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REVIEW ARTICLE

# Occurrence, sources, human health impacts and mitigation of microplastic pollution

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## Abstract
The presence and accumulation of plastic and microplastic (MP) debris in the natural environment is of increasing concern and has become the focus of attention for many researchers. Plastic debris is a prolific, long-lived pollutant that is highly resistant to environmental degradation, readily adheres hydrophobic persistent organic pollutants and is linked to morbidity and mortality in numerous aquatic organisms. The prevalence of MPs within the natural environment is a symptom of continuous and rapid growth in synthetic plastic production and mismanagement of plastic waste. Many terrestrial and marine-based processes, including domestic and industrial drainage, maritime activities agricultural runoff and wastewater treatment plants (WWTPs) effluent, contribute to MP pollution in aquatic environments. MPs have been identified in food consumed by human and in air samples, and exposure to MPs via ingestion or inhalation could lead to adverse human health effects. Regulations in many countries have already been established or will soon be implemented to reduce MPs in aquatic environments. This review focuses on the occurrence, sources, and transport of MPs in terrestrial and aquatic environments to highlight potential human health effects, and applicable regulations to mitigate impacts of MPs. This study also highlights the importance of personality traits and cognitive ability in reducing the entry of MPs into the environment.

**Keywords** Microplastics · Microfibres · Marine · Freshwater and terrestrial microplastics · Controlling sources of microplastics

## Introduction

Since the development of the first synthetic resin at the beginning of the twentieth century, plastics have become indispensable in society (Cole et al. 2011; Sivan 2011). Plastics or synthetic organic polymers are derived from natural, organic materials such as coal, natural gas, and crude oil by polymerisation or polycondensation processes (Phuong et al. 2016). There has been a significant increase in global production of plastics in the last 50 years, with production rising from 1.7 million tonnes in the 1950s to 335 million tonnes in 2016 (PlasticsEurope 2017). It is estimated that 8 million metric tons (Mt) of generated plastic waste on land entered the marine environment in 2010 alone (Geyer et al. 2017; Jambeck et al. 2015). Trends of global plastic production, consumer-use patterns, inappropriate disposal of plastic waste and demographics suggest an increase of plastic use in the future. Plastic demand is growing exponentially, and trends of production are expected to quadruple by 2050 (Suaria et al. 2016). Although industrial benefits of plastic are widespread, this valuable commodity has become a considerable environmental concern for governmental and private sectors, scientists, and general public (Seltenrich 2015). Key concerns include (a) plastic is a non-renewable resource; (b) persistent organic pollutants (POPs) are sorbed very efficiently to plastics; (c) durability of plastic makes it highly resistant to degradation; (d) plastic debris is vulnerable to fragmentation; (e) plastic debris can cause injury and death of marine birds, mammals, fish and reptiles towing to plastic entanglement and ingestion (Lopez Lozano and Mouat 2009; Trevail et al. 2015; Van Franeker et al. 2011; Wright et al. 2013); and (f) plastic debris can damage maritime equipment (Phuong et al. 2016). Furthermore, the presence of macroplastic debris

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(larger than 5 mm; Driedger et al. 2015) can create aesthetic issues, damage to the seabed because of sinking plastic debris, and economic pressures on the shipping sector (fouled motors, lost output and repair costs), fishing (lost or discarded nets), and tourism by loss of revenues (Cole et al. 2011). For example, in 2011 vast amounts of marine debris on the beaches of Geoje Island, South Korea affected the island's tourism industry, with lost revenue estimated at US $29–37 million (Jang et al. 2014).

The term microplastics (MPs), first coined in the scientific literature by Thompson et al. (2004), describe very small plastic particulates and fibres. The size definitions for MPs are non-uniform, and include: > 1.6 μm (Ng and Obbard 2006), < 1 mm (Browne et al. 2007, 2010; Claessens et al. 2011), < 2 mm (Ryan et al. 2009), 2–6 mm (Derraik 2002), < 5 mm (Barnes et al. 2009; Betts 2008), < 10 mm (Graham and Thompson 2009). The National Oceanic and Atmospheric Administration (NOAA) now defines the term MPs as tiny ubiquitous plastic particles < 5 mm in diameter (Arthur et al. 2009). Origins of MPs can be distinguished as primary and secondary sources (Cole et al. 2011). Primary MPs (microbeads) are defined as plastics produced at a micro-sized scale, including those generated for use in industrial and domestic products such as hand and facial cleansers to strengthen cleansing or exfoliating functions (Zitko and Hanlon 1991), cosmetics, medicine as drug vectors (Patel et al. 2009) and scrubbers in air-blasting (Gregory 1996). Lei et al. (2017) reported the presence of different shapes and contents of MPs, mainly polyethylene (PE) in various facial cleansers and shower gels in Beijing, China. Secondary MPs result from the breakdown of macroplastics items both at sea and on land (Ryan et al. 2009; Thompson et al. 2004); plastics undergo different degradation processes in the environment, including mechanical (erosion, wave action, abrasion), chemical (photooxidation, temperature, corrosion) and biodegradation activities, which lead to their fragmentation into MP (Andrady 2011; Zettler et al. 2013). It is believed that the fragmentation of plastic debris in coastal environment is much faster than in water because plastic degrades mainly via solar UV-radiation-induced oxidation and the rate of degradation can be accelerated by the high temperature and UV radiation on the coast surface in comparison to the sea surface (Andrady 2015). Also, chemical and mechanical breakdown of plastic debris is increased during saltation in coastal environments (Corcoran et al. 2009).

MPs are generated mainly from land-based sources (~ 80%), and also from sea-based sources (~ 20%) (Barboza et al. 2019), and are able to move great distances throughout the world because of their properties such as lightweight, durability, buoyancy, shape and colour (Fig. 1). MPs are also widespread in the terrestrial environments as a consequence of daily human activities. Terrestrial ecosystems are considered as major sources and transport pathways of MPs into the marine environment (Horton et al. 2017b). MPs are dispersed into the ocean all over the world. Often detected in beaches (Herrera et al. 2017; Imhof et al. 2018; Naji et al. 2017), seabed sediments (Karlsson et al. 2017; Van Cauwenberghe et al. 2013), wastewater effluents (Magnusson and Norén 2014; Murphy et al. 2016; Ziajahromi et al. 2016), surface waters (Eriksen et al. 2013; La Daana et al. 2017), freshwater systems (Horton et al. 2017b) and even sea ice in the Arctic (Lusher et al. 2015; Obbard et al. 2014; Waller et al. 2017) and the Antarctic (Waller et al. 2017) transported by ocean and wind. MPs also have been observed in the atmosphere, as well as in indoor and outdoor environments (Dris et al. 2017; Gasperi et al. 2018). Bergmann et al. (2017) found high concentrations of MPs (including fibres) in Arctic sea ice, potentially originating from the atmosphere.

A potential environmental risk associated with MPs is their bioavailability throughout the food-web (Cole et al. 2011; Kolandhasamy et al. 2018). Due to the large presence of MPs in aquatic and terrestrial environments, they would be present in food products that sold for human consumption. Identifying original sources and distribution of MPs in terrestrial and aquatic environments will help to identify potential mitigation options to decrease transport of MPs into the environment. This review focuses on the occurrence, sources, and transport of MPs in terrestrial and aquatic environments to highlight potential human health effects, and applicable regulations to mitigate impacts of MP pollution.

