetheless, research on the impact of microplastic on the internal human body is limited. Moreover, the literature review revealed the importance of methodology differences in terms of data comparison and validity. For example, a few aspects in the methodology that should be contemplated include pre-incubation or pre-sorption, depuration, combined co-exposure of a microplastic and pollutant mixture, pollutant-only effect analysis, and dose–response analysis. Furthermore, because of the lack of standardised methods, studies have adopted different

microplastic sampling and analytical methods (Jiang et al. 2020). The different definitions, sampling methods, and analytical methods of microplastic could affect the comparability and consistency of data among different researchers.

Furthermore, mixed findings from the past literature suggest that the effect of microplastic ageing on pollutant sorption capacity depends on the type of pollutant or microplastic, and the gradual changes in their chemical bonds, forces, or interactions over time. Although some studies have reported the lack of negative effects from microplastic, other reports on the hazards of microplastic toward the environment should not be ignored. Additionally, questions still exist concerning the bioaccumulation factors of microplastic-bound pollutants in organisms, as internal environments vary from one organism to another.

The possible future approaches that can alleviate microplastic pollution include the collection and dissemination of insightful scientific data and facts, public awareness, stakeholder engagement, and the incorporation of environmentally friendly, alternative polymeric material (Gong et al. 2019; Govindasamy et al. 2019; Zuo et al. 2019). However, an in-depth investigation is required to confirm the biological, physical, or chemical properties that are responsible for the sorption capacity of a microplastic with a specific pollutant. Currently, the biofilm presence on microplastic has been reported to be one of the factors that can contribute to pollutant sorption by microplastic. Furthermore, the maximum sorption capacity of a material can become a limiting factor of toxicity. Moreover, a broad comparative or statistical approach can link the associative patterns of microplastic and pollutants with environmental conditions, such as pH and salinity. Research in the direction of investigating the chemical interactions between microplastics, pollutants, and the environment can disclose new indirect factors of pollutant sorption by microplastic. Additionally, future studies need to focus on the plastic debris that may have higher chemical affinities than the others. For example, PBAT has a greater vector effect than traditional materials because of its higher sorption and desorption capacities as an organic pollutant. Moreover, instead of that of pre-sorbed microplastic, the potential of unsorbed or low-contaminated microplastic to be egested in a sorbed state and carry pollutants out of the gut of organisms is also worth investigating in future experiments. Therefore, future microplastic research is expected to move toward achieving a deeper understanding of the factors related to pollutant sorption and bioaccumulation by microplastic as well as standardising the protocols involved in microplastic research.

Microplastic pollution has been a long-standing cause for concern with respect to environmental engineering, ecology, and materials sciences. However, to solve the problems associated with microplastic pollution, at least accumulatively at a small scale, research efforts toward understanding plastic compositions, characterising plastics, tracing the sources and sinks of plastics, grasping the complex microplastic–environment interactions, implementing policies, biotechnological approaches, and infrastructure and technology advances are required in combination (Hale et al. 2020; Barceló and Picó 2020; Agamuthu et al. 2019). This may include ways of reducing plastic waste materials at the source (i.e. recycling and compostable packaging), reducing microplastic leakage from waste treatment facilities, or increasing the use of biodegradable or renewable-sourced plastic (Amelia et al. 2020; Neufeld et al. 2016; Zhang et al. 2020b). Systematic, innovative research and approaches are needed to minimise microplastic contamination.

3 Conclusions

In conclusion, the microplastics commonly found to pollute the marine environment were generally of the fibre or fragment type, contained PE or PP material, and were typically contributed by anthropogenic, urban, fishery, or marine activities. The currently known factors of microplastic bioavailability toward organisms are the size, colour, density, morphology, and mobility of microplastic, as well as the species, morphology, and physiology of organisms. Additionally, the factors known to affect the pollutant sorption of microplastic are the colour, density, age, and chemical properties of the microplastic, type of pollutant, biofilm presence, and environmental conditions such as dissolved organic matter, pH, and salinity. Understanding, providing further evidence, and disseminating the currently known factors of microplastic pollution, bioavailability, sorption, and bioaccumulation will be constructive and valuable in alleviating the global microplastic pollution crisis.

