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Microplastics in the environment: A critical review of current understanding and identification of future research needs☆

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ABSTRACT

Microplastics (plastic particles <5 mm) are a contaminant of increasing ecotoxicological concern in aquatic environments, as well as for human health. Although microplastic pollution is widespread across the land, water, and air, these environments are commonly considered independently; however, in reality are closely linked. This study aims to review the scientific literature related microplastic research in different environmental compartments and to identify the research gaps for the assessment of future research priorities. Over 200 papers involving microplastic pollution, published between 2006 and 2018, are identified in the Web of Science database. The original research articles in 'Environmental Sciences', 'Marine/Freshwater Biology', 'Toxicology', 'Multidisciplinary Sciences', 'Environmental Studies', 'Oceanography', 'Limnology' and 'Ecology' categories of Web of Science are selected to investigate microplastic research in seas, estuaries, rivers, lakes, soil and atmosphere. The papers identified for seas, estuaries, rivers and lakes are further classified according to (i) occurrence and characterization (ii) uptake by and effects in organisms, and (iii) fate and transport issues. The results reveal that whilst marine microplastics have received substantial scientific research, the extent of microplastic pollution in continental environments, such as rivers, lakes, soil and air, and environmental interactions, remains poorly understood.

1. Introduction

The amount of anthropogenic litter in aquatic and terrestrial environments has increased dramatically over the last few decades; approximately 60–80% of which is plastic (Derraik, 2002). Mass production of plastics began in the 1950s, and currently exceeds 280 million tones globally (PlasticsEurope, 2017). It is estimated that 4.8 to 12.7 million metric tons of mismanaged plastic waste enters the oceans from coastal countries each year (Jambeck et al., 2015). Microplastics can be defined as tiny plastic particles smaller than 5 mm in size, which originate from primary and secondary sources (Cole et al., 2011; Horton et al., 2017b). Primary source microplastics include polyethylene (PE), polypropylene (PP), and polystyrene (PS) particles in cosmetic and medical products (Horton et al., 2017b). Due to their adverse effects in the environment, the sale of cosmetic products containing microplastics has been banned in several countries, including Canada and the United States (Ballent et al., 2016). Secondary microplastics originate from physical, chemical, and biological processes resulting in fragmentation of plastic debris (Thompson, 2006; Ryan et al., 2009). Exposure to ultraviolet (UV) radiation catalyzes the photo-oxidation of plastic, causing it to become brittle and fragment into microplastics. While the heat and sunlight, and the well-aerated conditions are ideal for generating microplastics through iterative fragmentation processes, the cold and anoxic conditions of aquatic environments and sediments can cause very slow degradation of plastic particles for centuries (Harshvardhan and Jha, 2013; Zhang, 2017). Different sources of microplastics cause them to occur in diverse shapes such as pellets, fibers, and fragments in environmental samples (Klein et al., 2015).

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Primary microplastics are most likely entering the aquatic environment through household sewage discharge or spillage of plastic resin powders or pellets used for airblasting (Gregory, 1978, 1996). Another significant source of primary microplastics is the application of sewage sludge containing synthetic fibers or sedimented microplastics from personal care or household products to land (Horton et al., 2017b). Fibers are the most commonly reported form (Browne et al., 2011), most likely due to the continual abrasion of clothes and upholstery made from synthetic textiles, and washing machine effluent release (Napper and Thompson, 2016). Although synthetic fibers primarily made of polyester, acrylic, and polyamide, are secondary microplastics, they are released to the environment along with primary microplastics (Horton et al., 2017b). It has been estimated that 1900 fibers per item may come out during washing, and be released to aquatic and terrestrial environments through wastewater effluents and sewage sludge applications (Browne et al., 2011). In this context, textile mills could also be a point-source release to the environment; which has not been investigated. Areas in proximity to plastics industry are predicted to be hotspots; concentrations of approximately 100 000 plastic particles per m^3 of seawater have been reported in a Swedish harbor area adjacent to a polyethylene (PE) production plant (Noren and Naustvoll, 2010).