## Occurrence and pollution of MPs in terrestrial and aquatic ecosystems

### MPs in terrestrial ecosystems

A wide range of MPs are found in terrestrial ecosystems owing to a plethora of anthropogenic activities, yet only a limited number of studies have explored the abundance of MPs on land; typically, current studies consider terrestrial ecosystems only as sources and distribution pathways of MPs to aquatic or marine environments (Horton et al. 2017b; Jambeck et al. 2015; Lechner et al. 2014; Rillig 2012). Lechner et al. (2014) showed the release of substantial amounts of industrial MPs from a production plant into the River Danube, which is legally discharged into the river (Lechner et al. 2014). Soils are essential components of terrestrial ecosystems that can experience heavy anthropogenic pollution pressure (Walker et al. 2003). Fragmentation of plastics at coastal areas or in surface water occurs as a consequence of direct exposure to UV-radiation from sunlight and physical abrasion processes; however, both drivers are mostly missing in soil, therefore fragmentation of plastic in soil could be very slow. Some studies reported minimal degradation of

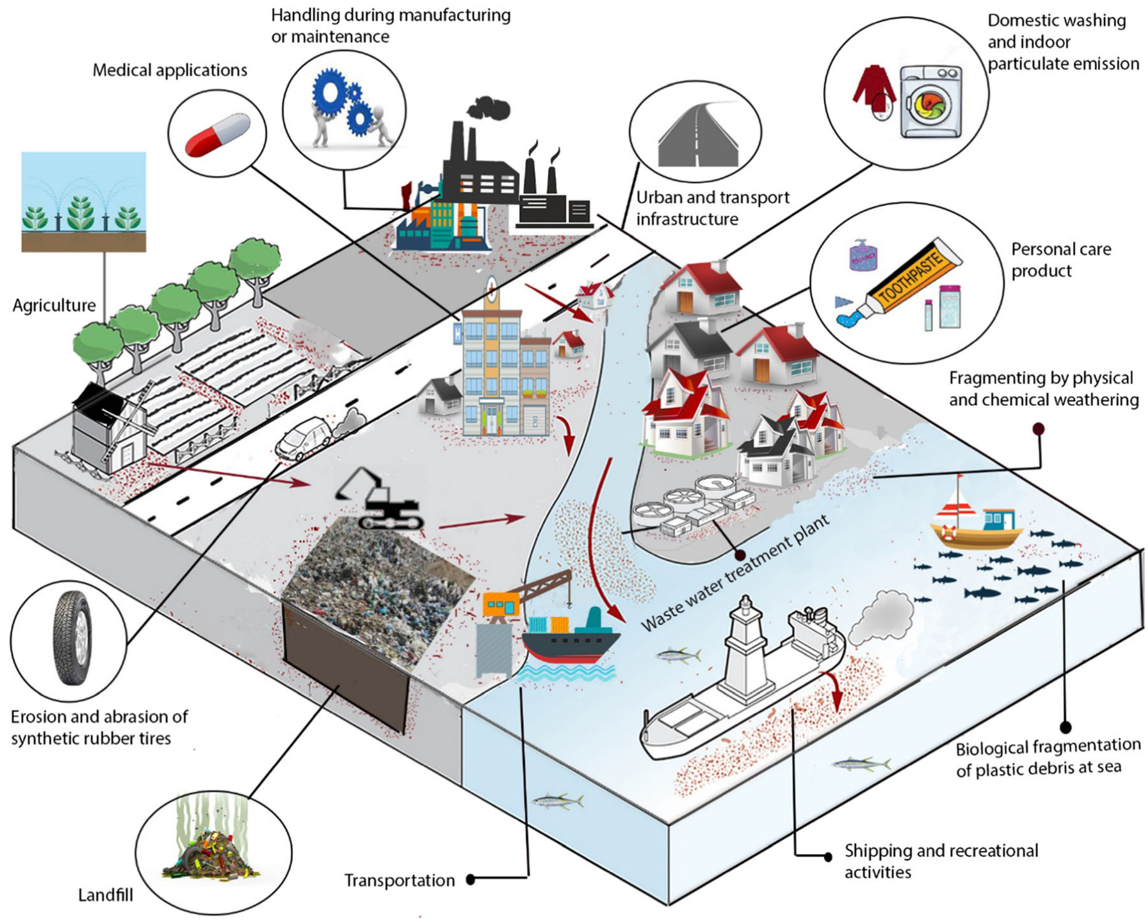


**Fig. 1** How microplastics contaminate the Earth's ecosystems

synthetic polymers in soil. Albertsson (1980) detected only 0.1 to 0.4% weight loss of PE in soil after 800 days. Another study by Arkatkar et al. (2009) reported only 0.4% weight loss of polypropylene (PP) in soil after one-year incubation, while no degradation of polyvinyl chloride (PVC) was found in soil after 10 to 35 years (Ali et al. 2014; Otake et al. 1995; Santana et al. 2012). Soil texture is also a key factor that affects the rate of polymer degradation. César et al. (2009) found that degradation was enhanced in clay soils when compared with sandy soils. De Souza Machado et al. (2018) showed the potential of MPs to disturb vital relationships between soil and water due to the effect of MPs on bulk density, and water-holding capacity, as well as its consequences for soil structure and microbial activity. Studies on the existence of MPs in soil are scarce because of the quantification of MPs in soil is a challenge and analytical techniques required for this type of analysis is still relatively new. Thus, the identification of MPs in soils needs to be developed before assessing the MP content of different soils. There are three promising methods for characterising MPs in soils published recently; however, they have still limitations in their methods. For instance, Fuller and Gautam (2016) developed a method based on pressurised fluid extraction (PFE) for measuring MPs in soil samples, but the limitation is inability of the method to measure size fractions of MPs. PFE is a standard extraction technique that uses solvents at subcritical temperature and pressure conditions, and is commonly applied in environmental laboratories for the extraction of organic pollutants from soils, sediments and wastes. Using this method, they found that topsoils near industrial areas around Sydney (Australia) contain 0.03% to 6.7% of MPs. Shan et al. (2018) showed that hyperspectral imaging technology is a novel way to measure and visualize the MPs with particle size from 0.5 to 5 mm on soil surface directly. Another study conducted by Zhang et al. (2018) introduced a simple and cost-saving method (floatation and heating method) to extract, distinguish and quantify light density MPs (e.g. PE and PP) in soil. In a floatation method, distilled water was used to extract the MPs from soil samples with recovery rates of approximately 90%. Then, samples were exposed to heat and MPs in the soil samples melted and transformed into circular transparent particles. This method is limited to measure light density plastics. On the contrary, Scheurer and Bigalke (2018) developed a promising alternative method to analyse size distribution, composition and concentrations of most commonly produced MPs in soils by FT-IR microscopy.

Many factors affect the amount of deposition, retention and transport of MPs including human behaviours (e.g. general littering, dumping of plastic waste, inappropriate waste management), characteristics of particles (e.g. density, shape and size), weather conditions (e.g. wind, rainfall) and environmental topography (Zylstra 2013). A considerable direct input of primary MPs to terrestrial ecosystems and soils have been identified as being through areal deposition, sedimented MPs from personal care products or household items, landfills or other surface deposits and sludge application to agricultural land (Horton et al. 2017b; Rillig 2012; Steinmetz et al. 2016). Typically, in industrialised countries, landfills are surrounded by fences, and waste is usually covered with soil or a synthetic material, which helps mitigate MP run-off from such sites. However, in developing countries, these best management practices are often not followed (Duis and Coors 2016). There is a significant plastic waste accumulation in soils in many tropical and subtropical countries, also agricultural/municipal plastic wastes are buried or disposed in open fields, gardens, or landfills. In Europe, around 1000 to more than 4000 MPs particles/kg of dry mass sludge were found in agricultural and landfill sites (Huerta Lwanga et al. 2016). In another study by Fuller and Gautam (2016), the presence of MPs was observed in various soils from an industrial area in Australia. It is estimated that up to 700,000 tons of MPs may enter farmland annually through biosolids application in Europe and North America (Nizzetto et al. 2016).