Moreover, the sorption, aggregation, ingestion, retention, egestion, reingestion, and release of chemicals present potential mechanisms for the transport of environmental contaminants (POP, PAH, and EDC), hydrophobic organic compounds, and metals. The literature review revealed that the chief and direct factor of pollutant sorption by microplastic is chemical interaction. Research in the direction of investigating the chemical interactions between microplastics, pollutants, and the environment can disclose new indirect factors of pollutant sorption by microplastic. Overall, the clear extent of pollutants to be accumulated in the food web via the pathway of microplastic as a vector is still inconclusive. For example, the chemical affinities of the plastics may

influence their vector role, and thus, future studies need to focus on the plastic debris that may have higher chemical affinities than the others. Numerous diverse factors, such as the species of organisms and the type of pollutants, govern the sorption and bioaccumulation (i.e. desorption) of pollutants in the presence of microplastic. It is still a research challenge to achieve a clearer and more comprehensive understanding of the underlying factors influencing the sorption and bioaccumulation behaviours. Progressive and objective research coupled with management could determine the future production, consumption, and leakage of plastic debris into the oceans.

Abbreviations

4MBC: 4-Methylbenzylidene camphor; AMX: Amoxicillin; BaP: Benzo [a]pyrene; CBZ: Carbamazepine; CIP: Ciprofloxacin hydrochloride; EDC: Endocrine-disrupting chemical; EE2: 17 α -Ethinylestradiol; EVA: Ethylene-vinyl acetate; FTIR: Fourier-transform infrared spectroscopy; HBCD: Hexabromocyclododecane; HCH: Hexachlorocyclohexane; HDPE: High-density polyethylene; HeCB: Hexachlorobenzene; LDPE: Low-density polyethylene; NP: Nonylphenol; PA: Polyamide; PAH: Polycyclic aromatic hydrocarbon; PB: Polybutylene; PBAT: Polybutylene adipate terephthalate; PBDE: Polybrominated diphenyl ether; PCB: Polychlorinated biphenyl; PE: Polyethylene; PeCB: Pentachlorobenzene; PES: Polyethersulfone; PET: Polyethylene terephthalate; PFAA: Perfluoroalkyl acids; PFOS: Perfluorooctanesulfonate; PFOSA: Perfluorooctanesulfonamide; PMMA: Poly (methyl methacrylate); POP: Persistent organic pollutant; PP: Polypropylene; PPCP: Pharmaceuticals personal care product; PS: Polystyrene; PU: Polyurethane; PVA: Polyvinyl acetate; PVC: Polyvinyl chloride; SBR: Styrene butadiene rubber; SDZ: Sulfadiazine; TC: Tetracycline; TCS: Triclosan; TMP: Trimethoprim; TYL: Tylosin; USEPA: US Environmental Protection Agency

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Authors' contributions

KB proposed the topic, conceived and designed the article. TSMA, YTS, WMAWMK, and HP carried out the review of literature and writing. MCO proofread and helped with the interpretation of review of literature. All authors read and approved the final manuscript.

Authors' information

TSMA is a graduate student in Universiti Malaysia Terengganu, Malaysia. KB, MCO, and YTS are her advisors. WMAWMK is a senior lecturer in Universiti Malaysia Terengganu, Malaysia. HJP is an assistant researcher in Institute of Earth Sciences, National Taiwan Ocean University, Taiwan.

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Availability of data and materials

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Competing interests

The authors declare that they have no competing interest.

Author details

¹Faculty of Science and Marine Environment, Universiti Malaysia Terengganu, 21030 Kuala Nerus, Terengganu, Malaysia. ²Microplastic Research Interest Group, Universiti Malaysia Terengganu, 21030 Kuala Nerus, Terengganu,

Malaysia. ³Ocean Pollution & Ecotoxicology Research Group, Universiti Malaysia Terengganu, 21030 Kuala Nerus, Terengganu, Malaysia. ⁴Institute of Marine Biology, National Taiwan Ocean University, Keelung 20224, Taiwan. ⁵Center of Excellence for the Oceans, National Taiwan Ocean University, Keelung 20224, Taiwan. ⁶Institute of Earth Sciences, College of Ocean Science and Resource, National Taiwan Ocean University, Keelung 20224, Taiwan. ⁷Center of Excellence for Ocean Engineering, College of Engineering, National Taiwan Ocean University, Keelung 20224, Taiwan. ⁸Institute of Marine Biotechnology, Universiti Malaysia Terengganu, 21030 Kuala Nerus, Terengganu, Malaysia. ⁹Malaysian Institute of Pharmaceuticals and Nutraceuticals, National Institutes of Biotechnology Malaysia, 11700 Penang, Malaysia.

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