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Review article

Particulate plastics as a vector for toxic trace-element uptake by aquatic and terrestrial organisms and human health risk

Lauren Bradney[a], Hasintha Wijesekara[a,b,*], Kumuduni Niroshika Palansooriya[c], Nadeeka Obadamudalige[a], Nanthi S. Bolan[a,d], Yong Sik Ok[c], Jörg Rinklebe[e,f], Ki-Hyun Kim[g,**], M.B. Kirkham[h]

[a] Global Centre for Environmental Remediation (GCER), Advanced Technology Centre, Faculty of Science, The University of Newcastle, Callaghan, NSW 2308, Australia
[b] Department of Natural Resources, Faculty of Applied Sciences, Sabaragamuwa University of Sri Lanka, Belihuloya 70140, Sri Lanka
[c] Korea Biochar Research Center, O-Jeong Eco-Resilience Institute (OJERI) & Division of Environmental Science and Ecological Engineering, Korea University, Seoul 02841, Republic of Korea
[d] Cooperative Research Centre for High Performance Soil (Soil CRC), The University of Newcastle, Callaghan, NSW, 2308, Australia
[e] University of Wuppertal, School of Architecture and Civil Engineering, Institute of Foundation Engineering, Water- and Waste-Management, Laboratory of Soil- and Groundwater-Management, Pauluskirchstraße 7, 42285 Wuppertal, Germany
[f] Department of Environment, Energy & Geoinformatics, Sejong University, 98 Gunja-Dong, Guangjin-Gu, Seoul, South Korea
[g] Air Quality & Materials Application Lab, Department of Civil & Environmental Engineering, Hanyang University, 222 Wangsimni-Ro, Seoul 04763, Republic of Korea
[h] Department of Agronomy, Throckmorton Plant Sciences Center, Kansas State University, Manhattan, KS 66506-5501, United States

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ABSTRACT

Particulate plastics in the terrestrial and aquatic environments are small plastic fragments or beads (i.e., 5 mm down to the nanometre range). They have been frequently referred to as 'micro-plastics' or 'nano-plastics'. Research has identified particulate plastics as a vector for toxic trace elements in the environment. The adsorption of toxic trace elements by particulate plastics may be facilitated by their high surface area and functionalized surfaces (e.g., through the attachment of natural organic matter). Other factors, such as environmental conditions (e.g., pH and water salinity), surface charge, and trace element oxidation status, also influence the adsorption of trace elements onto particulate plastics. Because of their small size and persistence, particulate plastics and the associated toxic trace elements are readily ingested and accumulated in many terrestrial and aquatic organisms. Thus, these plastics can have severe environmental consequences, such as the development of metal toxicity, within aquatic and terrestrial organisms. Humans could also become exposed to particulate plastics through food chain contamination and airborne ingestion. This review provides an overview of the sources of particulate plastics in the environment. To this end, we describe particulate plastics made of synthetic polymers, their origin, and characteristics with emphasis on how particulate plastics and associated toxic trace elements contaminate terrestrial and aquatic ecosystems. Future research needs and strategies are discussed to help reduce the environmental risks of particulate plastics as a potent vector for the transportation of toxic trace elements.

1. Introduction

Particulate plastics encompass a group of plastic polymers smaller than 5 mm down to the nanometre range (Frias and Nash, 2019; Turner and Holmes, 2015). Depending on the size or shape of the particulate plastics, they can be classified as microbeads or micro- (i.e., 1 μm to 5 mm) or nano- (i.e., < 1 μm) plastics (Frias and Nash, 2019). Furthermore, the sources of particulate plastics can be grouped into two broad categories: primary particulate plastics and secondary particulate plastics (Duis and Coors, 2016). Primary particulate plastics are manufactured and are a direct result of anthropogenic use of plastic-based materials. They are used in a wide variety of applications as the essential components of industrial products (e.g., cosmetics and clothing) or as raw materials for industrial processes. Secondary particulate

* Correspondence to: H, Wijesekara, Global Centre for Environmental Remediation (GCER), Advanced Technology Centre, Faculty of Science, The University of Newcastle, Callaghan, NSW 2308, Australia.
** Corresponding author.
E-mail addresses: hasintha.wijesekara@newcastle.edu.au (H. Wijesekara), kkim61@hanyang.ac.kr (K.-H. Kim).

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plastics are derived from the breakdown of larger plastic debris, and they include, but are not limited to, plastic packaging, plastic bags, or bottles (Duis and Coors, 2016). This breakdown can occur through physical and chemical processes and is, therefore, dependent on the surrounding environment (Eerkes-Medrano et al., 2015). Both primary and secondary particulate plastics persist in terrestrial (i.e., soil) and aquatic (i.e., marine and freshwater) environments. Particulate plastics have also been identified in the atmospheric environment (i.e., atmospheric fallout) (Prata, 2018).

Large quantities of particulate plastics retained within the marine environment have originated from land-based sources, having been transported through processes such as sediment transfer or soil erosion. Despite this link to land-based sources, the majority of scientific research on particulate plastics has focused on their effects in aquatic environments (Browne et al., 2011; Cole et al., 2011; Thompson et al., 2009). However, studies are now demonstrating the effects that terrestrial based particulate plastics have on the surrounding environment. Particulate plastics can accumulate in the terrestrial environment through their indiscriminate disposal in landfills or through compost and biosolid application (Kilponen, 2016; Rochman, 2018; Zubris and Richards, 2005). Alternatively, the weathering of plastic film mulch used over agricultural fields is also a source of plastic particles in soils (Rochman, 2018).

The potential for particulate plastics to cause environmental contamination with their associated impacts upon organisms is a major concern. Upon reaching the terrestrial ecosystem, the degradation of many synthetic polymers is prolonged, and, thus, they are likely to persist in soil for a long time (Rillig, 2012). Degradation can occur through physical breakdown and biologically or chemically initiated processes (Dindar and Içli, 2001). Particulate plastics can interact with inorganic pollutants, including trace elements (e.g., aluminium (Al), cadmium (Cd), cobalt (Co), chromium (Cr), copper (Cu), iron (Fe), nickel (Ni), lead (Pb), and zinc (Zn)) in the environment. Adsorption and desorption of pollutants associated with particulate plastics in the soil environment are influenced by factors such as weathering and surface area, dissolved organic matter (DOM), and microbial activity. The high surface area of particulate plastics, which can be increased through weathering, not only aids adsorption of contaminants but also supports chemical transport of the pollutants through leaching (Holmes et al., 2014; Teuten et al., 2009). Results also indicate that particulate plastics impact aquatic environments through long-range transport of pollutants. Particulate plastics interactions with contaminants, such as trace elements, have gained significant attention and focus in recent years (Brennecke et al., 2016; Holmes et al., 2012; Holmes et al., 2014; Rochman et al., 2014; Turner and Holmes, 2015; Wang et al., 2018; Wang et al., 2017; Wijesekara et al., 2018).

Interaction of trace elements and particulate plastics can occur through three main processes. These three processes involve the uptake of trace elements through direct adsorption of cation complexes onto charged sites or neutral regions of the surface of the particulate plastic, co-precipitation, and, finally, adsorption onto hydrous oxides (Ashton et al., 2010). The adsorption of trace elements to particulate plastics may result in their accumulation in aquatic and sedimentary environments and provide a source of toxic trace elements (Munier and Bendell, 2018). Research regarding aquatic environments has demonstrated uptake of trace elements from particulate plastics (Holmes et al., 2012; Holmes et al., 2014; Turner and Holmes, 2015). Although there is limited research with terrestrial environments, agricultural practices and landfill leachate are potential sites for particulate-plastic and trace-element interactions. Aside from environmental adsorption, trace elements can also be present in particulate plastics due to the manufacturing process. For example, additives such as heat stabilizers and slip agents (i.e., which reduce the friction coefficient) are often added to particulate plastics (Hahladakis et al., 2018). Although these additions help with the plastic's properties, they can contain toxic trace elements such as Zn, Pb, and Cd (Hahladakis et al., 2018; Munier and Bendell, 2018). The effects regarding particulate plastics and trace element interactions in marine and aquatic environments are still in their early stages. However, the long-term consequences regarding particulate plastics, especially regarding human health, are largely unknown (Kontrick, 2018). Particles of plastics may cause an increase in chemical and trace element exposure, causing toxicity and increased bioaccumulation in aquatic, and potentially terrestrial, environments (Ng et al., 2018).

Because particulate plastics do not break down readily they can be ingested, incorporated into, and accumulated in the bodies and tissues of many terrestrial and aquatic organisms (Browne et al., 2008; Horton et al., 2017). Once entering the environment, particulate plastics then make their way up the food chain. For example, particulate-plastic contamination of earthworms (Huerta Lwanga et al., 2016), birds (Holland et al., 2016; Zhao et al., 2016), freshwater invertebrates (Blarer and Burkhardt-Holm, 2016), fish (Lusher et al., 2017), and oysters (Cole and Galloway, 2015; Vegter et al., 2014) has been reported. Because of these environmental consequences, regulatory bodies have called for the ban of consumer use of particulate plastics. Countries such as, but not limited to, the United States of America, Canada, Sweden, the Netherlands, and New Zealand have banned cosmetic particulate plastics. However, even if the use of particulate plastics is banned, those that have already reached our environments will continue to persist there for many more years.

In this study, we critically review the knowledge concerning particulate-plastic-assisted trace-element uptake by aquatic (i.e., freshwater and marine) and terrestrial organisms and the subsequent toxicity to humans. Additionally, we evaluate the sources and fate of particulate plastics; mechanisms involved in forming trace element-particulate plastics complexes; and their interactions and transportation in the environment. Due to the relatively limited information surrounding particulate plastics and human health, a significant focus is also given to understanding the processes involved in the uptake of trace elements associated with particulate plastics into the tissues and organs of terrestrial and aquatic organisms. This uptake links human exposure to particulate plastics through their presence in the food chain.

2. Sources and input of particulate plastics in the environment

Both primary and secondary particulate plastics can reach the environment through a number of pathways. For terrestrial ecosystem, the pathways include indiscriminate disposal in landfills, application of organic amendments (e.g., compost and biosolids), atmospheric deposition, and plastic film mulching (Rillig, 2012). Earthworms can ingest plastic and distribute it around the soil, but in doing so, they may create particulate plastics by breaking down brittle plastic within their gizzard (Rillig, 2012). Aquatic environments receive particulate plastics from land-based sources (e.g., run-off and soil erosion), effluent from wastewater treatment plants, and marine activities (e.g., shipping). Fig. 1 shows the pathways of transport of particulate plastics to aquatic and terrestrial ecosystems.