Secondary sources of microplastics are considered as a great contributor of microplastic pollution given the large amount of macroplastic wastes entering the environment (Duis and Coors, 2016). Secondary microplastics, originate from anthropogenic activities, such as littering and are released during municipal solid waste collection and disposal processes (Horton et al., 2017b). These large plastic items and their degraded products may be introduced to aquatic environments by wind dispersal, soil erosion or surface runoff. Likewise, light macro- and microplastics can be transported across the land by wind, while denser polymers are more likely to be buried deeper in soil layers (Horton et al., 2017b). Surface runoff from agricultural lands and urban areas is another significant source of microplastic load to surface waters. Recent studies suggest that agriculture is one of the main anthropogenic activities that contribute to microplastic pollution in soil both due to the application of sewage sludge for soil amendment, and the use of agricultural plastics, such as plastic mulches to increase the crop yield (Nizzetto et al., 2016b; Rodríguez-Seijo and Pereira, 2019). Additionally, there is evidence to suggest that tires and road markings may also cause microplastic pollution, stormwater runoff acting as a prominent transport pathway for carrying tire and road wear particles (TRWP) to surface waters (Horton et al., 2017a; Kole et al., 2017; Unice et al., 2019a). Moreover, recent studies have demostrated that large amounts of fibers have been transported, particularly in highly urbanized areas via atmospheric fallout (Dris et al., 2016; Cai et al., 2017). Possible sources of airborne microplastics include, synthetic fibers from clothes and houses, artificial turf, landfills and waste incineration (Dris et al., 2016; Magnusson et al., 2016). These particles in the atmosphere can be transported by wind to the aquatic environment or deposited on the terrestrial environment. Consequently, spatial distribution of microplastics between different environmental compartments are shaped by physical processes, such as wind, tides, surface runoff and flooding that change by climatic forces (Zhang, 2017). Sources, sinks, and pathways of microplastic transport between terrestrial, freshwater, and marine environments are illustrated in the graphical abstract (Modified and adapted from Critchell and Lambrechts, 2016 and Horton et al., 2017b).

Microplastics are of increasing concern in aquatic environments due to the ecotoxicological risks they pose. Microplastic ingestion by a range of species can compromise energy reserves, and can bioaccumulate and biomagnify through the food chain. Moreover, due to their relatively large surface area and hydrophobic composition, they are prone to adsorbing many substances including heavy metals (Cole et al., 2011; Avio et al., 2016; Wang et al., 2017a), and may transfer priority pollutants, such as Polycyclic Aromatic Hydrocarbons (PAHs) and Polychlorinated Biphenyls (PCBs), to aquatic life (Frias et al., 2010; Bakir et al., 2014a; Bakir et al., 2014b; Klein et al., 2015). Sorption behavior of persistent organic pollutants (POPs) with changing water quality parameters and polymer types has been subject of research, especially, over the last few years (Lee et al., 2014; Velzeboer et al., 2014; Huffer and Hofmann, 2016).

This review aims (i) to present the current state of microplastics research and provide a comprehensive overview of microplastic abundance in six different environmental compartments: sea, estuary, river, lake, soil and atmosphere, and, (ii) to identify the research gaps to guide for future research priorities.

2. Methodology

This paper reviews, in detail, a range of key studies concerning microplastic pollution in environmental matrices. It should be acknowledged that this review provides a cross sample of the studies, and not every paper dealing with microplastics has been included. Here, over 200 papers published between 2006 and 2018 are reviewed. Web of Science Core Collection is selected as the search database. First of all, articles with the keyword 'Microplastic' are derived from the database in eight different categories including; 'Environmental Sciences', 'Marine/Freshwater Biology', 'Toxicology', 'Multidisciplinary Sciences', 'Environmental Studies', 'Oceanography', 'Limnology' and 'Ecology' (Fig. 1a). Secondly, different environmental compartments; seas, estuaries, rivers, lakes, soil and atmosphere are taken into consideration to investigate microplastic research (Fig. 1b). Thirdly, these papers are further subcategorized according to the issues investigated; (i) occurrence and characterization, (ii) uptake by and effects in organisms, and (iii) fate and transport (Fig. 1c). Finally, research gaps and inadequately studied areas are determined and discussed, in full.