MPs from personal care products (e.g. shower gels, hand cleaners, shampoos, facial cleaners and toothpaste) can reach the environment through wastewater treatment plants (Duis and Coors 2016). Industrial abrasives such as polystyrene (PS), polyester (PES), melamine can end up in the environment, if not used properly in closed systems. Other sources of primary MPs including plastic resin pellets/flakes and plastic powder/fluff can reach the environment after improper handling (Duis and Coors 2016). Similarly, residues from plastic processing and plastic recycling plants can end up in the environment (Andrady 2011; Duis and Coors 2016). The presence of high concentrations of raw materials used for the manufacture of plastic supplies was often observed on beaches nearby the plastic-producing/processing sites (Duis and Coors 2016).

Some organisms including earthworms could contribute in formation of secondary MPs by grinding up brittle plastic debris in their gizzard (Rillig 2012). Collembola or mites may also contribute to breakdown of plastics by scraping or chewing off pieces of plastic. Similarly, digging mammals could contribute to abrasion and incorporation of plastics into the soil (Rillig 2012). MPs could sorb chemical contaminants from the soil and concentrate them in the soil. Additionally, MPs could alter physical properties of the soil. Rillig et al. (2017) showed the downward movement of PS MPs from the soil surface to the soil profile via earthworms. According to the study by Rillig et al. (2017), there are several possible implications to carry MPs down the soil profile via available organisms: (a) Decomposition of organic material in the deep part of the soil is generally much slower due to few populations of microbes. It would mean that MPs may have longer durability in greater depths in the soil profile; (b) MPs after passing through the soil profile could potentially reach groundwater and lead to adverse effects on in other aquatic environments; (c) MPs may convert to nano-sized material due to further fragmentation in the soil, which poses different environmental risks.

However, some sewage treatment plants are effective at removing the majority of MPs (up to 99.9% removal of MP debris) from wastewater, but a significant number of MP debris will be retained within the sludge (Gies et al. 2018; Mintenig et al. 2017; Prata 2018a). In Portugal, more than 87% of the total amount of sewage sludge is applied as agricultural fertiliser (using biosolids) either directly or after composting (Alvarenga et al. 2016). Additionally, in the European Union (EU), between 4 and 5 million tonnes of sewage sludge (dry weight) are used to agricultural land every year. Zubris and Richards (2005) showed soils that received sludge products had significantly higher concentrations of synthetic fibre compared to field site soil which had not received sludge products. In some land sites, synthetic fibres were found after 5 years sludge application.

## Sources and transport of MPs into aquatic environments

In the early 1970s, the first reports of plastic litter in the marine environment drew the minimal focus of the scientific society (Carpenter and Smith 1972; Colton et al. 1974). In recent decades, direct or indirect indiscriminate disposal of waste items to the marine environment and associated ecological impacts of debris has increased sustained research interest (Walker et al. 1997). Early studies focused on the entanglement of marine fauna such as whales (France 2016), fur seals (Boren et al. 2006), turtles (Mascarenhas et al. 2004), seabirds (O'Hanlon et al. 2017), and cetaceans (Baulch and Perry 2014) in net fragment litter or via 'ghost fishing'. More recently, studies showing ingestion of MPs by marine biota including fish, mussels and shrimps have become extensively documented (Catarino et al. 2018; Devriese et al. 2015; Neves et al. 2015; Rummel et al. 2016). Further, toxicological studies of MPs and chemical co-transport on marine organisms are assessed for these multifaceted stresses (Akhbarizadeh et al. 2018; Batel et al. 2016). MPs are transported and dispersed throughout the oceans, including beaches, in deep-sea and coastal sediments, and on surface waters from the Arctic to the Antarctic from remote locations (Barnes et al. 2010; International Maritime Organization 2015). Recently, a predominant abundance of PE and PP MPs were found in subsurface waters in the coastal area of the Ross Sea (Antarctica) (Cincinelli et al. 2017). Plastic particles also detected in sediments collected in Terra Nova Bay (Ross Sea, Antarctica), which fibres were the most frequent type of plastics debris found (Munari et al. 2017). Waller et al. (2017) reviewed several sources of MPs within the Southern Ocean: (a) MPs are discharged from wastewater in scientific research stations (52% of research stations had no wastewater treatment systems); (b) fishing and tourist ships; (c) MPs may release into the Southern Ocean from personal care products and laundering synthetic fabrics because of human presence in the region, scientific research stations, and vessels; (d) MPs from degradation of floating debris pollution originating in the Southern Ocean due to high UV levels and (e) Plastics originating outside the Southern Ocean due to major current systems of the Southern Ocean. Another study by Bergmann et al. (2017) showed high quantities of MPs including chlorinated PE (38%), polyamide (PA) (22%) and PP (16%) in Arctic deep-sea sediments from the HAUSGARTEN observatory. In 2016, a field survey was conducted to collect MPs with sizes < 5 mm in the Southern Ocean. Of the 44 fragments found in this study, 29 were made of PE, PP and PE combined with unidentified polymers. However, 14 of the remaining 15 MPs were made of PS, and 1 was made of PVC (Isobe et al. 2017). High level of MPs was also observed in Arctic seabird, the northern fulmar (*Fulmarus glacialis*), which highlight the risk to seabirds and other sea ice animals from plastic pollution (Trevail et al. 2015). Similarly, a recent study by Avery-Gomm et al. (2018) also showed plastic ingested by *F. glacialis* (with an average of 0.151 g/bird) from the southeastern Canadian waters of the Labrador Sea. Also, Provencher et al. (2018) found MPs in 47% of the faecal precursor from *F. glacialis*, which suggest that seabirds are acting as vectors of MPs in the marine environment through guano deposition near their colonies.

It is generally considered that 75–90% of marine plastic originates from land-based and the rest (about 10–25%) originates from ocean-based sources (Andrady 2011; Duis and Coors 2016; Walker et al. 2006). The MPs debris present in cosmetics such as scrubbers in cleaning products, exfoliating creams, air blasting media, and fibres from laundry can enter the aquatic environment through industrial and domestic drainage systems. Wastewater treatment plants (WWTPs) are one of the dominant point sources of MPs to the marine environment (Magnusson and Norén 2014; Mintenig et al. 2017; Murphy et al. 2016). High concentrations of MPs release every day at WWTPs (Prata 2018a). Recent cases reported by Cesa et al. (2017) also showed synthetic fibres from textile produce MPs sheds which originated from domestic washing and WWTPs. Although, WWTPs have an up to 95% removal of MPs (Prata 2018a; Talvitie and Heinonen 2014; Talvitie et al. 2017) and tertiary treatment can remove a 90% of debris in a size larger than 10 μm (Wardrop et al. 2016), however, there is a significant amount of MPs being discharged into aquatic environment through WWTPs. Gies et al. (2018) estimated that 1.76 (0.3) trillion MPs were discharged from a WWTP annually in Vancouver, Canada, of which 1.28 (0.54) trillion MPs settled into primary sludge, 0.36 (0.22) into secondary sludge, and 0.03 (0.01) trillion MPs were released into the marine receiving environment. Thus, this corresponds to a retention of MPs of up to 99% in this WWTP. A recent study by Mason et al. (2016) studied the effluent samples of 17 WWTPs across the United States (US) and predicted that the average discharge of MPs particles was 13 billion pieces/day. In Finland, an annual estimate of 154,000 to 411,000 kg of PES and cotton microfibres is released in washing machines (Sillanpää and Sainio 2017). Browne et al. (2011) compared MPs found in shoreline sediment samples with MPs detected from samples of marine wastewater effluent disposal site and showed mostly PES and acrylic fibres in both sample types. Further, a study by Talvitie et al. (2015) suggested that wastewater effluents act as a transport pathway for MPs to be released in the environment as they found similar types of MPs (mostly fibres and synthetic particles) in both effluent from a WWTP in Finland and seawater from the Gulf of Finland.