2.1. Wastewater treatment plants

Certain brands of soaps, toothpaste, and facial scrubs can contain particulate plastics to prolong shelf-life, increase durability, improve aesthetics, or to be used as an exfoliating agent (Fendall and Sewell, 2009; Murphy et al., 2016). However, these personal care products are commonly designed to be washed down the sink after their application to the face, body, or teeth. Therefore, if particulate plastics are present, this can result in their transportation to wastewater treatment plants via raw effluent (Murphy et al., 2016; Sundt et al., 2014). Despite efforts to remove particulate plastics from wastewater at treatment plants, a small percentage still bypass the removal stage. However, despite many particles being successfully removed, a study conducted by Murphy et al. (2016) estimated that there were still 65 million particles released

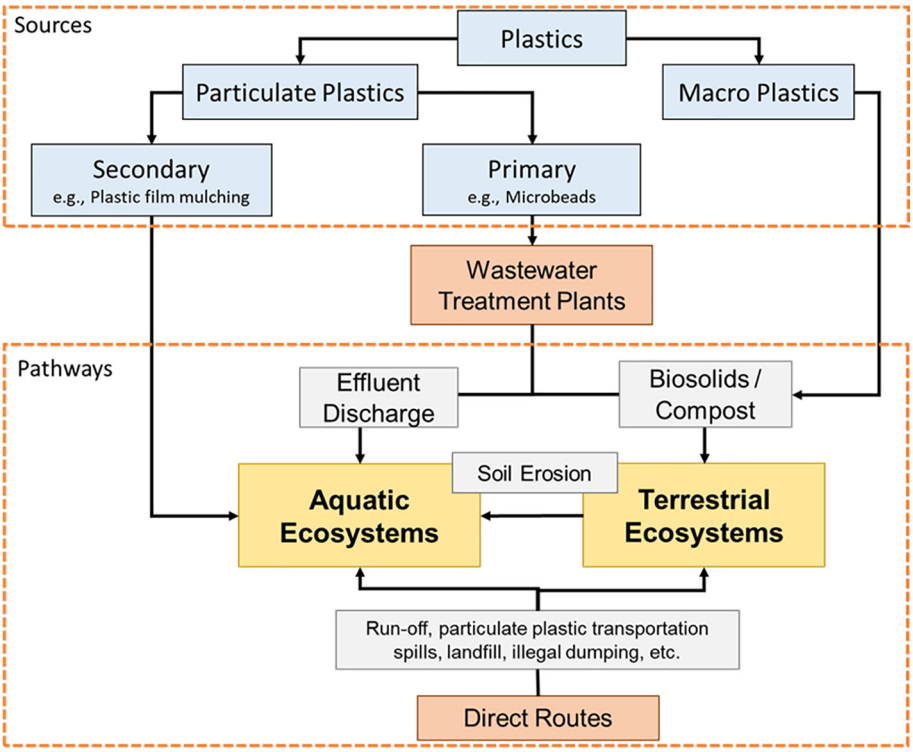

Fig. 1. Conceptual diagram illustrating sources and pathways of particulate-plastics-associated trace elements.

every day from a wastewater treatment plant in Glasgow, Scotland.

Biosolids can also contain particulate plastics as they are often ineffectively removed from treated sludge (Rochman et al., 2015a). Furthermore, the treatment process can also influence the particulate plastic concentration, as well as the plastic's size and morphology. Biosolids derived from anaerobic digestion have been reported to contain a lower abundance of particulate plastics when compared to other treatment processes (e.g., lime stabilization and thermal drying). Furthermore, alkaline hydrolysis used during lime stabilization can shear the particulate plastics thereby causing an increased abundance. Therefore, the generation process is an important factor for the existence of particulate plastics (Mahon et al., 2016; Zubris and Richards, 2005). Finally, the concentration of particulate plastics in Australian biosolids was found to be 898 particles/kg of biosolids (Bradney, 2017). Fig. 2 shows a selection of the more commonly shaped particles extracted (i.e., irregular, fibres, spherical, hexagonal, square, and triangle) in this study.

2.2. Compost

There has been renewed interest in the large-scale application of compost to soil, mainly to increase soil health (Ng et al., 2018). Although medium and large-size plastic materials are generally segregated during the composting process through sieving and hand picking, a percentage of smaller plastic material will remain in the final product (Weithmann et al., 2018). Additionally, because of the subsequent milling of composts, many larger plastic fragments will also end up as particulate plastics and remain in the finished compost. Furthermore, compost regulations in many countries still allow a portion of plastic to remain within the final product. For example, in 2016 the NSW Environmental Protection Agency specified that composts supplied for land application should not contain > 0.5% (dry weight) of glass, metal, and rigid plastics (> 2 mm), or 0.05% (dry weight) for flexible plastics (> 5 mm) (NSW Environmental Protection Agency, 2016).

2.3. Landfill

An estimated 22–43% of waste production is stored within landfills worldwide, some of which have conditions less than desirable for the disposal of plastic (Nizzetto et al., 2016; UNEP, 2014). Landfills with high exposure to sunlight (UV degradation) and weathering have a higher chance of becoming a source of secondary particulate-plastic pollution to the aquatic environment (Nizzetto et al., 2016). Because plastics take a long time to degrade, relatively old (i.e., methanogenic phase) landfills may still be leaching particulate plastics into the surrounding environment (Sundt et al., 2014).

2.4. Erosion and run-off

Particulate plastics in aquatic ecosystems often have a terrestrial origin (Andrady, 2015). Land-based sources include overland run-off, sediment transfer, and soil erosion (Cole et al., 2011; Horton et al., 2017). Primary particulate plastics can also enter the marine environment or soils, through littering and accidental spillage during the transportation or manufacturing stage (Sundt et al., 2014). The extent of erosion, runoff, and concentration of particulate plastics within soils is linked to the surrounding land-use. Furthermore, land-uses such as contamination from manufacturing industries, agricultural and horticultural industries that use plastic mulch, and lands where biosolids have been applied (e.g., mine rehabilitation sites) have the potential to contain a high level of plastic and particulate plastics (Rochman et al., 2015a). Generally, particulate plastics that have originated on land will first reach rivers before being transported into the ocean. However, particulate plastics that directly enter the marine environment, such as via ships, are not considered to have first originated from land.

2.5. Marine activities

Marine environments often contain particulate plastics that have originated from secondary sources, such as paint chips and fibres from marine ropes and nets for aquaculture (Duis and Coors, 2016; Ivar do

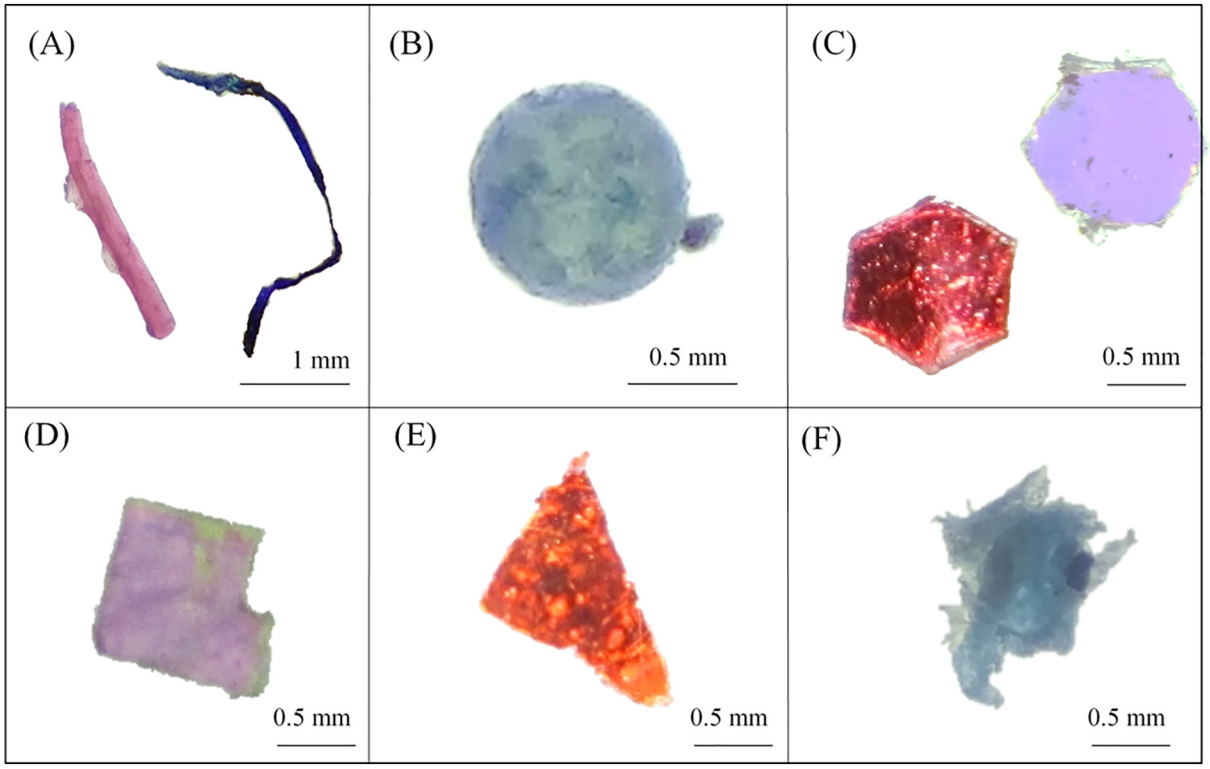

Fig. 2. Particulate plastics found in biosolids under stereomicroscopy. (A) fibres, (B) spherical, (C) hexagonal, (D) square, (E) triangle, and (F) irregular.

Sul et al., 2014). A study conducted by Song et al. (2014) examined the extent of particulate plastics that reside on the sea surface microlayer around Korea. It was found that the particles present consisted mainly of alkyds (81%) and poly(acrylate/styrene) (11%). Both these polymers are used in industrial paints, while poly(acrylate/styrene) is also used in fibre-reinforced plastic. Due to the characteristics of the polymers, Song et al. (2014) suspected that the source of these polymers originated from ships and fishing boats. Particulate plastics used within paints can enter the marine environment through weathering or during the application and maintenance stage and should, therefore, be considered a source of particulate plastics (Sundt et al., 2014). Equally, weathering can also contribute to the creation of secondary particulate plastics (Eerkes-Medrano et al., 2015). The generation of particulate plastics due to weathering may differ depending on the surroundings and environmental conditions. For example, the breakdown from wave action is more extreme in marine environments compared to freshwater environments (Eerkes-Medrano et al., 2015). Table 1 shows synthetic polymers made of particulate plastics, their origin, and characteristics.

3. Particulate plastics as a vector for toxic trace elements in the environment

Particulate plastics are capable of accumulating trace elements from the surrounding sediment or aquatic environment (Holmes et al., 2012; Holmes et al., 2014; Turner and Holmes, 2015). Particulate plastic's hydrophobic nature and large surface area facilitate the adsorption of contaminants. Furthermore, organic contaminants prefer to bind with particulate plastics as opposed to larger plastic fragments (Anderson et al., 2016; Teuten et al., 2009). Particulate plastic's ability to act a vector can thereby result in numerous detrimental effects to both terrestrial and aquatic ecosystems (Brennecke et al., 2016; Zhao et al., 2016).

3.1. Particulate plastics - toxic trace element interactions

Metal interactions with particulate plastic take place through three main processes: firstly, direct adsorption of cations (physical); secondly, complexation onto charged sites or neutral regions of the surface of the particulate plastic and co-precipitation (chemical); and thirdly, adsorption onto hydrous oxides (Ashton et al., 2010). Many studies have revealed the interactions between trace elements and particulate particles in various aquatic environments. Holmes et al. (2012) measured the concentration of trace elements in plastic production pellets at four beaches (i.e., Saltram, Sharrow Point, Ninney Rock, and Watergate Bay) in southwest England. The highest mean concentrations of Al, Fe, manganese (Mn), Co, and Ni were found at “Saltram,” which were 55,800, 97,800, 20,500, 107, and 131 ng/g, respectively. “Sharrow Point” showed the highest mean concentrations for Cu, Zn, and Pb, which were 1.32, 23.3, and 1.64 ng/g, respectively. In addition, Cr and Cd concentrations found at “Ninney Rock” were 751 and 76.7 ng/g (Holmes et al., 2012). Interestingly, some of the trace element concentrations were greater than the concentrations of reported local estuarine sediments. This highlights the ability of particulate plastics to act as a vector for contaminants such as trace elements. In addition, it raises concerns regarding the potential increase in an organism's exposure to contaminants compared to that which would otherwise occur in the surrounding environment. A later study by Turner and Holmes (2015) found adsorption of trace elements (i.e., silver (Ag), Cd, Co, Cr, Cu, mercury (Hg), Ni, Pb, and Zn) in beached pellets in river water. The median concentrations of those metals ranged from as low < 3 ng/g for Ag and Hg to as high as 34,400 ng/g for Fe (Table 2). Another study by Holmes et al. (2014), who used the Langmuir and Freundlich models for both virgin and aged pellets, showed trace-element (i.e., Cd, Co, Cr, Cu, Ni, Pb) adsorptions under estuarine conditions (Holmes et al., 2014). In the study by Turner and Holmes (2015), it was found that, as pH increased, so too did the adsorption of Cd, Co, Ni, and Pb. However, Cr saw a decrease in adsorption with increasing pH and Cu adsorption remained the same. Furthermore, beached pellets adsorbed more of the

Table 1
Selected references on particulate plastics made of synthetic polymers, their origin, and characteristics.