3. Microplastic research in the environment

Considering the eight selected categories of Web of Science Core Collection database, distribution of microplastic research in

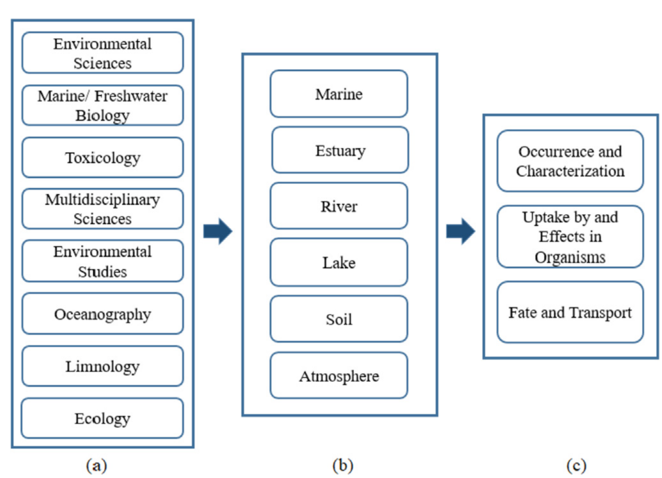

Fig. 1. Research methodology. (a) Selected categories of Web of Science Core Collection database. (b) Categories of environmental compartments. (c) Subcategories of research issues.

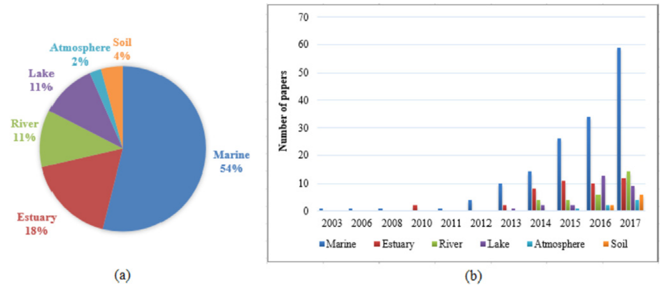

Fig. 2. Microplastic related research in Web of Science database (retrieved on 15/11/2018). (a) Distribution in environmental systems (b) number of publications by year.

different environmental compartments including seas, estuaries, rivers, lakes, soil and atmosphere and number of publications by year are determined (Fig. 2). According to our viewpoint, microplastic related studies in marine, estuary, river, and lake are classified under three main categories, being (i) occurrence and characterization (ii) uptake by and effects in organisms, and (iii) fate and transport (Fig. 3). Microplastic research in the soil and atmosphere are not further categorized because of the limited number of the studies related to the subject.

3.1. Marine environment

Oceans are generally considered as the ultimate sinks for microplastics (Horton and Dixon, 2018). Research into marine microplastics has started to gain attention since first highlighted in the 1970s (Carpenter and Smith, 1972). Here, 166 research papers investigating microplastics in seas/oceans are reviewed in three different subcategories: Occurrence and characterization (76 articles), uptake by and effects in organisms (62 articles), and fate and transport (29 articles). Some publications fall into more than one category.