MPs get into the aqueous environment through storm sewers runoff, wind advection and currents (Murphy et al. 2016; Zalasiewicz et al. 2016). Additionally, storm drains from roads may also transport plastic debris such as fragments of road-marking paints (Horton et al. 2017a) and tyre wear particles into natural water (Galgani et al. 2015). Another direct source of MPs to terrestrial and marine environments is fragmentation of plastics that are used in agricultural lands. For example, plastic mulches are used to increases yields, fruit quality, water-use efficiency, control temperature, and moisture in agricultural and horticultural applications; however, MPs are the potential pollution by plastic mulches (Steinmetz et al. 2016). Additional products applied in agriculture are silage and fumigation films, anti-bird, fertiliser sacks, and containers, all of which have the potential for dispersion of MPs within the environment (Muise et al. 2016; Scarascia-Mugnozza et al. 2012). Large amounts of plastic debris can also enter the marine environment during natural disasters such as hurricanes, tsunamis and strong sea (Desforges et al. 2014).

Anthropogenic activities along the coast enter significant amount of MPs to the marine environment such as harbours, recreational, shipping and fishing activities (Driedger et al. 2015). For example, material lost or discarded from fishing ship, aquaculture facilities, and merchant ships are the sources of marine litter. Many ship paints contain synthetic polymers such as polyacrylate, PS, alkyds, and epoxy resins, which is in direct contacts with water (Sundt et al. 2014). MPs were also found in seven intertidal mangroves habitats of Singapore due to the degradation of marine plastic debris (Mohamed Nor and Obbard 2014).

To date, ingestion has been widely accepted as the common pathway for a wide range of aquatic organisms to uptake MPs (Ferreira et al. 2016; Rochman et al. 2013). Bivalves (Mathalon and Hill 2014; Li et al. 2015), zooplankton (Cole et al. 2013), fishes (Rummel et al. 2016), shrimps (Devriese et al. 2015), oysters (Green 2016), sea cucumbers (Graham and Thompson 2009), polychaete worms (Wright et al. 2013) and whales (Fossi et al. 2012) have been reported to ingest MPs. Ingestions of MPs by marine animals lead to adverse health effects including decreased food consumption, false satiation, decreased growth rate, reproductive complications, behavior, oxidative stress, decreased immune response, weight loss, pathological stress and blocked enzyme production could potentially threat marine populations and living resources (Fossi et al. 2016; GESAMP 2016; Lusher et al. 2017; Rochman et al. 2015; Sutton et al. 2016).

Despite the large and growing literature regarding the abundance, sources, and impacts of MPs in the marine environment (Galgani et al. 2015; Sundt et al. 2014; Waller et al. 2017), little information exists on MPs in freshwater aquatic ecosystems. Freshwater and terrestrial ecosystems are recognised as a main source and transport pathways of plastics to the marine environments (Su et al. 2018). A handful of recent studies have examined MPs in lakes (Driedger et al. 2015; Eriksen et al. 2013; Free et al. 2014; Imhof et al. 2018), lakeshore (Imhof et al. 2013; Zhang et al. 2016) sediments, pelagic MPs in rivers (Dubaish and Liebezeit 2013) and the ingestion of MPs by freshwater fauna (Pazos et al. 2017; Sanchez et al. 2014). Eriksen et al. (2013) show that plastic microbeads, which commonly used in facial cleansers and other personal care products, are a significant MP pollutant in the Great Lakes. In the study conducted by Su et al. (2018), the level of industrialisation and hydrological conditions were proposed as important contributors to MPs pollution in Taihu Lake, China. High concentration of MPs across the Rhine River, one of the largest European rivers, reflecting various sources and sinks of MPs to the river such as wastewater treatment plants, effluents of industries, tributaries and weirs, also highlights the important contribution of this river to the MPs mass in the North Sea (Mani et al. 2015).

Because of differences between freshwater and marine ecosystems including the close vicinity of point sources in freshwaters, proximity to urban centers, the smaller size of freshwater ecosystems, human population density proximal to the freshwaters, and differences in the spatial and temporal conditions in the mixing/transport of particles by physical forces, may lead to differences in the type and quantity of MPs present in freshwaters compared to marine systems (Eerkes-Medrano et al. 2015). Eriksen et al. (2013) found a major fraction of MPs in the surface waters of Lakes Superior, Huron and Erie are most likely microbeads that are used in consumer products, such as exfoliating creams, soaps, shampoos, toothpastes, sunscreens, and deodorants. Also, Pelagic MPs in the highly populated Great Lakes of North America were significantly higher than particle counts for the less populated Lakes (Eriksen et al. 2013). The Rochman laboratory is currently studying the fate and transport of MPs in the Great Lakes and is also investigating rates of plastic microfibres shedding from textiles during washing cycles and their accumulation in aquatic environments. Additionally, there are ongoing studies in this laboratory (https://rochmanlab.com) on the transfer of chemical contaminants (e.g. PCBs and PAHs) and metals from plastic debris.

## Sources and dispersion of airborne MPs

Vast data exist in the literature on the presence of MPs in the marine environment (Andrady 2011; Auta et al. 2017; Cesa et al. 2017; Cole et al. 2011; Wang et al. 2016; Wright et al. 2013), whereas there is a gap of knowledge on MPs in the air (Dris et al. 2017; Gasperi et al. 2018). The most important sources of airborne MPs are determined to originate from synthetic textiles, erosion and abrasion of synthetic rubber tires, city and household dust (Prata 2018b). A single item of clothing may be responsible for the discharge of around 1900 fibres per wash (Browne et al. 2011). Other sources of airborne MPs may include construction materials, waste incineration, landfilling (Dris et al. 2016), industrial outflows, roadway particles, resuspension of particles (Dris et al. 2015), synthetic particles such as PS peat, which applied in horticultural soils, use of sewage sludge in agriculture as a fertiliser and tumble dryer exhaust (Prata 2018b). Dris et al. (2016) investigated the presence of fibrous MPs in the total atmospheric fallout at an urban site and suburban site in the Paris Megacity. Chemical characterisation appeared that 29% of the fibres measured in total atmospheric fallout are synthetic (e.g. made with petrochemicals), or a composition of natural and synthetic materials.

Factors affecting MP behavior and transport in the atmosphere may also be similar to those of fine particulate matter including vertical pollution concentration gradient (higher concentrations close to the land), wind speed (increasing of wind speed lead to a decrease in concentration), wind direction (downwind, upwind, and parallel directions), precipitation, temperature and humidity (Kaur et al. 2007; Zhao et al. 2014). Also, urban topography (e.g. tall buildings, trees and space between buildings) can affect wind modulation and distribution of air pollutants in urban environments (Fernando et al. 2001). Lighter polymers can be transported easily by the wind and further contaminate the terrestrial and marine environments (Horton et al. 2017b).