Chemical compound	Chemical formula	Origin or source to the environment	Characteristics	References
Polyolefins				
• Polypropylene	$(C_3H_6)_n$	Reusable food containers, bottle caps, drinking straws	Low density 0.90–0.91[a]	Engler (2012); Lassen et al. (2015); Shah et al. (2008); Vona et al. (1965); Wijesekara et al. (2018)
• Low density polyethylene	$(C_2H_4)_n$	Plastic bags, shrink wrap, general packaging, water pipes, garment bags	Low density 0.91–0.94[a] non-biodegradable, most common plastics	Engler (2012); Shah et al. (2008); Vona et al. (1965); Wijesekara et al. (2018)
• High density polyethylene	$(C_2H_4)_n$	Milk jugs, toys, plastic bags, detergent and oil bottles, cable insulation	High density 0.92–0.99[a], non-biodegradable	Nikles and Farahat (2005); Shah et al. (2008); Vona et al. (1965); Wijesekara et al. (2018)
Polystyrene	$(C_8H_8)_n$	Foam (i.e., "styrofoam"), CD crystal cases, service ware, blown foam packaging, disposable cups, packaging materials, laboratory ware, electronic uses	High density 1.04–1.13[a]	Engler (2012); Lassen et al. (2015); Shah et al. (2008); Vona et al. (1965); Wijesekara et al. (2018)
Acrylic	based with acrylic acid: CH_2 = CHCOOH	Used in textiles	High density 1.16[a]	Lassen et al. (2015); Shah et al. (2008); Vona et al. (1965)
Polyamide/ nylon	CO-NH	Small bearings, windshield wipers, water-hose nozzles, helmets, racehorse shoes, inks, rainwear	1.06–1.39[a] High durability and strength	Lassen et al. (2015); Shah et al. (2008); Vona et al. (1965)
Polycarbonate	$C_{16}H_{18}O_5$	Reusable beverage bottles, CDs, DVDs, street and car lights, sky-lights, baby bottles, roofs of greenhouses, glasses lens, water pipes	1.20[a]	Lassen et al. (2015); Shah et al. (2008); Vona et al. (1965)
Polyvinyl chloride	$(C_2H_3Cl)_n$	Pipes, building materials, shower curtains, car seat covers, raincoats, shower curtains, bottles, visors, shoe soles, garden hoses, electricity pipes	High density 1.39–1.43[a], non-biodegradable	Engler (2012); Lassen et al. (2015); Shah et al. (2008); Vona et al. (1965); Wijesekara et al. (2018)
Polyethylene terephthalate	$(C_{10}H_8O_4)_n$	Soda bottles, processed meat packages, peanut butter/jam jars, pillow and sleeping bag filling, textile fibres	High density 1.41[a]	Engler (2012); Lassen et al. (2015); Shah et al. (2008); Vona et al. (1965)
Polytetrafluorethylene	$(C_2F_4)_n$	Specialized chemicals, electronics, bearings, non-stick kitchen utensils	2.28–2.29[a]	Lassen et al. (2015); Shah et al. (2008); Vona et al. (1965)

[a] All units are g/cm^3.

Table 2
Trace-element concentrations associated with particulate plastics in the aquatic environment.

	Type of particulate particle	Al	Fe	Mn	Ag	Cd	Co	Cr	Cu	Hg	Ni	Pb	Zn	Reference
Fresh water (Watergate Bay)	Beached pellets	7430 *	34,400	712	< 3	5	13.8	42.5	47	< 3	29.3	109	196	Turner and Holmes (2015)
Seawater (Southwest England)	Virgin and beached PE pellets	16.9–55.8 **	41.9–97.8**	1.16–20.5**		1.09–76.7*	17.7–107*	44–751 *	0.064–1.32**		40.0–131*	0.149–1.64**	0.299–23.3**	Holmes et al. (2012)
River water	Virgin pellets***					0.0894	1.17	nd	1.58		0.282	0.922		Holmes et al. (2014)
	Beached pellets***					2.21	1.35	1.79	nd		2.58	13.2		
Seawater	Virgin pellets ***					0.00383	0.299	5.72	4.11		0.129	nd		
	Beached pellets ***					0.0904	0.717	8.48	nd		1.25	3.3		
Seawater (Portugal)	Virgin PS beads**								11.70				29.33	Brennecke et al. (2016)
	Aged PVC** fragments								3.11				6.12	
Coastlines (England)	Plastic production pellets (3–5 mm)													Ashton et al. (2010)
Soar Mill		7.05**	25.85**	1.58**	2.4*	1.7*	27*	19*	0.06**			0.15**	0.55**	
Thurlestone		9.44	35.67	1.28	4.9	10	25	63	0.14			0.41	0.42	
Bovisand		17.61	54.6	2.01	6	3.6	53	69	0.29			0.73	0.94	
Saltram		49.79	64.97	8.31	30	5	101	151	0.61			1.08	2.34	
San Diego Bay														Rochman et al. (2014)
Coronado Cays	PET (12 months)*	38,235.5	41,204	113,386.5		2	558.5	103.5			367	732.5	5802	
	HDPE	34,241	90,848.5	68,711		nd	584	134.5			73.5	534.5	6525.5	
Shelter Island	PET	27,154	45,440.5	28,048		nd	184	81.5			53	765.5	2743.5	
	HDPE	20,681	50,505.5	44,739		nd	491.5	96			254.5	925	3899	
Nimitz Marine Facility	PET	104,255	101,376	52,882		3.5	217.5	424			73.5	1419	6046	
	HDPE	17,052	49,516	21,173		nd	288	79			Nd	594	2573	

*ng/g; **μg g^{-1}; *** nmol g^{-1}; nd = not detected; PET = polyethylene terephthalate; HDPE = high density polyethylene; PS = polystyrene; PE = polyethylene; PVC = polyvinyl chloride.

metals than virgin pellets. In an additional study, particulate plastics showed interactions with metals in the surface of the Beijiang River littoral zone sediments (i.e., depth up to 2 cm), suggesting that most of the heavy metals carried by particulate plastics were derived from an inherent load (Wang et al., 2017).

The Langmuir or Freundlich models are commonly used to assess sorption or interactions between particulate plastics and contaminants. The Langmuir model is typically used in relation to surface adsorption in the crystalline regions of a polymer. Alternatively, the Freundlich model tends to focus on the internal partition into amorphous regions (Velez et al., 2018). However, it is important to realise that these models are simplified conceptual models and, therefore, may not be appropriate for all contaminants. A review by Velez et al. (2018) examined these models in relation to particulate plastics and hydrophobic organic contaminants. Because interactions between contaminants and particulate plastic are often complex, factors such as crystallinity of the plastic or concentration of the contaminant need to be considered when choosing an appropriate model. Although the study by Velez et al. (2018) examined hydrophobic organic contaminants, these factors should also be considered when examining the results for the sorption of other contaminants such as trace elements.

Compared to those for the aquatic environment, there have been limited scientific studies undertaken regarding particulate plastics and trace element interactions in the terrestrial ecosystem. According to Horton et al. (2017), the annual amount of plastic released to the land is estimated to be 4–23 fold larger than the plastics released to the oceans. Therefore, both the interactions and the consequences of particulate plastics and trace elements might be higher in terrestrial systems compared to aquatic systems. However, a preliminary study by Mersiowsky et al. (1999) found no significant contribution of polyvinyl chloride waste to concentrations of heavy metals in landfills, which might cause risk to aquatic environments.

Despite this, agroecosystems have been contaminated with particulate particles due to various agricultural practices and applications, such as the utilization of recycled organic waste (e.g., biosolids and compost) and plastic film mulching. Plastic film mulches are applied worldwide in modern agricultural practice and are considered to have many benefits (i.e., improved crop quality, prevention of evaporation, and a reduction in weeds and pests). However, an adverse effect of this application is the pollution caused by particulate plastics on agricultural lands. These particulate plastics can adsorb trace metals and other agrochemicals (Ng et al., 2018; Steinmetz et al., 2016).

A case in point is that of biosolids, the majority of which are applied on agricultural lands. As an example, 64% of biosolids are used for agricultural purposes in Australia. For every 1.5 million tonnes of wet biosolids, there are approximately 2800 to 19,000 t of particulate plastics dispersed in Australian agricultural soils (Ng et al., 2018). In general, biosolids contain high concentrations of trace elements, resulting in a high probability of their interaction with particulate plastics and thereby transporting them into various aquatic and terrestrial environments. Wijesekara et al. (2018) reported the concentrations of Cu, Zn, Mn, antimony (Sb), molybdenum (Mo), and Cd in biosolids derived particulate plastics as 180.64, 178.03, 71.33, 14.43, 2.01, and 2.34 ng/g, respectively. The concentrations of Ag, arsenic (As), Pb, and selenium (Se) attached to the same particulate plastics varied within the range of < 1–1.72 ng/g. Additionally, the mineral elements Al, calcium (Ca), Fe, potassium (K), and magnesium (Mg) were also found to be attached to particulate plastics with concentrations reported as 73,069, 1240, 91,502, 77,170, and 28,595 ng/g, respectively. A study by Bradney (2017) used a scanning electron microscope (SEM) coupled with Energy-Dispersive X-ray (EDX) to examine whether or not trace elements were adsorbed to pristine particulate plastic when exposed for two-weeks to biosolids. Trace elements were sorbed onto the particulate plastic (Fig. 3).

3.2. Factors affecting particulate plastics-toxic trace element interactions

Although plastic has no inherent charge or significant porosity, there are various environmental processes that can alter its composition. Weathering and surface modification, for example, provide favourable conditions for adsorption of metal ions (Holmes et al., 2014). There are several factors related to particulate plastics-trace element complexation. These factors can be divided into two sub-factors: 1) particulate-plastics and trace-element related factors and 2) environmental factors.

3.2.1. Particulate-plastics and trace-element related factors

Particulate plastics show a high affinity for DOM and, therefore, these particles can be commonly found attached to DOM in the environment. This is especially so considering the ubiquitous nature of DOM. Where these DOM-attached particles are present, there is likely to be more interaction and greater retention of trace elements than in the case of pristine particulate particles (Fig. 4). For example, using an isotherm study, Wijesekara et al. (2018) investigated the trace element (i.e., Cu) adsorption on to surface-modified particulate plastics with different organic matter. They found that the amount of adsorbed Cu in surface modified particulate plastics was significantly higher than the amount adsorbed to pristine particles (Fig. 5). Moreover, aged particulate plastics have shown a greater metal sorption due to the attachment of DOM and long-term pre-modification (e.g., photooxidation and attrition of charged materials) (Turner and Holmes, 2015).