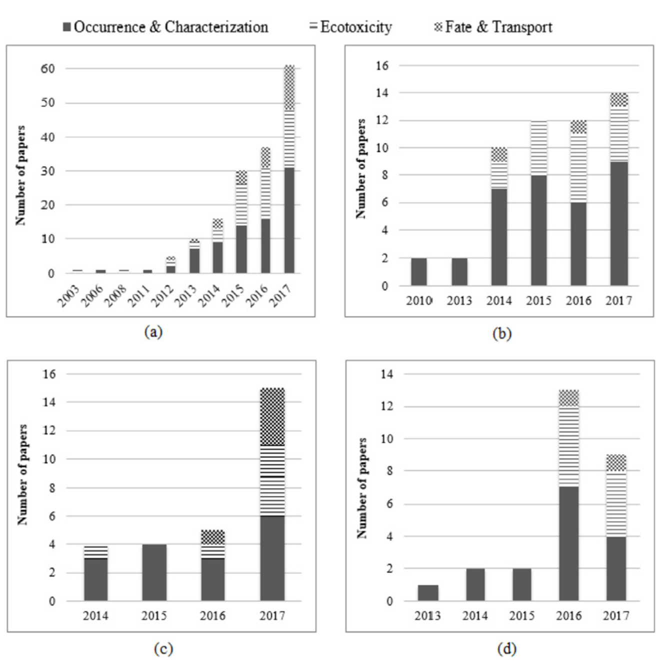

Fig. 3. Microplastic related research in Web of Science database (retrieved on 15/11/2018). Number of publications by subject issue for (a) marine (b) estuary (c) river and (d) lake.

3.1.1. Occurrence and characterization

Microplastics are considered as a significant component of marine debris; however, assessing the abundance, density and distribution of these contaminants within the marine environment is difficult due to spatial and temporal variability owing to oceanic currents and seasonal patterns (Doyle et al., 2011; Ryan et al., 2009; Cole et al., 2011). Nevertheless, scientific research has mainly focused on occurrence of microplastics in the marine environment. According to a model developed by Eriksen et al. (2014), approximately 5.25 trillion particles weighing 268940 tons float in the world's oceans. The knowledge on microplastic pollution in all matrices, such as sea surface (de Lucia et al., 2014; Chae et al., 2015; Castillo et al., 2016; Anderson et al., 2017), subsurface waters (Ivar do Sul et al., 2013; Enders et al., 2015; Lusher et al., 2015b; Kanhai et al., 2017) and sediments (Claessens et al., 2011; Vianello et al., 2013; Qiu et al., 2015; Lots et al., 2017; Mistri et al., 2017; Munari et al., 2017) has rapidly increased to date. According to articles that are considered in this paper, majority of the studies related with microplastic research in the marine environment have mainly based on field measurements of microplastics in surface and subsurface waters. Microplastics have been reported in surface and subsurface waters of the Atlantic Ocean (Ivar do Sul et al., 2013; Ivar do Sul et al., 2014; Lusher et al., 2014; Enders et al., 2015; Reisser et al., 2015; ter Halle et al., 2016; Debroas et al., 2017; Kanhai et al., 2017), North-eastern Pacific Ocean (Goldstein et al., 2013; Desforges et al., 2014; Mendoza and Jones, 2015; Díaz-Torres et al., 2017), Arctic Polar waters (Lusher et al., 2015b), as well as in surface waters of the North Sea (Dubaish and Liebezeit, 2013; Dekiff et al., 2014), Adriatic (Gajšt et al., 2016), Bohai and South China Seas (Zhang et al., 2017a,b; Cai et al., 2018). There are numerous studies investigated the occurrence of microplastics in surface waters and sediments of the Mediterranean Sea including coastline of Italy (de Lucia et al., 2014; Faure et al., 2015a,b; Panti et al., 2015; Pedrotti et al., 2016; Ruiz-Orejón et al., 2016; Suaria et al., 2016), Greece (Ioakeimidis et al., 2014), Turkey (Gündoğdu and Çevik, 2017; Güven et al., 2017), Spain (Alomar et al., 2016), Israel (van der Hal et al., 2017) and France (Schmidt et al., 2018). Microplastic pollution has also been studied in deep sediments of the Central Adriatic (Mistri et al., 2017), Baltic Seas (Gewert et al., 2017; Zobkov and Esiukova, 2017), coastal beach sediments of South Korea (Heo et al., 2013; Lee et al., 2015; Eo et al., 2018), Northern Adriatic (Munari et al., 2017), and Saudi Arabian Red Sea (Ruiz-Compean et al., 2017). These studies mostly focus on the investigation of the abundance and spatio-temporal distribution of microplastics, as well as the classification of these synthetic particles according to their shapes, sizes, colors and types.