Indoor air is another important source and the main place of exposure to airborne MPs due to lower removal by dispersal mechanisms (Prata 2018b) and people spend around 70–90% of their time inside (Alzona et al. 1979). The behavior of indoor airborne MPs depends on room partition, ventilation and airflow (Alzona et al. 1979). Catarino et al. (2018) compared the potential exposure of humans to household dust fibres and MPs in caged mussels (*Mytilus edulis*). The result showed the risk of plastic ingestion through consumption of mussel is minimal compared to fibre exposure during a meal through household dust fallout. The MPs from the indoor air could contaminate the outside air, because they are diluted in the atmosphere (Dris et al. 2017), while only 30% of outdoor particulate matter can enter indoors in a closed room (Alzona et al. 1979). Fibres are the most common microplastic type observed in sediments (Claessens et al. 2011), surface seawaters (Lusher et al. 2014), aquatic biota (Rochman et al. 2015) as well as atmospheric fallout (Dris et al. 2016) and in indoor environments (Dris et al. 2017).

## MPs and potential human health impacts

As an emerging area of concern to MPs is that they can also enter the human food chain through ingestion of seafood and terrestrial food products causing potential human health impacts (Rist et al. 2018; Wright and Kelly 2017) (see Table 1). The presence of MPs in the guts and tissues of aquatic species including some commercially important bivalves (Mathalon and Hill 2014; Li et al. 2015; Naji et al. 2018), crustaceans (Bos et al. 2018) and fish (Bessa et al. 2018a, b; Neves et al. 2015) is well documented. Key factors contributing to the bioavailability of MPs are size, density, abundance and color (Wright et al. 2013). The small size of MPs makes them available to the lower trophic organisms, which can capture anything of appropriate size. The density of the MPs will determine bioavailability in the water column; therefore, the type of plastic debris ingested may vary between organisms. An increase in the abundance of MPs in the marine environment will also affect its bioavailability because the chance of organisms to encounter MPs is enhanced (Wright et al. 2013). In a study by Moore et al. (1998), the uptake of latex spheres (1 μm) from the water in rainbow trout (*Oncorhynchus mykiss*) were observed in the surface and subsurface epidermal cells and underlying phagocytes of the skin and gill surface. This highlights the importance of epithelial cells in the adherence and entry of MPs to the fish body. Thus, consumption of the skin or gill tissue could also be a direct route of human exposure to MPs even $\geq 1$ μm. In a study recently performed on the soft tissue of mussels strongly suggested that adherence is a novel way for organisms to accumulate MPs beyond ingestion (Kolandhasamy et al. 2018).

In addition to seafood, MPs have been reported in other food products, such as sea salt. The presence of MPs in sea salt has recently been reported through studies by Iñiguez et al. (2017), Karami et al. (2017a), and Yang et al. (2015). Karami et al. (2017a) investigated concentration of MPs from 17 brands of salt originating from eight different countries and the number of MPs in salt was nil, in the range 0–10 MPs/kg. On the contrary, the amounts of MPs found in different Chinese and Spanish table salt were in the range of 7–680 MPs/kg and 50–280 MPs/kg, respectively (Iñiguez et al. 2017; Yang et al. 2015). Differences between the studies may be related to the errors in the experimental procedures used to extract the MPs. Therefore, further studies are required to improve the reliable method for quantifying MPs in salt. Additionally, both honey and sugar have been found to contain a small number of fibres and fragments (Liebezeit and Liebezeit 2013). An average of 174 (kg/honey) fibres and 9 (kg/honey) fragments, and an average of 217 (kg/sugar) fibres and 32 (kg/sugar) fragments were found. Authors suggested that the sources of synthetic MPs to contamination of honey are airborne. In contrast, a similar study on honey samples did not find a notable contamination of MPs (Mühlschlegel et al. 2017). Previous research has indicated that the methodology used in the studies on honey, sugar and beer (Liebezeit and Liebezeit 2013, 2014) was questioned due to the background contamination and wrong identification of plastic particles (Lachenmeier et al. 2015).

Recently, MPs have been found in tap water (Kosuth et al. 2017) and bottled water (Schymanski et al. 2018). MP contamination was investigated in tap water from six regions on five continents. Plastic particles were found in 83% of analyzed samples (the range was between 0 and 57 particles/L) (Kosuth et al. 2017). However, the results lack a thorough analysis in the confirmation of synthetic origin of the particles. On the contrary, in the study on bottled water, a precise analysis was conducted, and the average MP content was 118 particles/L in returnable and 14 particles/L in single-use plastic bottles that suggested found particles correlated with the materials the bottles were made of (Schymanski et al. 2018). There are thus still many uncertainties in methodology and analysis of MPs which should be improved by further researches.

A recent study by Karami et al. (2018) examined the potential presence of micro- and mesoplastics in canned sardines and sprats. Results showed the MPs were absent in 16 brands from 20 analyzed brands of canned sardines and sprats, and between 1 and 3 plastic particles per brand were found in the rest. The authors suggested that food safety management systems are urged to place test of MPs in their guidelines because of the possible increase in loads of MPs in canned fish.

Table 1 lists examples of studies on the presence of MPs in products consumed by humans which have mostly been conducted since 2016, because several jurisdictions such as the UK, US and Canada have passed legislation to ban MPs (specifically microbeads) at or around that time (Schnurr et al. 2018; Xanthos and Walker 2017). The consumption of MP

**Table 1** Studies that have detected microplastics in products consumed by humans

| Products | Concentration | Plastic polymer | References |
|---|---|---|---|
| Yellowfin bream (*Acanthopagrus australis*) | Mean 0.6 MPs per fish | PET, PP | (Halstead et al. 2018) |
| Sea mullet (*Mugil cephalus*) | Mean 2.5 MPs per fish | | |
| Silverbiddy (*Gerres subfasciatus*) | Mean 0.1 MPs per fish | | |
| Drinking water | 50 (52) particles/L | PET, PP, PE | (Schymanski et al. 2018) |
| Canned sardines | 1 and 3 plastic particles per brand | PP, PET, PVC, PE | (Karami et al. 2018) |
| Commercial fish species | 0–3 MPs per species | PP, PE, PS, PVC | (Baalkhuyur et al. 2018) |
| European sardine *(Sardina pilchardus)* | 0 to 3 items per fish | PET, PA, polyacrylamide | (Compa et al. 2018) |
| European anchovy *(Engraulis encrasicolus*) | | | |
| Mussels (*Mytilus edulis*) | 3.0 (0.9)/g | PET, PUR, Polyether | (Catarino et al. 2018) |
| Oysters (*Saccostrea cucullate*) | 1.4 to 7.0 items per individual | PET, PP, PS, PA, PVC | (Li et al. 2018) |
| Sea bass (*Dicentrarchus labrax*) | 0.30 (0.61) MPs per fish | PES, PP, Polyacrylonitrile, PE, PA, nylon | (Bessa et al. 2018b) |
| Seabream (*Diplodus vulgaris*) | 3.14 (3.25) MPs per fish | | |
| Flounder (*Platichthys flesus*) | 0.18 (0.55) MPs per fish | | |
| Sea salts | 50–280 MPs/kg | PET, PUR, PP, PE, PMMA, PA, PVC | (Iñiguez et al. 2017) |
| Honey | 1760/kg and 8680/kg | PET | (Mühlschlegel et al. 2017) |
| Flounder (*Platichthys flesus*) and European smelt (*Osmerus eperlanus*) | 75% European flounder and 20% smelt | PA, Acrylic, Nylon, PE, and PET | (McGoran et al. 2017) |
| Dried fish | 61 particles | PP, PE, PS, PET, Nylon | (Karami et al. 2017b) |
| Sea salt | 72 particles | PP, PE, PET, Nylon6, PS, Polyacrylonitrile | (Karami et al. 2017a) |
| Demersal (cod, dab, flounder/pelagic fish (herring and mackerel) | 54 Particles $mg^{-1}$ | PE, PA, PP, PS, PET, PES, PUR | (Rummel et al. 2016) |
| Nile perch (*Lates niloticus*) and Nile tilapia (*Oreochromis niloticus*) | 20% of each fish species | PE, PES, PP, PU | (Biginagwa et al. 2016) |
| Japanese anchovy (*Engraulis japonicus*) | mean 2.3 MPs per individual | PE, PP, PS | (Tanaka and Takada 2016) |
| Atlantic cod (*Gadus morhua*) | 18.8% MPs | PES, PP, PVC, PS, Nylon, PE | (Bråte et al. 2016) |
| Table salts | 550–681 particles/kg | PET, PE, cellophane | (Yang et al. 2015) |
| Lake salts | 43–364 particles/kg | | |
| Rock salts | 7–204 particles/kg | | |
| Commercial fish | 19.8% of fish from 26 species | PP, PE, Alkyd resin, Rayon, PES, Nylon and Acrylic | (Neves et al. 2015) |
| Marine fish | 2.6% of fish | PE, PP, PET, SA | (Foekema et al. 2013) |
| Marine fish | 1.90 (0.10) particles/fish | PS, PES, PA, Rayon | (Lusher et al. 2013) |
| Marine fish | 7.9% of the fish | PA | (Dantas et al. 2012) |
| Catfish species | (Possatto et al. 2011) | | |
| (*Cathorops spixii*) | 18% of individual fish | PA | |
| (*Cathorops agassizii*) | 18% of individual fish | | |
| (*Sciades herzbergii*) | 33% of individual fish | | |