The particulate plastic's base material also plays a significant role in the adsorption of trace elements. The trace-element adsorption rate can differ for each type due to various physical and chemical properties, such as surface area (affecting diffusivity) and hydrophobicity (affecting polarity). As an example, adsorption of Cu (released from antifouling paint) was notably lower on polystyrene than polyvinyl chloride. The differences between the ability of the two plastics to adsorb Cu were most likely a result of the smaller surface area of polystyrene (i.e., polystyrene's spherical shape compared to polyvinyl chloride's irregular shape) as well as polyvinyl chloride's polarity. Therefore, size is an important factor. Smaller particulate plastics have a larger specific surface area (surface area per unit mass) and in turn, a greater ability to sorb trace elements. Different porosity characteristics could be another reason for different adsorption. The polyvinyl chloride fragments accumulate Cu and Zn in greater amounts without reaching equilibrium, while virgin polystyrene beads reached constant Cu concentration at the end of 14 days (Brennecke et al., 2016). In contrast, according to Rochman et al. (2014) five different polymer types, namely polyethylene terephthalate, high-density polyethylene, polyvinyl chloride, low-density polyethylene, and polypropylene, showed a similar accumulation trend of trace elements in San Diego Bay, USA. This trend indicated that plastic type is less important in the accumulation of metals than the chemical or physical properties of the particulate plastics. In addition, during the 12 months of their study, the trace elements, Cr, Mn, Co, Ni, Zn, and Pb, did not reach equilibrium on any plastic type.

Besides this, plastic pellets found on beaches and virgin pellets have different adsorption capacities. Beached pellets show greater adsorption capacities than the virgin pellets (Holmes et al., 2012; Turner and Holmes, 2015), probably due to the increase of surface area and anionic active sites as a consequence of weathering and fouling with organic matter over time (Brennecke et al., 2016; Holmes et al., 2012; Holmes et al., 2014). Furthermore, polar functional groups, such as ketone, developed in the weathered plastics, bond with metals (Wang et al., 2017). Chemicals generally adsorb to the non-crystalline regions of the plastic polymers (Anderson et al., 2016). Acosta-Coley et al. (2019) found secondary particulate plastics contained higher detectable concentrations of trace elements (28 metals) than white pristine particulate plastics (7 metals). This study also demonstrates that weathering and degradation are important factors in trace element redistribution and

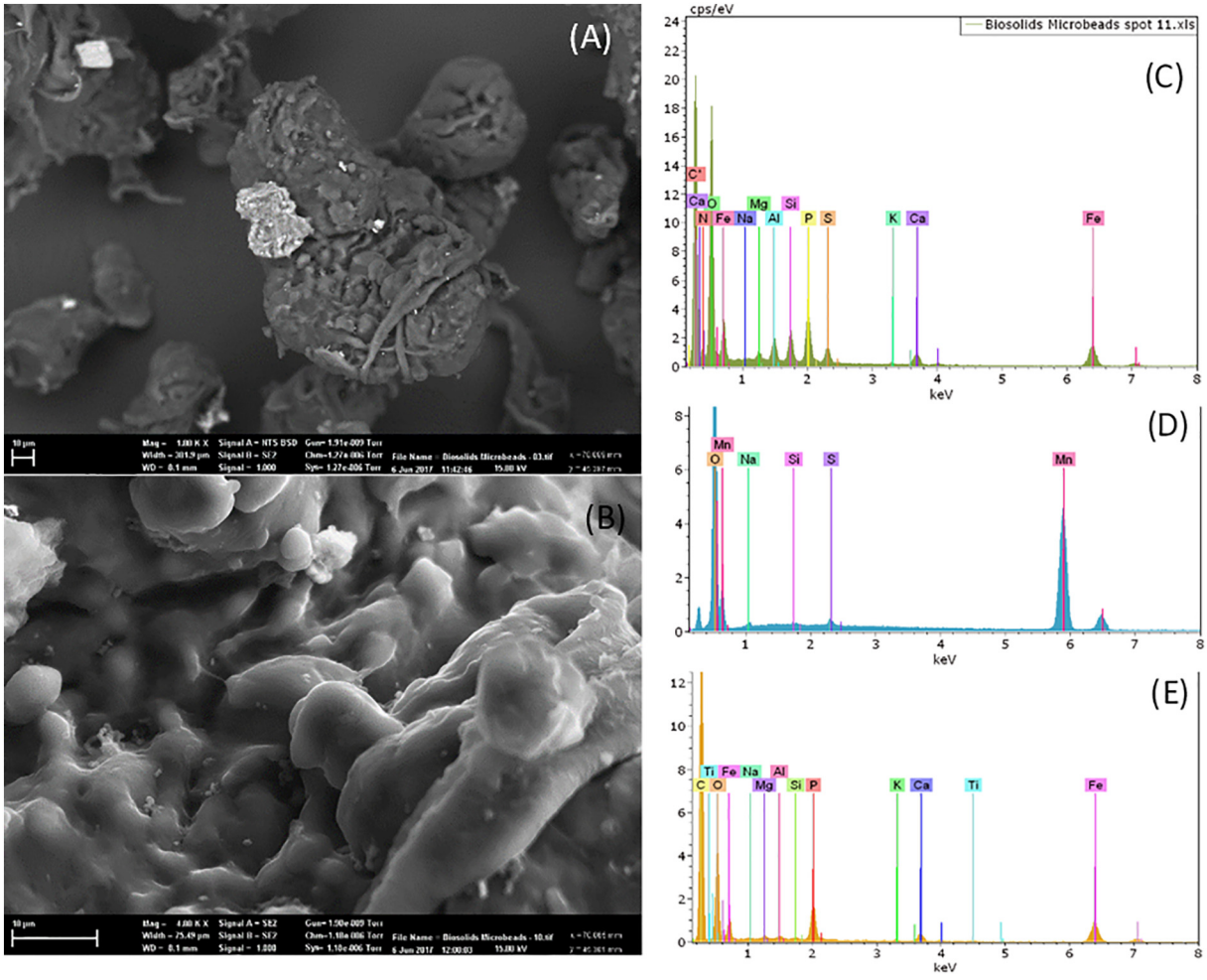

Fig. 3. Scanning Electron Microscope (SEM) and Energy Dispersive X-ray (EDX) showing trace metals sorbed to biosolids treated particulate plastics. Examples of SEM images (A) and (B); Examples of EDX spectrums (C), (D) and (E). Elements adsorbed onto biosolids treated particulate plastics were Al, C, Ca, Fe, K, Mg, Mn, N, Na, O, P, S, Si, and Ti.

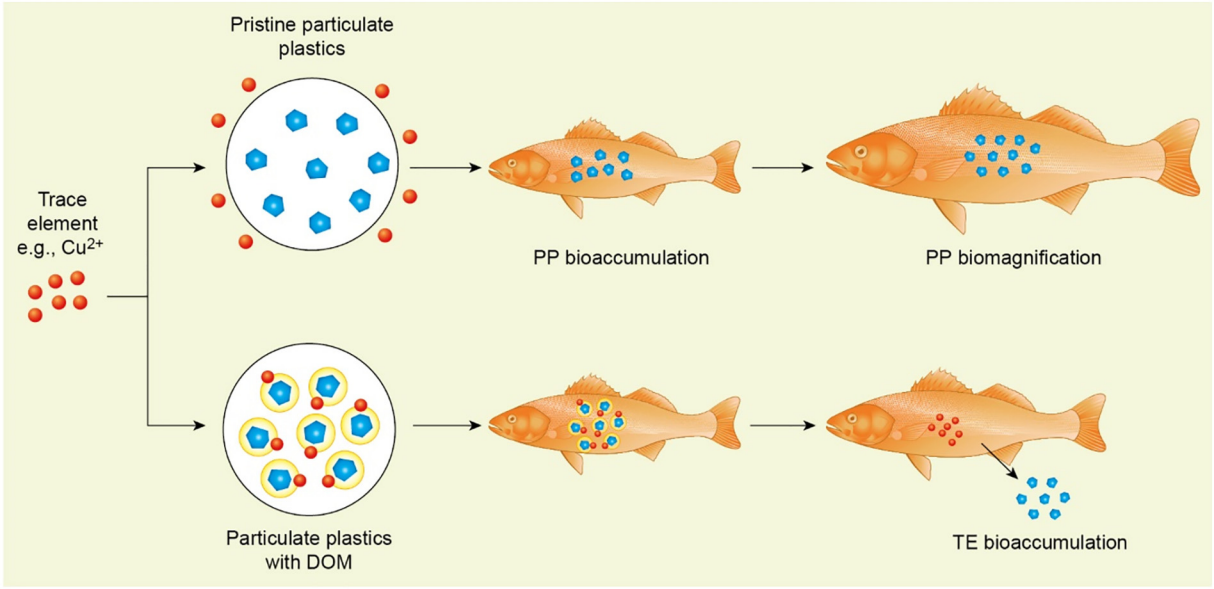

Fig. 4. Conceptual diagram illustrating the impact of particulate plastics (PP) derived trace elements (TE) to aquatic organisms (DOM = dissolved organic matter; Cu = copper).

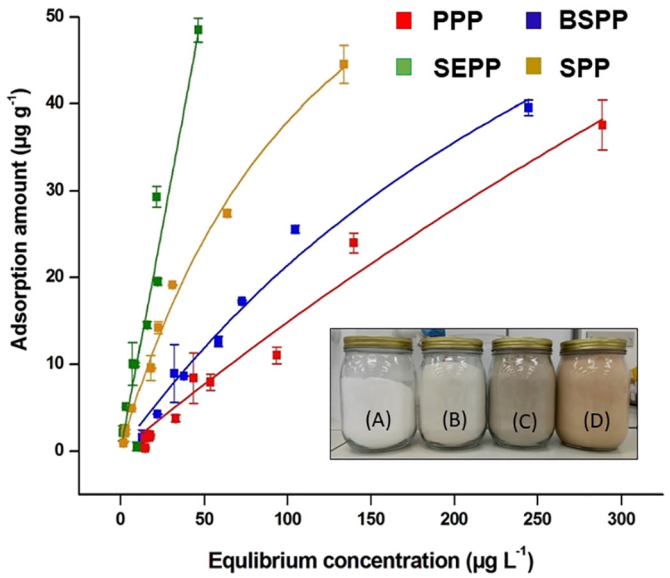

Fig. 5. Adsorption amount ($\mu g\, g^{-1}$) versus equilibrium concentration ($\mu g\, L^{-1}$) for pristine particulate plastics (PPP), biosolids sorbed particulate plastics (BSPP), sediment sorbed particulate plastics (SEPP), and soil sorbed particulate plastics (SPP). Inset image shows colour change of pristine particulate plastics caused by exposure to biosolids, sediment, and soil where (A) PPP, (B) BSPP, (C) SEPP, and (D) SPP (Reprinted from Wijesekara et al., 2018).

concentrations.

Furthermore, the surface charge of elements, trace-element oxidation status, and oxides are other factors that can affect trace-element adsorption. The exchange of trace elements depends on the concentration gradient between the particulate particles and the surrounding water (Anderson et al., 2016; Turner and Holmes, 2015). Based on K_D values (distribution coefficient where magnitude is independent of metal concentration), it was found that the trace elements Co, Cd, and Ni appeared to have adsorptions that were the most sensitive to the aging process (Turner and Holmes, 2015).