Fishery, marine vessels and marine industries are direct sources of microplastics entering to the marine environment as in the form of macroplastics that will form secondary microplastics following long-term degradation (Cole et al., 2011). Majority of marine microplastics originate from land-based sources; such as degrading beach litter, freshwater sources; such as stormwater runoff, wastewater discharge, and atmospheric deposition of airborne microplastics (Ryan et al., 2009; Zhang, 2017). Identification of microplastic composition has been a prominent issue in the research area for accurate determination of the potential sources of plastic particles (Castillo et al., 2016; Nel et al., 2017; Lo et al., 2018). Fibers are often the dominant microplastic form found in marine sediments (Claessens et al., 2011; Frias et al., 2016; Martin et al., 2017), surface waters and water column (Lusher et al., 2015b; Aytan et al., 2016), as well as in biota (Neves et al., 2015; Bellas et al., 2016; Güven et al., 2017). Martin et al. (2017) defined fibers being the principal form of microplastic pollution (85%) in samples collected from the sediments and bottom water from the Irish

Continental Shelf. Furthermore, majority of microplastic composition in the Arctic Polar waters are defined as fibers (95%) (Lusher et al., 2015b). Large proportion of synthetic fibers suggests that they may either be derived from wastewater as a consequence of laundry activities (Frias et al., 2016), or result from the local vessel activity and be transported over large distances by surface currents (Lusher et al., 2015b). Fibers are generally followed by fragments as the most dominant microplastic form in the marine environment (Lusher et al., 2015b; Frias et al., 2016; Martin et al., 2017). Synthetic polymers identified in the various environmental matrices mostly include PE, PP and PS. For example, Ng and Obbard (2006) found PE, PP and PS both in the surface microlayer and subsurface layer of coastal waters, as well as in beach sediments of Singapore coastline. A study in the Italian coast (Vianello et al., 2013) found PE (48%) and PP (38%) as the most predominant microplastics in sediment samples, in line with the most commonly produced polymers (Frias et al., 2016). According to a recent study implemented in the surface waters of the Bohai Sea (China), the main microplastic types were identified as PE, PP and PS (Zhang et al., 2017a,b). The authors also revealed the relationship between polymer size and type, upon observing that as the size of the plastics decreased, the percentage of PP increased, whereas the percentages of PE and PS decreased.