*PET* polyethylene-terephthalate, *PP* polypropylene, *PE* polyethylene, *PS* polystyrene, *PES* polyester, *PUR* polyurethane, *PVC* polyvinyl chloride, *PA* *Polyamide*, *PMMA* polymethyl-methacrylate, *SA* styreneacrylate contaminated foods is a potential source of human MPs intake. Recent studies showed the most commonly plastic polymers found in food products are polyethylene-terephthalate (PET), PP, PE, PES, PVC, PS, PA, and nylon (Table 1). Plastic particles can impact on human health, with effects mainly related to toxicity of the chemical that absorbs from the environment or additives that are used in plastic materials. Non-enzymes are reported to be available to degrade the synthetic polymers in any organism (Wang et al. 2016). Based on the UN Globally Harmonised System, more than 50% of plastics are accompanied with hazardous monomers, additives, and chemical by-products (Lithner et al. 2011). PET is commonly used in the production of drink bottles, plastic film, microwavable packaging, pipes, and insulation molding, which is considered as a potential human carcinogen (Li et al. 2016). PS is also commonly used in the production of Packaging foam, disposable cups, plates, tableware, CD, tanks, and building materials (insulation). PS and PVC have been shown to release toxic monomers that lead to cancer and reproductive abnormalities in humans, rodents, and invertebrates (Wang et al. 2016). Additives in PVC could transfer from medical supplies to humans and indicate that additives could accumulate in the blood (Mettang et al. 1996). In a study conducted by Forte et al. (2016), PS nanoparticles affected cell viability, inflammatory gene expression, and cell morphology of human gastric adenocarcinoma epithelial cells.

Uptake of plastics especially MPs by humans through inhalation has the potential to cause adverse health effects by particle toxicity, chemical toxicity and pathogen and parasite vectors (Vethaak and Leslie 2016). The common mechanism of MP uptake and clearance in the lung is thereby several factors, including hydrophobicity, surface charge, surface functionalisation, the associated protein corona, and particle size, which cause lung injury (Rist et al. 2018). The translocation of smaller particles within the gastrointestinal tract is likely more efficient since nanoparticle PS microspheres found in the blood and organs of rat (Jani et al. 1990). Many of the findings from the particulate materials studies support the notion that micro- or nanometer size of plastic particles could adversely affect human health. Two previous studies observed cellulosic and plastic fibres in human lung tissue which are taken from patient with different types of lung cancer (Pauly et al. 1994, 1998). Recently, in a study conducted by Chan et al. (2017), a questionnaire survey of 46 workers who used different types of 3D printers by the most frequent printing materials including polylactic acid (64%), acrylonitrile-butadiene-styrene plastic filaments (27%) and nylon (23%) showed respiratory symptoms in 57% of participants who worked more than 40 h/week. A work-related interstitial lung disease that induces coughing, dyspnea, and reduction of lung capacity was observed in 4% of workers from nylon flock plants in the US and Canada (Boag et al. 1999; Eschenbacher et al. 1999). The health impact of MPs exposure has shown in Fig. 2.

Toxic chemical additives in the plastic which are a palpable concern for human health include phthalates, bisphenol A (BPA), brominated flame retardants (BFR), triclosan, bisphenone and organotins (Galloway 2015). Little information is available on the leaching of additives into biological tissues directly, although Koelmans et al. (2014) showed that additives including nonylphenol (NP) and BPA could leach from plastics ingested by marine organisms. BPA has received considerable press and scientific attention in relation to human health implications (Galloway 2015). At present, BPA is the main chemical used as a monomer for polycarbonate (PC) plastics and in the epoxy resins lining layer of food and beverage cans (Crain et al. 2007). BPA can also be used as an antioxidant or as a plasticizer in other polymers (e.g. PP, PE and PVC) (Rani et al. 2015). Several studies have shown that BPA can migrate out of PC and contaminate foodstuffs and drinks (Calafat et al. 2008; Galloway 2015; Guart et al. 2013), and lead to liver function alternation, changes in insulin resistance, development of offspring in the womb of pregnant women, reproductive system and brain function (Srivastava and Godara 2017). BPA acts as an agonist for the estrogen receptors, inhibits thyroid hormone-mediated transcription by acting as an antagonist (Moriyama et al. 2002), altered pancreatic beta cell function (Ropero et al. 2008). Adverse human health effects including the onset of obesity and cardiovascular disease (Cipelli et al. 2014; Lang et al. 2008; Melzer et al. 2012) and with numerous reproductive and developmental outcomes (Galloway 2015) were observed in exposure to BPA at levels around 0.2–20 ng/mL (In the general population with measured urinary BPA). Phthalate esters are applied as plasticizers to enhance the flexibility and durability of various materials and also used in manufacturing of PVC polymers and plastisol (Gómez and Gallart-Ayala 2018). Phthalate esters are potentially harmful when exposed to humans, which may possible cause abnormal sexual development and birth defects (Cheng et al. 2013). Additionally, USEPA classified butyl benzyl phthalate (BBP) and di-2-ethylhexyl phthalate (DEHP) as probable and possible human carcinogens (USEPA 2007).