3.2.2. Environmental factors

Environmental factors also play an important role in trace-element adsorption by particulate plastics. For many metals, pH is highly influential for particulate plastic and metal adsorption. According to Turner and Holmes (2015), for both virgin and aged pellets, the adsorption of Cd, Co, Ni, and Pb was higher in river water, whereas Cr adsorption was higher in seawater (where pH of seawater is greater than that of river water). Furthermore, adsorption of Ag, Cd, Co, Ni, Pb, and Zn increased proportionately as the pH increased in the river water. With increasing pH, metal cations interacted consistently with the plastic, decreasing the relative abundance of them in solution. In contrast, Cr adsorption was reduced, and there were no significant effects on Cu or Hg with increasing pH. Adsorption of Cr was reduced due to Cr remaining as Cr^{6+}, and it existed in oxyanionic forms ($HCrO_4^-$ and CrO_4^{2-}), which interacted relatively weakly with positively charged regions of beached pellets.

The passage of time also plays an important role in metal adsorption. A study done by Rochman et al. (2014) revealed that, over a 12 month period, concentrations of all metals adsorbed onto plastics increased. This indicated a positive relationship between trace element adsorption and the length of time the plastic debris remained in the sea.

Plastic debris in rivers is less likely to be weathered and chemically changed (due to non-tidal and unidirectional flow) than the marine-derived material, which is usually suspended or has been beached for a considerable time. Estuaries may act as mixing zones for new and aged particles and may be a place where adsorption properties of plastics can be changed, which, in turn, affect the volume uptake of trace elements (Holmes et al., 2014). As a result, estuaries represent important potential sources and sinks of contaminated particulate plastics. The concentrations of contaminants in estuaries can be greater than those in riverine or marine waters (Anderson et al., 2016).

Developed biofilms in the environment can facilitate the accumulation of trace elements in particulate plastics. This is due to a change in the polymer surface that is immersed in natural waters (Rochman et al., 2014; Turner and Holmes, 2015). The sorptive properties of biofilms facilitate the chelation of organic compounds and trace elements. They are then transferred into food webs (Decho, 2000). Plastic debris provides a substrate for microbes that exist for a much longer time than most natural floating substrates, providing more time for the accumulation and transportation of harmful substances (Zettler et al., 2013). It has been proposed that biofilm constituents do not differ greatly across different types of plastics. This could point towards biofilm impacts having greater significance than any correlation between metal accumulation and different types of plastics (Rochman et al., 2014).

3.3. Fate of particulate plastics-toxic trace element complexes in the environment

The accumulation of particulate plastics and trace elements in aquatic and terrestrial environments is an emerging concern. Research has focused predominantly on trace element adsorption to marine particulate plastics rather than adsorption onto particulate plastics found in freshwater and terrestrial environments. Therefore, there is limited information regarding the environmental impacts and consequences (such as bioaccumulation, reaching food webs) associated with these environments (Wagner et al., 2014). The effects of particulate plastics and trace element interactions in freshwater and marine organisms are hardly known. Yet, their long-term consequences may be greater than first anticipated. As mentioned before, particulate plastics act as vectors for carrying trace elements and they may increase ingesting organisms' chemical exposure and the resultant toxicity (Wagner et al., 2014). Surprisingly, at least 44% of marine bird species are known to have ingested plastics (Andrady, 2011), resulting in many adverse effects on marine food webs. Furthermore, the shore crab (*Carcinusmaenas* sp.) was also found to take up particulate plastics (Watts et al., 2014). It was found that uptake occurred via the ventilation route, with this being a common uptake route in marine nonfilter feeding species. However, this process is not restricted to marine species alone. A freshwater species, zebrafish (*Danio rerio*), accumulated 5 µm diameter particulate plastics in its gills, liver, and gut after seven days of exposure. This caused many detrimental effects to internal body functions such as inflammation, lipid accumulation in the liver, and alterations of metabolic profiles (Lu et al., 2016). The ingested particulate plastics can affect the metabolism and induce inflammation in different species (Lu et al., 2016; Wagner et al., 2014). Taking into account these freshwater and marine organism studies, the results are highly indicative of trace-element uptake via particulate plastics. It is, therefore, reasonable to conclude that further investigation and data collection in this area are warranted.

The above-mentioned studies pose further implications for agroecosystems, where there is a lack of available evidence on the ecological impacts of particulate plastics and trace elements. It is important to address this issue in regard to supplying safe food to human populations (Ng et al., 2018). Studies have also examined the uptake of particulate plastics by plants. Bandmann et al. (2012), using cell culture techniques, showed that BY-2 cells (i.e., tobacco (*Nicotiana tabacum*) Bright Yellow 2 cells) were capable of taking up 20 and 40 nm polystyrene beads. However, there are no data available on toxicity, translocation, and storage of nanoplastics in plants (Ng et al., 2018).

The ingested particulate plastics can transfer pollutants and additives to terrestrial animals. Particulate plastic, for example, transfer pre-adsorbed pollutants and chemicals into the gut tissues of lugworms (*Arenicola marina*), initiating some biological effects. High

concentrations of particulate plastic and additives can harm the eco-physiological functions of these organisms (Browne et al., 2013). Furthermore, earthworms act as transport agents for particulate plastics by facilitating their passage to soils via their casts and burrows. As a result, the other soil biota can be exposed to particulate plastics once this material is incorporated into the soil (Rillig et al., 2017). The survival and fitness of the earthworms Lumbricusterrestris (*Oligochaeta, Lumbricidae*) were measured after exposing them to different concentrations of a particulate plastic (i.e., polyethylene < 150 μm). A correlation was found between an increasing mortality rate and an increase in the number of particulate plastic present. Also, as the particulate-plastic concentrations increased, the earthworm's growth rate was significantly reduced (Huerta Lwanga et al., 2016). These studies show the importance of particulate plastics in soil biota.

The above-mentioned plant and organism studies show the mobility of particulate plastics in different species in both terrestrial and aquatic ecosystems. Thus, there is a high probability of trace-element uptake with these particulate plastics. Increase uptake creates a higher risk of bioaccumulation of trace elements or their release back into the environment in a more soluble and biologically available form (Holmes et al., 2012). Therefore, it is important to have collaboration between different disciplines to fulfil this knowledge gap.

4. Particulate-plastic assisted toxic trace-element uptake by aquatic and terrestrial organisms

Studies of particulate plastics in different environments are rapidly advancing, with particulate plastics found across a range of environments worldwide, (i.e., atmospheric, terrestrial and aquatic ecosystems, and polar regions) (Li et al., 2018; Wijesekara et al., 2018). Ingestion of particulate plastics by organisms ranging in size from microscopic ones to large vertebrates has been documented in recent years. These studies emphasize the health risks of particulate plastics. Risks arise from the ingestion of particulate plastics associated with the chemicals themselves and the higher surface area to volume ratio, because this may cause particulate plastics to release some toxic chemical additives and adsorbed pollutants upon ingestion (Rochman et al., 2013; Wright et al., 2013). Consequently, the released toxic compounds become bioavailable and finally end up in human diets via food chains (Rochman et al., 2015b). However, the bioavailability and the accumulation of particulate-plastic derived toxic elements in organisms mainly depend on the surrounding environment, physiological features, and feeding habits of an organism along with the physiochemical properties of the contaminant (Akhbarizadeh et al., 2018).

Particulate plastics can act as a carrier for trace elements, such as Cd, Co, Cr, Cu, Ni, and Pb, owing to their high adsorption capacity (Holmes et al., 2014). In addition, trace elements present on the surface of particulate plastics can also originate from the manufacturing process (i.e., additives) rather than from the environment (Wang et al., 2017). Eventually, uptake of these particulate plastics may lead to the development of metal toxicity in both aquatic and terrestrial wildlife. For instance, ingested metals caused reproductive perturbation and physiological changes in the gut environment of fish species, causing cytotoxic damage (Khan and McGeer, 2013; Khan et al., 2015). So far, there are few studies available in the literature on particulate-plastic assisted toxic trace-element uptake and its impacts on aquatic and terrestrial organisms. Table 3 summarizes some selected studies on the occurrence and impacts of particulate-plastic-derived trace elements in aquatic and terrestrial organisms.

4.1. Ingestion and accumulation by aquatic organisms

The alarming concentration of particulate plastics in aquatic ecosystems has been found to adversely affect aquatic organisms such as fish, freshwater invertebrates, oysters, and lugworms (Besseling et al., 2013; Blarer and Burkhardt-Holm, 2016; Cole and Galloway, 2015; Lusher et al., 2017). The size, colour, density, and abundance of particulate plastics are the key factors that make them bioavailable to aquatic organisms (Wright et al., 2013). Especially due to their small size, these materials can be readily ingested and accumulated in the tissues, brain, and circulatory system of organisms creating severe health defects (Auta et al., 2017). Yet, direct or indirect adverse effects of particulate plastics to aquatic species are not fully understood (Murphy and Quinn, 2018; Rochman et al., 2013). However, one of the possible reasons for adverse effects could be exposure of aquatic organisms to toxic trace elements associated with particulate plastics after the ingestion of the plastics.

The size of particulate plastics influences how easily organisms ingest them, with smaller marine organisms, e.g., zooplankton, demonstrating size selectivity. One reason for this may be due to the organism's mouth capacity and body size (Botterell et al., 2019). Cole et al. (2013) found that *Acartia clausi* and *Calanus helgolandicus* both showed a preference for the smallest plastic particles (i.e., 7.3 μm) when compared to the larger particles (i.e., 30.6 μm). The life-stage of the decapod species *Brachyurans*, was shown to influence the size of plastics ingested. *Brachyuran zoea* (less developed) did not ingest 20.6 μm particulate plastics, while *Brachyuran megalopa* (more developed) did (Cole et al., 2013).

Akhbarizadeh et al. (2018) studied the relationship between particulate plastics and potentially toxic elements in fish muscles of pelagic and benthic fish species in the northeast of the Persian Gulf. They observed maximum concentrations of As, Fe, Hg, and vanadium (V) in *Sphyraena jello*, whereas *Epinephelus coioides* exhibited maximum levels of Cr, Ni, Se, and Zn in their fish muscles. *Alepes djedaba* showed the highest concentration of Cu and Pb while *Platycephalus indicus* displayed the highest Mn level. In the case of particulate plastics, *P. indicus* exhibited the highest concentration with the mean value of 18.50 items/10 g fish muscle, while *S. jello* showed the lowest concentration with the mean value of 5.66 items/10 g fish muscle. In addition, the average concentrations of particulate plastics in *E. coioides* and *A. djedaba* were 7.75 and 8.00 items/10 g fish muscle, respectively. However, positive relationships between particulate plastics and metals in fish muscles were seen only for *A. djedaba* and *P. indicus*, while *E. coioides* exhibited a negative effect. Moreover, there was no significant relationship between particulate plastics and metals for *S. jello* (Akhbarizadeh et al., 2018). Because there was a positive correlation between particulate plastics and metals for *A. djedaba* and *P. indicus*, the highest concentration of Cu, Pb, and Mn in fish muscles could have been derived from the particulate plastics.

Holmes et al. (2012, 2014) observed enhanced adsorption of Cd, Co, Cr, Cu, Ni, Pb, and Zn into both virgin and beached particulate plastics in aqueous environments. Moreover, Rochman et al. (2014) reported that different types of plastics tended to accumulate elevated concentrations of metals (such as Al, Fe, and Cd) when they exist for a long time in the sea. In addition, several other studies also reported that high concentrations of major and trace elements can adhere to and become entrapped with particulate plastics, which act as vectors for metal transportation in both marine and freshwater ecosystems (Ashton et al., 2010; Brennecke et al., 2016; Rochman et al., 2014). Consequently, on the one hand, one of the reasons for the positive correlation between particulate plastics and metals in fish muscles in the aforementioned study could be due to the bioavailability of adsorbed metals on surfaces of particulate plastics. On the other hand, both major and trace elements can exist in plastics as an inherent load due to the incorporation of different additives (i.e., pigments, lubricants, stabilizers, and catalysts) during plastic production (Hahladakis et al., 2018). Eventually, dynamic changes in the aqueous, as well as in the gut, environments may lead to the release of the inherent metal load and thereby become bioavailable for organisms (Brennecke et al., 2016; Silva et al., 2018; Wang et al., 2017).