3.1.2. Uptake by and effects in organisms

Microplastic uptake by marine organisms represents a growing concern due to toxicological risks associated with these micro particles. Small dimensions of microplastics increase their bioavailabilities to aquatic organisms. To date, ingestion has been considered as the fundamental pathway for marine species to uptake microplastics. Many of the studies have proved the ingestion of microplastics by detecting different type of polymer particles in the intestines and stomachs of various organisms, such as zooplanktons (Setälä et al., 2014; Desforges et al., 2015; Jeong et al., 2017; Sun et al., 2017), shellfish (Browne et al., 2008; Van Cauwenberghe et al., 2015; Li et al., 2016; Gandara E Silva et al., 2016; Tosetto et al., 2016), corals (Hall et al., 2015; Reichert et al., 2018), fish (Neves et al., 2015; Bellas et al., 2016; Hermsen et al., 2017; Jabeen et al., 2017), and marine mammals (Eriksson and Burton, 2003; Lusher et al., 2015a; Fossi et al., 2016; Nelms et al., 2018). Ingestion may occur directly due to misidentification or indiscriminate consumption of microplastics for feeding (Ory et al., 2017), or indirectly as a result of trophic transfer along the food web (Nelms et al., 2018). Setälä et al. (2014) demonstrated the presence and transfer of microplastics along planktonic food web examining zooplankton samples collected from the Baltic Sea. Nelms et al. (2018) observed the trophic transfer of microplastics from mackerel (*Scomber scombrus*), caught within the Celtic Sea, to grey seals (*Halichoerus grypus*). Transfer of microplastics along the food chain was also confirmed under laboratory conditions from mussel (*Perna perna*) to crab (*Callinectes ornatus*) and to the puffer fish (*Spheoeroides greeleyi*) (Santana et al., 2017), as well as from seaweed (*Fucus vesiculosus*) to periwinkle (*Littorina littorea*) (Gutow et al., 2016). In addition to trophic transfer, adverse effects of microplastics on marine organisms have also been investigated performing both lab-based and field-based experiments. According to a study carried out on the blue mussel *Mytilus edulis*, after exposure to microplastics under laboratory conditions, it was observed that smaller microplastics tend to accumulate in the tissues more than larger particles, and can translocate from the gut to the circulatory system within three days and persisted for over 48 days (Browne et al., 2008). Additionally, impact of microplastics alone or in combination with toxic chemicals including heavy metals, PCBs, PAHs and polybrominated diphenyl ethers (PBDEs) have been examined performing exposure experiments for microalgae (*Tetraselmis chuff*), Norway lobsters (*Nephrops norvegicus*), common gobies (*Pomatoschistos microps*), and amphipods (*Allorchestes compressa*); respectively (Oliveira et al., 2013; Chua et al., 2014; Davarpanah and Guilhermino, 2015; Devriese et al., 2017).

The number of microplastic studies related with invertebrates are relatively high. Numerous studies have investigated the ingestion, translocation and bioaccumulation of microplastics within shellfish including mussels (Browne et al., 2008; von Moos et al., 2012; Avio et al., 2015; Van Cauwenberghe et al., 2015; Li et al., 2016; Paul-Pont et al., 2016), oysters (Cole and Galloway, 2015; Green, 2016; Sussarellu et al., 2016), clams (Green, 2016; Ribeiro et al., 2017) and crabs (Brennecke et al., 2015; Santana et al., 2017). For example, von Moos et al. (2012) revealed the significant effect of high-density polyethylene (HDPE) particles on cells and tissue of blue mussel. According to other studies, prolonged exposure to polystyrene significantly alter the feeding capacity of the copepod *Calanus helgolandicus* (Cole et al., 2015), whereas no significant effect was observed on Pacific oyster (*Crassostrea gigas*) (Cole and Galloway, 2015). Field-based experiments reported the uptake of microplastics by the mussel *Mytilus edulis* and the lugworm *Arenicola marina* collected from the six locations along the French-Belgian-Dutch coastline (Van Cauwenberghe et al., 2015). Similarly, Li et al. (2016) investigated microplastics in mussels (*Mytilus edulis*) from coastlines of China and found fibers as the most common microplastics followed by fragments. The authors proposed that mussels can be used as a potential bioindicator of microplastic pollution in the coastal environment. Field-based studies have also been implemented to represent the presence of microplastics in zooplankton samples from the Northeast Pacific Ocean (Desforges et al., 2015) and Baltic Sea (Setälä et al., 2014), fiddler crabs collected from Itaipu Lagoon, Brazil (Brennecke et al., 2015), benthic invertebrates from the Northeast Atlantic Ocean (Courtene-Jones et al., 2017), as well as in pelagic and benthic fish species collected from the English Channel (Lusher et al., 2013), the North Sea (Hermsen et al., 2017), the Atlantic and the Mediterranean coasts (Neves et al., 2015; Bellas et al., 2016; Avio et al., 2017; Güven et al., 2017), and the Persian Gulf (Akhbarizadeh et al., 2018). Most of these studies have shown that the majority of ingested particles consist of fibers, which are generally in the range of 66–71% of the total count and followed by fragments and pellets (Lusher et al., 2013; Bellas et al., 2016; Güven et al., 2017).