## MP regulations

Despite the global attention to plastic pollution and its environmental effects in recent years, there are currently no regulations established yet to manage impacts of secondary MPs. In contrast, many regions have established or implemented regulations to ban production and use of primary MPs, including microbeads, which could reduce MPs entering the aquatic environment (Beat the Microbead 2016; CEPA (Canadian Environmental Protection Act) 2016; Legislative Assembly

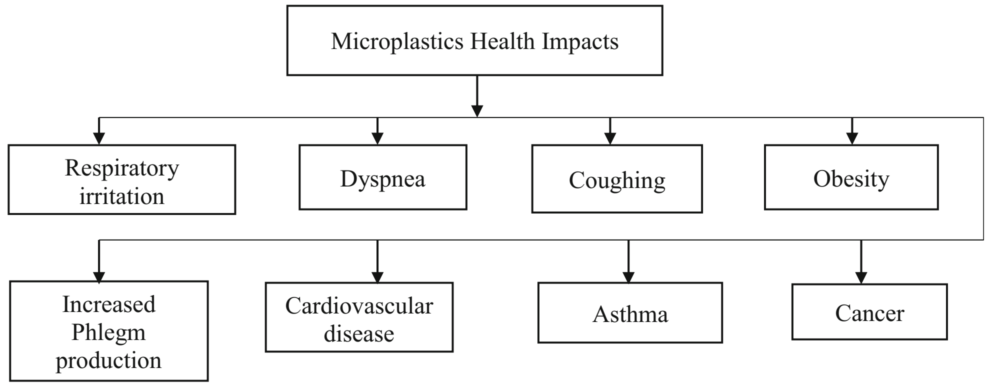


**Fig. 2** Summary of potential human health impacts of microplastics exposure

of Ontario 2015; Pettipas et al. 2016; United States Congress 2015; United Kingdom Department for Environment Food and Rural Affairs 2016), as well as bans or limits on use of single-use macroplastics (e.g. drinks bottles, carrier bags). The first country that declared its intent to produce microbead free cosmetic products was the Netherlands, with a target of 2016. The province of Ontario, Canada, passed legislation to ban the microbeads production in 2015 (Legislative Assembly of Ontario 2015), and since the Canadian federal government classified plastic microbeads as a toxin under the Canadian Environmental Protection Act (CEPA 2016). In December 2015, national legislation was passed by the US Congress to control microbead plastics in the US (United States Congress 2015). Yet, secondary MPs are the major contributors to environmental MP pollution, and therefore, we advocate that new legislation and management policies need to be established to control widespread MPs in the environment. In 1973, the International Convention for the Prevention of Pollution from Ships (MARPOL 73/78) was signed; however, a complete ban on the disposal of plastics at sea was not enacted until 1988. Despite the presence of 134 countries to eliminate the plastics disposal at sea, studies have revealed the increasing problem of marine debris since MARPOL 73/78 was signed. Reasons vary depending on jurisdiction, but are most often related to mismanaged waste on land (Xanthos and Walker 2017). The United Nations Environmental Programme (UNEP) has called for immediate action to rid the oceans of MPs as they have noted that MPs are consumed by a wide number of marine organisms, and this leads to both physical and chemical harm (Jiang 2017; UNEP 2014). Therefore, UNEP developed a program by 40 million people from 120 countries, which set up educational procedures to make awareness and encourage the decrease of plastic use, recycling, and evaluate disposal facilities. The United Nations Environment Program/Mediterranean Action Plan (UNEP-MAP), the Oslo/Paris convention (for the protection of the marine environment of the North-East Atlantic (OSPAR), and the Baltic Marine Environment Protection Commission-Helsinki Commission (HELCOM) also have expanded guidelines to evaluate marine litter such as MPs (Jiang 2017). Non-Governmental Organizations (NGOs) have also presented plans to increase awareness and aid to quantify the level of MPs pollution and their impacts at the national and international scales. For example, the 5 Gyres Institute and the Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection engage in awareness campaigns (Xanthos and Walker 2017). Also, on Earth Day 2018, an End Plastic Pollution campaign was launched in response to the substantial increase of plastic waste on our planet. This campaign used the high profile of Earth Day to increase awareness about the issue of plastic pollution and to highlight the issue on the global agenda to demand effective action from governments and individuals for reducing and managing plastic, specifically single-use plastic (Earthday Network 2018). Similarly, there have also been recent studies aimed at highlighting various global strategies to reduce single-use plastic use at national, regional and municipal levels of government, corporations and at the individual level (Schnurr et al. 2018; UNEP 2018). Single-use plastic bag interventions (e.g. bans or levies) have been reported to reduce plastic bag use between 33 and 96% which help mitigate single-use plastic marine pollution (Schnurr et al. 2018).

The US and France are the first and second countries respectively to ban MPs from rinse-off cosmetic products with Microbead-Free Waters Act (Kentin 2018; McDevitt et al. 2017). Similarly, this legislation is proposed in Taiwan, South Korea and Sweden, which is waiting for approval and adoption (Kentin 2018). According to the REACH Regulation in the European Union, hazardous substances are regulated to have a high level of protection to human health and the environment (European Chemical Agency 2017). Under REACH Regulation any new substance must be registered and evaluated, then can be authorised or restricted, but polymers do not have to be registered so far (Vaughan 2015). The European Commission has to review the exemption by strict criteria to include polymers in the REACH Regulation.

In 2015, the Group of 7 (G7) including Canada, France, Germany, Italy, Japan, the UK, and the US discussed plastic pollution in marine environments and confirmed that marine litter, especially plastic litter, poses a global challenge, which affects aquatic ecosystems and potentially also human health (G7 2015). Hence, they are trying to develop an action plan to combat marine litter and reduce waste from land- and sea-based sources. One of the priorities of the G7 action plan is "Investigating sustainable and cost-effective solutions to reduce and prevent sewage and storm water-related waste, including MPs entering the marine environment" (Brennholt et al. 2018). A unique opportunity was provided by Canada's G7 presidency to accelerate domestic action and demonstrate international leadership for reducing use and recycling of single-use plastics to ameliorate marine pollution effected by plastic litter (Walker and Xanthos 2018). In 2018, the ocean plastics charter was adopted by five-member nations of the G7 (Canada, France, Germany, Italy, and the UK) which includes 23 specific actions in five broad categories: (1) sustainable design, production and after-use markets; (2) collection, management and other systems and infrastructure; (3) sustainable lifestyles and education; (4) research, innovation and new technologies; and (5) coastal and shoreline action. The ocean plastics charter includes increasing recycling by at least 50% in plastic products by 2030, recycling and reuse of at least 55% of plastic packaging by 2030, and to recover 100% of all plastics by 2040, and developing research and technologies to remove plastics and MPs from waste water and sewage sludge (G7 2018). In October 2000, the European Union Water Framework Directive (WFD) has been enacted by the European Commission and focuses on "maintaining and improving the aquatic environment in the community". A Pollution Management and Environmental Health (PMEH) program was established by the World Bank in 2015, which covers technical assistance and financing to decrease pollution and improve health. The Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection (GESAMP) advocates cost-effective way to conduct urgent action to reduce the volume of plastics releasing the ocean by adopting the 3Rs (reduce-reuse-recycle circular economy) (GESAMP 2015). Current circular economy principles have been extended to include the 6Rs (reuse, recycle, redesign, remanufacture, reduce, recover) (Liu et al. 2018). In this way, the flow of technical materials is returned via society for remanufacturing by a material recovery of products and packaging designed, repair and reuse, and where these cases are not suitable, a biological material such as bioplastics may substitute. Today, bioplastics or plant-based plastics are viewed with new interest, which create reliable resources, decoupled from fossil fuels (Bioplastic Feedstock Alliance 2018).