In contrast, Davarpanah and Guilhermino (2015) reported that there was no significant impact on the average specific growth rate of

Table 3
Selected references on the occurrence and impacts of plastic-derived trace elements in aquatic and terrestrial organisms.

Species	Association between particulate plastics and trace elements in the organisms	Reference
A. djedaba *P. indicus* *E. coioides* *S. jello* (Fish spp.)	Positive relationship between particulate plastics and metals in fish muscles for *A. djedaba* and *P. indicus* Negative effect for *E. coioides* No significant relationship for *S. Jello*	Akhbarizadeh et al. (2018)
Danio rerio (zebrafish) [4 and 24 h exposure of MP]	Co-exposure of particulate plastic beads and Ag had no effect on the uptake or localization of Ag in zebrafish Exposure to the Ag-incubated particulate plastic beads (~75% of the Ag bound to particulate plastic beads) significantly reduced Ag uptake and significantly increased the proportion of intestinal Ag	Khan et al. (2015)
Oncorhynchus mykiss Rainbow trout fish [3 h exposure]	Co-exposure of particulate plastics and Ag and Ag-incubated particulate plastics had no effect on the uptake of Ag in the anterior/mid intestine of rainbow trout	Khan et al. (2017)
Marine microalgae (*Tetraselmis chuii*) [96 h of exposure of MP to concentrations up to 1.472 mg/L)]	Exposure of various particulate plastic concentrations had no significant effects on average specific growth rate of *T. chuii* in the presence or absence of Cu No interaction between particulate plastic and Cu to cause Cu toxicity in *T. chuii*	Davarpanah and Guilhermino (2015)
American black ducks (*Anas rubripes*), mallards (*A. platyrhynchos*), Common eider (*Somateria mollissima*) (Coastal waterfowl)	Plastic were present in the stomachs of 46.1% (6/13) of mallards, 6.9% (6/87) of black ducks, and 2.1% (1/48) of eiders Metals were present in the stomachs of 30.8% (4/13) of mallards, 2.3% (2/87) of black ducks, and in 2.1% (1/48) of eiders	English et al. (2015)
Dovekies (*Alle alle*) (Bird spp.)	Plastic debris in 9 of 65 (14%) of gizzards samples Greater hepatic Hg levels in dovekies feeding at higher trophic levels	Fife et al. (2015)
Laysan albatross (*Phoebastria immutabilis*) Bonin petrel (*Pterodroma hypoleuca*) (Seabirds)	Increased levels of ingested plastic caused an increase the concentrations of chlorine, iron, lead, manganese, and rubidium in feathers	Lavers and Bond (2016)
Flesh-footed shearwaters (*Puffinus carneipes*) (Seabirds)	High concentrations of chromium and silver were positively related with the mass of ingested plastic	Lavers et al. (2014)
Earthworms (*Lumbricus terrestris*)	Particulate plastics could increase Zn bioavailability *Lumbricus terrestris* ingested the Zn-bearing microplastic	Hodson et al. (2017)
Soil microorganisms	Reduced soil respiration (up to ~26%) and dehydrogenase activity (up to ~ 39%), due to the microbead-metal complexes	Wijesekara et al. (2018)

Tetraselmis chuii (marine microalgae) after exposure to various particulate plastics in the presence or absence of Cu. Furthermore, the results did not show any interaction between particulate plastics and Cu that would cause Cu toxicity in *T. chuii*, even though Cu alone significantly reduced the average specific growth of the microalgae (Davarpanah and Guilhermino, 2015). The study suggested that the Cu bioavailability was decreased due to the binding of Cu to the particulate plastics. Nevertheless, the reduced metal toxicity in the presence of particulate plastics could be attributed to the lower uptake or non-uptake of the resultant Cu-plastic complexes by organisms. However, the experiment was conducted under laboratory conditions with 96 h of exposure of particulate plastics, which had a maximum concentration of 1.472 mg/L of Cu. Thus, long-term exposure of microalgae to particulate plastics and associated Cu needs to be further examined to understand the underlying mechanisms of Cu bioavailability and toxicity in the presence of particulate plastics.

Similarly, Khan et al. (2017) reported that the co-exposure of particulate plastics and Ag, and Ag-incubated particulate plastics, had no effect on the uptake of Ag in the intestine of rainbow trout fish. Furthermore, Khan et al. (2015) also documented that there was no effect on uptake or localization of Ag in zebrafish with the co-exposure of particulate-plastic beads and Ag. However, remarkably reduced Ag uptake and an elevated amount of intestinal Ag were observed with the exposure of Ag-incubated polyethylene particulate plastic beads (Khan et al., 2015). A study by Bradney (2017) examined the effect of Cu-sorbed particulate plastics on *Daphnia magna* mobility. It was found that Cu caused 100% immobilisation (i.e., mortality) in 50 mg/L of Cu-sorbed particulate plastic after 24 h exposure, suggesting the occurrence of Cu toxicity to *Daphnia magna* (Fig. 6).

4.2. Ingestion and accumulation by terrestrial organisms

The effect of particulate plastics on terrestrial organisms is poorly understood. Hodson et al. (2017) observed that earthworms could ingest Zn-bearing particulate plastics, and the particulate plastics increased the Zn bioavailability. However, they found that there was no significant difference in weight change, Zn accumulation, and mortality between the earthworms that were exposed for 28 days to Zn-laden particulate plastics and those that were not exposed to Zn-laden particulate plastics. Differences in these results may be attributed to many factors, including the size of the particulate plastics and the nature of the vector compounds. These results indicate that further research is needed to better understand the vector effect of particulate plastics. More research also is needed to study a wide range of particulate plastics and potentially different toxic trace elements. In addition, Hodson et al. (2017) reported that particulate plastics can adsorb metals in the soil, and those plastics can be ingested by earthworms. The study further revealed that particulate plastics can act as vectors for metal exposure in soil invertebrates. Concerns about the ecotoxicity of particulate plastics to soil microbial activities have been studied by Wijesekara et al. (2018). The study documented that Cu-sorbed particulate plastics reduced soil respiration and dehydrogenase activity by ~ 26% and ~ 39%, respectively, compared to the control. The reduced microbial activities were mainly due to the formation of particulate plastics-metal complexes in the studied soil (Wijesekara et al., 2018). Furthermore, a study by Bradney (2017) also examined the effect of Cu-sorbed particulate plastics on microbial respiration. Overall, it was found that soil basal respiration decreased when the soil was treated with Cu-sorbed particulate plastic, suggesting the occurrence of Cu toxicity to soil microbial activity. This demonstrates the need for further research on the effects that metal-sorbed particulate plastics have on terrestrial environments, particularly regarding processes that affect soil health (Fig. 6).

The occurrence of particulate plastics in marine ecosystems creates a great threat to the entire ecosystem, including both aquatic and terrestrial organisms. In the past decade, many studies have shown that

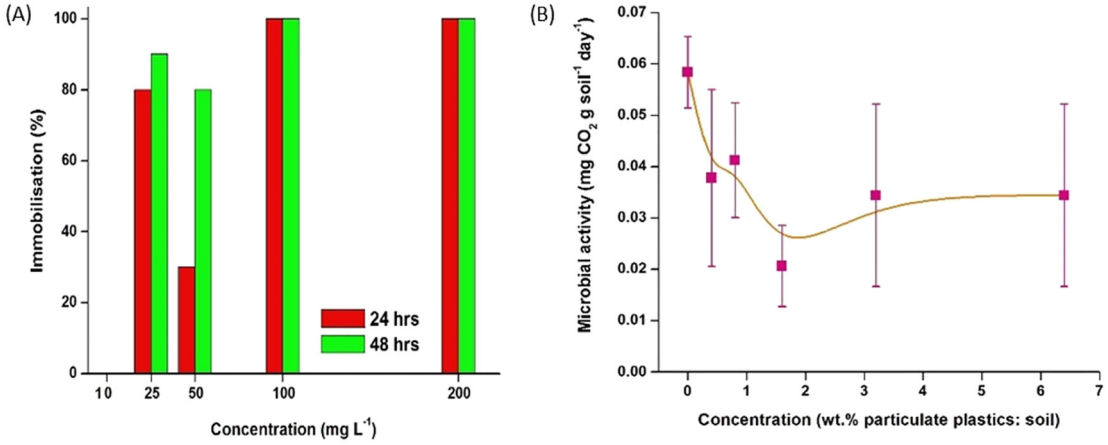

Fig. 6. Impacts of copper sorbed particulate plastics on *Daphnia manga* and microbial respiration. (A) shows the immobilisation (%) of daphnids when exposed to copper sorbed particulate plastics for 24 h (red) and 48 h (green). (B) shows the microbial activity (mg CO_2 g soil^{-1} day^{-1}) for basal respiration plotted against copper sorbed particulate plastics concentration (wt% (particulate plastics:soil)) where error bars show standard deviation. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

terrestrial birds around aquatic environments (especially sea birds) were largely affected due to the passive or mistaken ingestion of plastic debris (Auta et al., 2017; Zhao et al., 2016). For instance, English et al. (2015) observed plastic and metal debris in the stomachs of three different coastal waterfowls in Atlantic, Canada. Among the studied waterfowls, plastic debris was present in 46.1% of mallards, 6.9% of black ducks, and 2.1% of eiders, while 30.8% of mallards, 2.3% of black ducks, and 2.1% of eiders stomachs contained metal debris (English et al., 2015). Beyond the adverse effects of plastic debris itself, a few studies have shown the adverse effects among bird species due to the uptake of toxic trace elements via ingestion of plastics. Fife et al. (2015) studied trace elements and ingested plastic debris in dovekies, an Arctic seabird in White Bay, Newfoundland, Canada. They observed plastic debris in 9 gizzard samples among 65 samples, while, at the same time, they observed a high level of hepatic Hg in the dovekies which feed at high trophic levels. Hence, the study indicated that particulate plastics in Arctic marine food webs pose threats to marine biota as well as to humans who rely on different trophic levels.

An experiment was carried out by Lavers and Bond (2016) to evaluate the vector effect of ingested plastics for trace element content within the stomach and feathers of seabirds. The study found that 25–100% of fledglings of the Laysan albatross (*Phoebastria immutabilis*) and Bonin petrel (*Pterodroma hypoleuca*) have exceeded international threshold levels for ingestion of plastics by seabirds. Additionally, a greater amount of ingested plastics caused an increase in levels of Fe, Pb, Mn, and rubidium (Rb) in the feathers of juvenile Laysan albatrosses and Bonin petrels. Similarly, Lavers et al. (2014) also observed a positive relationship between the ingested plastics and the level of Cr and Ag content in the stomach and feathers of the flesh-footed shearwater (*Puffinus carneipes*) fledglings in Lord Howe Island, New South Wales, Australia. Moreover, high levels of ingested plastics and toxic contaminants in birds caused a reduction in their body mass, head bill length, and wing chord, indicating a negative impact on seabird populations in Australian marine environments (Lavers et al., 2014). Furthermore, > 60% of flesh-footed shearwater fledglings have exceeded international threshold levels for plastic ingestion by seabirds, while 16% of fledglings exceeded the threshold limits with a single feeding.

5. Bioavailability of ingested particulate-plastic-derived toxic trace elements

As mention in Section 3.1, particulate plastics can adsorb contaminants, e.g., trace elements, onto their surface. The particulate plastics may then transport these contaminants to marine, freshwater, or terrestrial environments (Sleight et al., 2017). Although research has proven that the presence of particulate plastics and associated contaminants can impact organisms, there is limited information surrounding their bioavailability (Sleight et al., 2017). Therefore, to better understand the bioavailability of toxic trace elements, research should focus not only on the adsorption of contaminants onto the particulate plastic's surface but also their ability to desorb upon entering an organism (Sleight et al., 2017).

5.1. Factors affecting bioavailability

Bioavailability refers to a compound's ability to either adsorb onto or to cross an organism's biological membrane. Once exposed, the compound may cause changes to that organisms' biological molecules (McGeer et al., 2004; Semple et al., 2004; Sleight et al., 2017). Bioavailability of chemicals has long been used in research as a way to better understand a chemical's fate, transport, and environmental impact, thereby helping to aid decisions regarding a chemical's risk assessment (Cipullo et al., 2019). It is therefore important that research concerning particulate plastics also focuses on the bioavailability of contaminants such as toxic trace elements. This will help to enhance our understanding surrounding the impacts (both long and short term) of interactions of particulate plastics and trace elements within organisms (Sleight et al., 2017).

A lack of research surrounding bioavailability may well be due to the difficulties associated with accurately determining the effects (Sleight et al., 2017). A range of interacting factors influence the bioavailability of contaminants (including toxic trace elements) within organisms. These include variables such as temperature, the type of metal or plastic involved, chemical interactions, pH, salinity, as well as variables more specific to the organism, e.g., digestive system functions (Mota, 2017; Sleight et al., 2017).

A study conducted by Kedzierski et al. (2018) found that aged polyvinyl chloride adsorbed Cu and Ag. It was observed that degradation occurred through the formation of chips from the polyvinyl chloride surface (possibly due to plasticiser desorption). This caused an increase in surface area, which may have contributed to a greater adsorption of metals. Additionally, Brennecke et al. (2016) found that aged polyvinyl chloride adsorbed Cu and Zn. Both metals increased in concentration with time, which was potentially caused by the presence of DOM and greater reactivity on the surface. However, Brennecke et al. (2016) also looked at virgin polystyrene and found that, although Cu increased significantly over time, Zn concentration did not. Because the

polystyrene was spherical, it did not have as large a surface area. Therefore, this research suggests that both the plastic type and surface area (and therefore age) are important factors for adsorption and, in turn, bioavailability (Brennecke et al., 2016; Kedzierski et al., 2018). Furthermore, size plays an important role in the bioavailability of particulate plastics and associated trace elements as it influences their translocation into the tissues of organisms. Nanoplastics can passively cross the cell membrane, whereas microplastics would have to undergo active transport (Triebskorn et al., 2019).

The hydrophobic nature of plastics also plays a role in determining whether or not a chemical will become strongly adsorbed onto the surface of a particulate plastic. This, in turn, influences its bioavailability, especially if the particulate plastic is too large to cross the cell membrane. A hydrophobic contaminant will adsorb more strongly to a particulate plastic than a contaminant that is hydrophilic (Bakir et al., 2012; Sleight et al., 2017; Teuten et al., 2009). However, because hydrophobic compounds are more strongly adsorbed, they are less likely to desorb than those that are hydrophilic. This potentially means that, despite ingestion, the contaminants may not become bioavailable and, therefore, bioaccumulate in the body and tissues of organisms.

5.2. Bioavailability within organisms

To understand the effects of particulate plastics, it is important to understand the 'vector-effect,' which looks at how the behaviour of contaminants (e.g., trace elements) is altered when adsorbed to particulate plastic. The behaviour occurs at three different levels (Khan et al., 2015; Syberg et al., 2015). The first level or 'environmental-vector' is where contaminants are transported to a new environment (Khan et al., 2015). The second level or 'organismal-vector' involves a contaminant that is ingested by an organism when adsorbed to particulate plastics and, therefore, is subject to dietary uptake (Khan et al., 2015). Lastly is the 'cellular-vector' where contaminants sorbed to particulate plastics are taken into cells, which results in bioavailability and bioaccumulation (Khan et al., 2015).

A study conducted by Khan et al. (2015) compared the uptake of Ag in zebrafish when exposed to no particulate plastic beads, pristine particulate plastic beads, and Ag-sorbed plastic-beads. It was found that there was no significant change in Ag uptake when particulate-plastic beads were simply present (pristine) in the experiment. However, Ag-sorbed plastic-beads did have a significant effect with results showing a decrease in Ag uptake and an increase in intestinal Ag. Khan et al. (2015) suggested that the particulate plastics acted as an 'organismal-vector' where the accumulation of Ag occurred through the dietary route. Thus, particulate plastics did not increase the bioavailability of Ag in organisms. However, the author did provide some factors that may have altered the outcome of the study. He suggested that Ag aggregated on the water's surface (thereby decreasing adherence time), and, as well, the density of the plastics changed, and these factors affected the bioavailability. Overall, this study demonstrates that particulate plastics do have the ability to alter the route and uptake of trace elements (e.g., Ag) given the right circumstances. However, the effects that occur are dependent on the extent to which trace elements are adsorbed onto the particulate plastics.

A study conducted by Besseling et al. (2013) found that the bioaccumulation of polychlorinated biphenyls (PCBs) within lugworms increased when exposed to low concentrations of polystyrene (0.074%). However, although an increase was observed, it only corresponded to an increase by a factor of 1.1 to 1.5. Alternatively, a study conducted by Sleight et al. (2017) saw a decrease in phenanthrene (Phe) and 17α-ethinylestradiol (EE2) bioaccumulation in larval zebrafish when exposed to particulate plastics. This decrease in bioavailability was due to the sorption of Phe and EE2 onto the particulate plastics. Furthermore, a study conducted by Mota (2017) examined the bioavailability of metals in the fish *Diplodus sargus* and the decapod crustacean *Palaemon serratus* (juveniles), when exposed to pristine and coated (anti-fouling paint) polystyrene. The metals in the anti-fouling paint were Cu and Zn. Results showed that, although plastic was ingested by the fish, there were no adverse effects due to ingestion. Additionally, neither organism showed any metal toxicity indicating the metal bioavailability did not increase significantly enough to cause adverse effects within the organisms. It was, therefore, suggested that the organisms may have a higher tolerance to these metals as they are essential to both species.

The lack of research surrounding the bioavailability of contaminants may be due to difficulties surrounding the detection limits of experimental equipment (Sleight et al., 2017). A review undertaken by Lanctôt et al. (2018) suggested that the use of radiolabelled particulate plastics may be used to better understand the fate of plastic particles upon entering an organism. Lanctôt et al. (2018) suggested that this method may have the ability to provide information relating to the different uptake routes of particulate plastics upon ingestion. The method also can reveal the retention time of the plastics in the gut and whether or not the particle has been assimilated. Overall, the method may help understand if particulate plastics play a role in the bioavailability and bioaccumulation of contaminants such as trace elements.

6. Trophic transfer of particulate plastics

Particulate plastics have been shown to undergo trophic transfer in the marine food web, e.g., zooplankton, shrimps, fish, crabs, and sea lions (Carbery et al., 2018; Cedervall et al., 2012; Farrell and Nelson, 2013). Cedervall et al. (2012) demonstrated the transfer of particulate plastics from algae to zooplankton to fish. Additionally, Farrell and Nelson (2013) found crabs contained low levels of plastic in their haemolymph stomach, hepatopancreas, ovary, and gills after ingesting mussels that had been exposed to particulate plastic. Therefore, primary consumers (e.g., zooplankton and mussels) could be a pathway for particulate plastics in higher order secondary organisms (Avio et al., 2017).

Unfortunately, studying the trophic transfer of particulate plastics is a complex issue. This is due to the difficulties in accurately determining the source of the particulate plastics within the environment (i.e., direct ingestion or trophic transfer). Laboratory studies have demonstrated trophic transfer in small organisms, but ethics prevents the study of larger organisms. Furthermore, some trophic transfer may occur between environments, e.g., fish could transfer particulate plastics to birds (Rutkowska et al., 2018). This could not only lead to plastic contamination but also trace element contamination. Studies have shown positive relationships between the concentration of plastic and trace elements (i.e., Ag, Cl, Cr, Fe, Pb, Mn, and Rb) in juvenile Laysan albatrosses, Bonin petrels, and flesh-footed shearwater (*Puffinus carneipes*) fledglings (Lavers et al., 2014; Lavers and Bond, 2016).

The trophic transfer of particulate plastics in terrestrial environments is not as widely studied. However, there is evidence that particulate plastics can enter the terrestrial food web. Huerta Lwanga et al. (2017) observed an increase in microplastic concentration from the soil, to earthworm casts, and finally to chicken faeces. Particulate plastic concentrations increased from 0.87 particles/g (soil) to 129.8 particles/g (chicken faeces). The evidence surrounding trophic transfer in organisms is concerning. It not only demonstrates an additional indirect source of particulate plastics for organisms but also raises concerns for impacts on human and environmental health (Carbery et al., 2018).

7. Human health risks of particulate-plastic-derived toxic trace elements

The consequences surrounding human's exposure to particulate plastics are largely unknown. Research has predominately focused on marine organisms rather than human health (Kontrick, 2018). However, this lack of research is concerning, because particulate plastics have the potential to be found in seafood. Thus, they could make their way up the food chain, where they could eventually be consumed by

humans (Akhbarizadeh et al., 2018).

7.1. Source of human exposure to particulate plastics

7.1.1. Seafood

Particulate plastics have been found in seafood sold for human consumption (Akhbarizadeh et al., 2018; Karami et al., 2017; Van Cauwenberghe and Janssen, 2014). A study conducted by Karami et al. (2017) examined four dried fish species for particulate plastics. Each fish examined contained between zero to three particulate-plastic particles. In total, they found 36 particulate-plastic particles in the fish that they examined. The abundant plastic type was found to be polypropylene (47.2%), followed by polyethylene (41.6%). Of the 36 particles, 29 were located in the flesh. This shows that plastic is present within the edible parts of fish, meaning that humans may ingest them. Furthermore, it was found that the particulate-plastic particles were not introduced during the handling and cleaning of the fish. Rather, it was suggested that translocation of particulate plastics occurred, although, the exact mechanisms were not studied. Small nanoplastics can passively pass through the cell membrane. Alternatively, larger nanoplastics require active transport, e.g., phagocytosis and persorption (Triebskorn et al., 2019). A study conducted by *Akhbarizadeh* et al. *(*2018*)* estimated the intake rate of particulate plastics to consumers. They found that the highest estimated intake rate was from *P. indicus* (Indian prawns), which resulted in an intake of 555 items/300 g/week for adults and 92 items/50 g/week for children. Akhbarizadeh et al. (2018) also examined the fish species, *A. djedaba*, and found 8.00 ± 1.22 items/10 g fish muscles (average), which represented an intake rate of 240 items/300 g/week (adult) and 40 items/50 g/week (children).

However, although particulate plastics have been shown to be in the edible parts of some fish (Karami et al., 2017), they are often found in the gastrointestinal tract, which is generally removed from many fish species before being consumed by humans (Lusher et al., 2013; Rochman et al., 2015b). Therefore, smaller fish species that are eaten whole, molluscs, and crustaceans pose a greater risk of exposing humans to particulate plastic (Barboza et al., 2018). A study by Van Cauwenberghe and Janssen (2014) examined particulate-plastic content in two different commercially grown bivalves (i.e., a mussel and oyster), which are eaten whole and, therefore, pose a greater chance for humans to become exposed to these particles. They found that *Mytilus edulis* (blue mussel), which originated in the North Sea, contained 0.36 ± 0.07 particles/g of tissue, while *Crassostrea gigas* (Pacific oyster), which originated in the Atlantic Ocean, contained 0.47 ± 0.16 particles/g of tissue. The average particle number per gram of tissue was specified at 0.42, and this was used to estimate that Europeans would consume between 1800 and 11,000 plastic particles/year (size range 5–1000 μm). The values depend upon whether they were low or high consumers of molluscs (Van Cauwenberghe and Janssen, 2014).