The Packaging and Packaging Waste Directive 94/62/EC (PPWD), and the amending Directive (EU) 2015/720 are about use and reuse of plastic bags and reducing the consumption of lightweight plastic carrier bags (Xanthos and Walker 2017). There are three main directives relevant to the regulation of waste and recycling of plastic bags including the Waste Framework Directive 2008/98/EC (WFD), the Directive on the Landfill of Waste 1999/31/EC and Urban Waste Water Treatment Directive 91/271/EEC. The WFD conduct the Member States on the correct management of their wastes (European Parliament and the Council 2008). It introduces the waste hierarchy, which prioritizes prevention as the first and most important waste management part, followed by reuse/recycling, and finally, disposal/landfilling (European Parliament and the Council 2008). Furthermore, new recycling and recovery purposes for plastics are set with the aim of reusing and recycling at least 50% of the total plastic waste of households by 2020 (European Parliament and the Council 2008). In 1999, the Directive on the Landfill of Waste 1999/31/EC was implemented to eliminate adverse impacts of the landfills to aquatic ecosystems and human health. At present, there are no restrictions to the quantity of plastic wastes, which deposit as non-hazardous waste in landfills within the Directive of Landfill of Waste (The Council of the European Union 1999); however, the Member States have individually implemented threshold restrictions for disposal of plastic wastes in landfills (PlasticsEurope 2015). In the last decades, several Member States (Germany, Denmark, Sweden, and Austria) banned landfilling of the plastics, which led to significant increase in the recovery of plastic waste in these countries (Steensgaard et al. 2017). The Urban Waste Water Treatment Directive (91/271/EEC) was implemented to maintain and protect aquatic ecosystems, setting thresholds for the discharge of wastewater to the environment to prevent adverse effects such as eutrophication (European Commission 1991).

Extended producer responsibility (EPR) is a public policy strategy that manufacturers are responsible legally and financially to mitigate the environmental impacts of their products throughout its life cycle stages. EPR emerged in Sweden and Germany in the early 1990s and had several desirable and interrelated goals including creating motivations for eco-design of packages and products, using the private sector expertise to reach public targets, internalizing the waste management costs into prices of the products, and shifting waste management costs from municipalities and taxpayers to producers and consumers (Lifset et al. 2013). The EU's Waste of Electrical and Electronic Equipment (WEEE) Directive is an example of EPR whereby producers have to return and recycle electronic equipment (Eriksen et al. 2018).

Previous studies revealed significant relationships between personality traits and cognitive abilities with waste-management behaviours (Karbalaei et al. 2013, 2014, 2015). For example, Swami et al. (2011) showed that individuals with less machiavellian, less politically cynical, and more conscientious traits showed better waste management behaviours. This may be because conscientiousness is associated with intellectualism and attend to acting based on the dictates of conscience. To the extent that, waste management is realised as a morally-appropriate behaviour, thus, highly conscientious individuals are more responsible to recycle, reuse, and reduce their waste. Similarly, another study conducted by Karbalaei et al. (2015) revealed that individuals with higher spiritual intelligence and lower machiavellianism were more likely to have a positive attitude towards waste-prevention behaviours. Therefore, waste management strategies need to consider personality traits and individual differences that affect environmentalism and could also be considered as a promising strategy to mitigate MP pollutions in the marine and terrestrial environments by improving positive personality traits in consumers, private and government sectors through education. This also could be achieved by education of students in schools and universities as it can provide a long-lasting solution to the environmental problem.

A continued relationship between science and policy can contribute to solutions for mitigation of MPs in the environment. Also, new scientific understanding could help scientists and policymakers to conduct policy-relevant research. Some regulations mentioned in this study are the results of the collaboration between scientists and policymakers which lead to establishing positive changes toward mitigation of MPs. Legislation to ban plastic microbeads is a good illustration of this collaboration because researchers showed that release of microbeads could be easily curbed, and therefore risk to marine life mitigated. Therefore, legislation to ban this source of plastic contamination has been introduced in the US, Canada, the European Union, and Australia (Rochman et al. 2016). In response to this legislation, some manufacturers have agreed to voluntarily remove plastic microbeads from their products (Schnurr et al. 2018). Recently, scientists have developed biodegradable cellulose microbeads from a sustainable source that could be a promising replace of persistent plastics microbeads in a range of applications from personal care products to abrasives (Coombs Obrien et al. 2017) and this could be obliged through legislation in the future. The report of plastic pellets in the oceans, on beaches, and in the digestive systems of seabirds lead to a response of both policy and industry to resolve this problem by Operation Clean Sweep, which was initiated by the plastics industry to reduce the loss of plastic pellets to the environment (Rochman et al. 2016). Thus, further studies are required in the area of MPs in order to be able to develop and implement effective management strategies.

Biodegradation of plastic polymers by some organisms such as bacteria, fungi and mealworms are reliable and environmentally safe action plan to tackle plastic pollution that will enable the management of MPs without negative effects. A recent study by Bombelli et al. (2017) showed fast bio-degradation of PE by larvae of the wax moth *Galleria mellonella*, producing ethylene glycol. Similarly, low-density polyethylene (LDPE) MP particle decayed with isolated bacteria from the gut of the earthworm (*Lumbricus errestris*) (Lwanga et al. 2018). Therefore, these organisms have prompted significant optimism about the use of "plastic eating organisms" in waste management. Governments need to fund most significantly additional researches and innovations to find organisms that will break down plastic more efficiently.

Collectively, all these strategies help to mitigate the presence of MPs in the terrestrial and aquatic environments. Industry plays a critical role in mitigation of MPs throughout the supply chain. As an example, IKEA has used EPR strategy in its business model by promoting reuse and recycling of materials throughout its supply chain (INGKA Holding 2017). Also, in 2017, Adidas sold 1 million shoes made from plastic debris, equivalent to 16.5 million plastic bottles and 14.3 t of nylon gill nets (Kharpal 2018). Improvements of circular economy principles such as recycling and waste management strategies can also be a catalyst in the reduction of plastic consumption with strong direct and indirect socioeconomic and environmental implications. Plastic pollution mitigations through coastal and ocean cleanups are important immediate activities that are needed to help reduce marine plastic pollution. Plastic bag bans can effectively reduce overuse of single-use plastic and following that mitigating plastic and MP pollutions if properly implemented and managed (Schnurr et al. 2018). Besides, plastic manufacturers must ensure that their products are standardised and labelled properly to facilitate recycling. Furthermore, increasing awareness through universities, school, organisations and networks about MPs issues by campaigns and educate personal responsibility of individuals for mitigating plastic and MPs pollution by choosing to reject, reduce, reuse and recycle plastics. Educational intervention (e.g. marine litter education) to children boosts their awareness, perceptions of consequences and self-reported action (Hartley et al. 2015).

## Conclusion

MPs are very tiny particles of plastics that find their way into the environment through primary and secondary sources. The presence of MPs in air, soil, and particularly in aquatic environments has become the focus of a dearth of environmental pollution research. This, combined with recent discoveries of MPs in plates of seafood, sea salt, canned fish, bottled water, tap water, honey, and sugar is an emerging area of concern related to the potential impacts of this plastic debris to human health. Potential health concerns have been related to the toxicity of harmful chemicals sorbed from the environment or from additives that are used in the plastic production process itself. As this review showed, lack of studies on MPs impacts on humans highlight the need for more studies focusing on human health risk assessment of MPs. Recently, some national regulations have been proposed or established to help reduce MPs in the environment. However, there are currently no regulations established to manage impacts from secondary MPs (fragments from larger plastic items). Involving the general public, the media, the socio-economic sectors, tourism, and companies is necessary to tackle the issue. New propose, new national, and international regulations should be established to prevent the exceeding critical environmental threshold concentrations. Implementing internationally harmonised regulations across developed and developing countries such as the circular economy could help proper waste management. Policies to reduce single-use plastics including, bans of single-use plastic bags, drinking straws, deposit and return plans for plastic bottles and EPR, which makes manufactures responsible for the entire product lifecycle, are positive steps that should be widely implemented.

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