7.1.2. Non-seafood

However, seafood is not the only food product to contain particulate plastics, as particles have also been found in foods such as beer, commercial sea salt, and packaged mineral water (Bouwmeester et al., 2015; Kim et al., 2018; Kontrick, 2018; Liebezeit and Liebezeit, 2014; Schymanski et al., 2018). A study conducted by Liebezeit and Liebezeit (2014) examined 24 different brands of German beer, all of which were shown to contain fibres, fragments, or granules of particulate plastics. Concentrations across the three types of plastics were found be as low as two (fibres and granules) and as high as 109 (fragments)/litre of beer.

A study by Kim et al. (2018) examined 39 commercial sea salt brands for particulate plastics. They were purchased from supermarkets across 17 different countries and four continents. According to the study, the Asian region was shown to have the highest content of particulate plastics within its sea salt. The amount of particulate plastic found within sea salt (across the 17 countries) was between 0 and 1674 particles/kg. Rock salt and lake salt were also examined, with their highest values being 148 and 462 particles/kg, respectively.

Finally, a study by Schymanski et al. (2018) examined 38 mineral waters packaged in either plastic or glass bottles, as well as beverage cartons, for particulate-plastic particles. They reported that plastic particles were found within all types of water, and 80% of the particles had sizes ranging between 5 and 20 μm. Concentrations varied depending on the type of container. Returnable and single-use plastic bottles had 118 and 14 particles/L, respectively. Interestingly, glass bottles contained a higher average number of particles than single-use plastic bottles, and the glass bottles had an average of 50 particles/L. However, only the returnable plastic bottles were found to have a statistically significant different number of particles compared to the blank sample (Schymanski et al., 2018). Nevertheless, the results of this study demonstrated the pervasive nature of plastic because the blank samples contained particles despite efforts to avoid contamination.

7.1.3. Airborne particulate plastics

Food products are not the only source of particulate-plastic exposure for humans. Exposure to particulate plastics can also occur from the inhalation of air or inhalable medicines (Browne, 2015; Kontrick, 2018; Prata, 2018). Particulate plastics are both small enough and light enough to be present in the air. They can be directly inhaled and, thus, can end up in the respiratory system of humans (Prata, 2018). A review by Prata (2018) looked at the current information surrounding airborne particulate plastics, and it is believed that exposure is more likely to occur indoors than outdoors. This is potentially due to many indoor sources such as synthetic textiles (e.g., clothing and furniture fibres). Furthermore, particulate plastics are less likely to be removed indoors, as wind and rain are two key features in dispersing or removing the particles from an area (Prata, 2018). Wind erosion in agricultural and natural areas can also distribute particulate plastics. A study by Rezaei et al. (2019) found higher concentrations of particulate plastics in wind eroded sediment that had originated from soils containing increased concentrations of particulate plastics. The results indicated that particulate plastics can be transported through the air. Furthermore, this study also demonstrated that not only is the inhalation of atmospheric plastics a potential source for humans, but those that work closely with the land (e.g., farmers) are more at risk. Finally, exposure to particulate plastics in humans could cause a range of health concerns, including upper respiratory issues and autoimmunity diseases. However, those most at risk include children and industry workers exposed to high levels of airborne particulate plastics (Prata, 2018).

7.2. Human exposure to particulate plastics and trace elements

It has been recently reported that particulate plastics have been found within human stools (Harvey and Watts, 2018). The reported study, which was conducted by the Environment Agency Austria, found 20 particulate plastics/10 g of stool (with their size ranging between 50 and 500 mm). The most common plastics were polypropylene and polyethylene, although nine different plastics were found. These results highlight that humans are exposed to particulate plastics and that the plastics can enter the human gut. Fig. 7 shows the potential fate of particulate plastics and associated trace elements once they have entered humans. Furthermore, the fact that particulate plastics are in the human gut emphasises the need to better understand the consequences regarding the exposure of humans to particulate plastics, such as the effect it has on the immune system and genetics (Brandts et al., 2018).

As mentioned previously, trace elements can adsorb onto the surface of particulate plastics. Thus, if particulate plastics are ingested by humans (particularly through the consumption of seafood), there is the potential for trace-element exposure. Unfortunately, research surrounding human exposure to trace elements through particulate plastics is scarce. It is important to note that trace elements are already found in

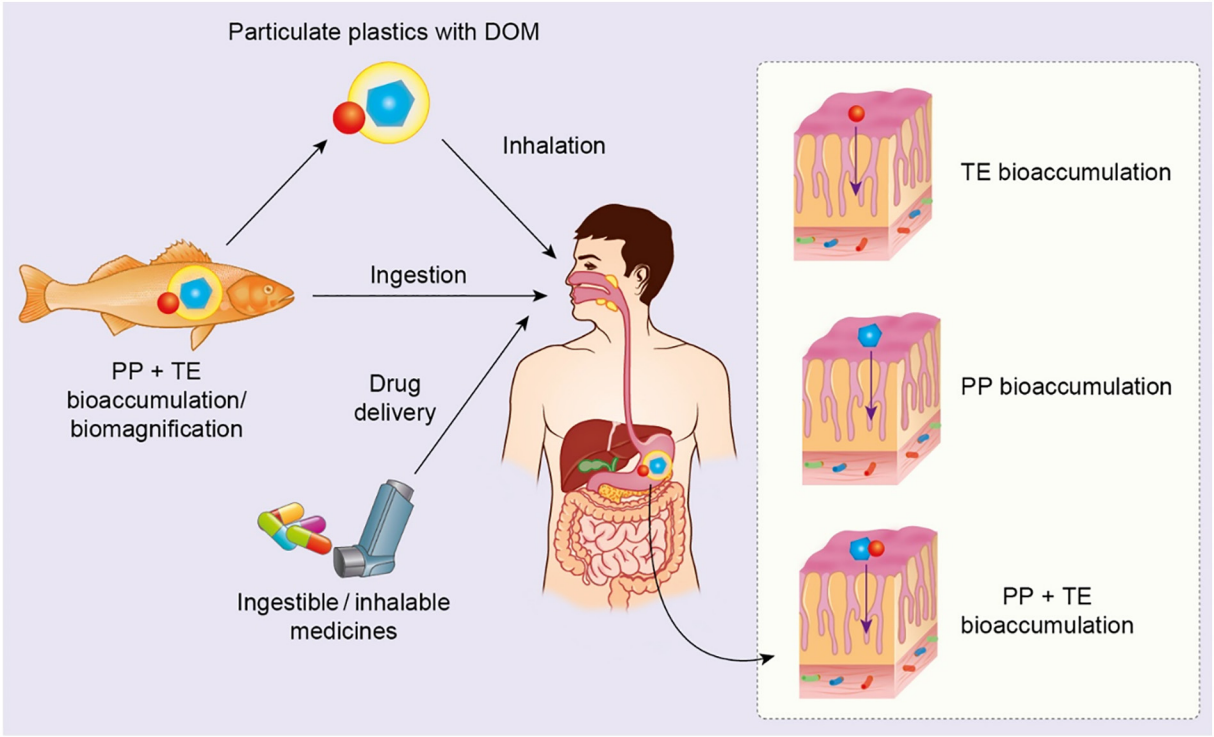

Fig. 7. Conceptual diagram illustrating fate of trace elements derived from particulate plastics in a human. Particulate plastics can be ingested or inhaled by humans. Once in the gastrointestinal and the lower respiratory tracts, the trace metal sorbed particulate plastics may release the metal which will pass into the tissues leading to trace element (TE) bioaccumulation; or the plastics itself will pass into the tissues leading to particulate plastics (PP) bioaccumulation; or both the metal and plastics will pass into the tissues leading to particulate plastic and trace element (PP + TE) bioaccumulation (DOM = dissolved organic matter).

seafood. Therefore, human exposure to high or toxic levels of trace elements may occur without the addition of particulate plastics. A study by Anandkumar et al. (2018) found nine different trace elements (i.e., Cr, Mn, Co, Ni, Cu, Zn, Rb, Cd, and Pb) present within fish. However, when compared to both the Malaysian and International seafood guidelines, it was found that the concentrations were within limits of the guidelines and, therefore, safe for human consumption. Nevertheless, research should examine whether particulate plastics enhance the bioavailability and toxicity of trace elements within humans.

8. Conclusions and recommendations for future research

This review aimed to provide an overview of the origin, characteristics, sources, and fate of particulate plastics in the environment. There are two major sources of particulate plastics in terrestrial and aquatic environments, i.e., primary or secondary particulate plastics. The primary source includes synthetic particulate plastics, and the secondary source includes plastics derived from the breakdown and weathering of primary plastics.

Furthermore, this review examined the types of contaminants adsorbed to the surface of particulate plastics, and the associated uptake of particulate plastics and contaminants by terrestrial and aquatic organisms. Toxic trace elements were the main contaminant focus. Hence, the mechanisms involved to form trace element-particulate plastics complexes, including their interactions and transport in the environment, were examined. Additionally, significant attention was also given to understanding the processes involved in the uptake of particulate plastics with associated trace elements into the tissues of terrestrial and aquatic organisms. Trace elements are not readily adsorbed onto pristine particulate plastics because they are hydrophobic. However, when particulate plastics reach terrestrial and aquatic environments, they readily adsorb DOM, which serves as a vector for the subsequent adsorption and mobility of trace elements. Trace elements become toxic when soil sorbs, and aquatic organisms ingest, the particulate plastics.

Finally, this review examined human's exposure to particulate plastics and associated contaminants. Once particulate plastics and associated trace elements enter marine organisms, they can then make their way up the food chain where humans eventually ingest them. Human exposure can occur not only through the consumption of seafood but also through consuming water, beer, or salt contaminated with particulate plastics. Once in the gut, particulate plastic may have the potential to affect the digestive and immune systems of humans. However, the effects surrounding the exposure of humans to trace-element-sorbed particulate plastics are largely unknown.

8.1. Recommendations and future research

- Forensic evaluation (i.e., number and comparison of the size, shape, and type of polymers) of trace-element-sorbed particulate plastics is needed to assess the likelihood of particulate plastics entering terrestrial and aquatic organisms, as well as humans.
- Long-term stability of trace elements adsorbed onto particulate plastics needs to be examined using advanced spectroscopic techniques.
- Further research should be conducted on the bioavailability and bioaccumulation of particulate plastics and associated contaminants (e.g., trace elements) within organisms. The research should also include the use of radiolabelled particulate plastics to better understand the fate of plastics upon entering an organism.
- Microbial interactions with particulate plastics in relation to the transformation of trace elements (e.g., reduction of less mobile arsenate to more mobile arsenite) and their subsequent mobility need to be examined using molecular fingerprinting techniques.
- The effect of weathering of particulate plastics, as impacted by thermal and chemical decomposition, needs to be examined in relation to the release and bioavailability of trace elements adsorbed onto these particulate plastics.
- Direct toxicity, resulting from the release of chemicals such as

bisphenol from particulate plastics, and indirect toxicity, resulting from trace elements adsorbed onto particulate plastics, need to be examined for terrestrial organisms, aquatic biota, and humans.

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