Microplastics have also been found in different organs besides the digestive tract, which are not involved in the process of ingestion (Kolandhasamy et al., 2018). In addition to stomach and hepatopancreas, Brennecke et al. (2015) observed microplastics also in the gills of the fiddler crab. Moreover, Kolandhasamy et al. (2018) investigated the adherence of microplastics to soft tissues of mussels as a novel way for organisms to uptake microplastics beyond ingestion. The authors found that adherence of microplastics to specific organs; such as adductor, foot and visceral tissue contributed about 50% of the microplastic uptake in mussels.

3.1.3. Fate and transport

Fate and transport of microplastics in the marine environment have generally been investigated using mathematical models that are based on principles of fluid mechanics and hydrodynamics (Law et al., 2010; Lebreton et al., 2012; Ballent et al., 2013; Critchell and Lambrechts, 2016; Sherman and Van Sebille, 2016; Bagaev et al., 2017; Chubarenko and Stepanova, 2017; Iwasaki et al., 2017). Water currents, wind and tides are the main components of microplastic transport in the marine environment, which lead to prolonged sinking process of micro particles and migrations in sea coastal zone (Bagaev et al., 2017). Buoyant polymer particles, such as PE and PP tend to be transported on sea surface, whereas denser polymers, such as PVC may still be transported by underlying currents (Engler, 2012; Wang et al., 2016). Bagaev et al. (2017) indicate that after reaching bottom, microfibers are captured for a certain time period by higher turbulence in the benthic boundary layer and/or resuspended by bottom currents. Numerical models have been developed to investigate the transport of macro and/or microplastics by surface currents and wind waves in the North Atlantic subtropical gyre (Law et al., 2010), world's oceans (Lebreton et al., 2012), eastern North and South Pacific Oceans (Law et al., 2014), Baltic Sea (Bagaev et al., 2017; Chubarenko and Stepanova, 2017) and Sea of Japan (Iwasaki et al., 2017). The model developed by Law et al. (2010) reveals that plastic pollution can quickly migrate from the US eastern seaboard to North Atlantic subtropical gyre in less than 60 days. One other advection-diffusion model developed by Critchell and Lambrechts (2016) demonstrates the relative importance of physical processes, such as beaching, settling, re-floating, degradation and wind drift governing macro- and microplastic accumulation and reveals that the topography of the source location has by far the largest influence on the fate of the microplastics.

Particle density and size distribution are dynamically changing parameters, which needs to be described explicitly to develop a comprehensive fate and transport modeling framework (Enders et al., 2015; Zhang, 2017). Several studies in the literature investigated microplastic distribution in water column, have shown high dispersal of small microplastics over surface mixed layer, and dependence on the size, density and shape of microplastic (Enders et al., 2015; Kooi et al., 2016; Kowalski et al., 2016). Indeed, most of the synthetic polymers, such as PE and PP are buoyant (Wang et al., 2016); however, aggregation with organic and inorganic particles or fouling with microbial organisms can increase the size and density of microplastics, which accelerate settling of microplastic particles onto deep sediments. Although PE and PP have densities less than that of fresh water, they have been regularly identified in deep sediments (Claessens et al., 2011; Vianello et al., 2013; Corcoran et al., 2015). Given the high concentrations of suspended sediments, particulate organic matter, and detrital particles in the aquatic environment, aggregation, biofouling and subsequent sedimentation might dominate the fate and transport of microplastics in aquatic environments (Besseling et al., 2017; Kooi et al., 2017; Long et al., 2017; Zhang, 2017). Kooi et al. (2017) developed a theoretical model based on settling, biofilm growth, and ocean depth profiles for light, water density, temperature, salinity, and viscosity to simulate the effect of biofouling on the fate of microplastics, and predicted size-dependent vertical transport of microplastics in the ocean. According to their outcomes, after initial settling, buoyant microplastics never settle a fixed water level, but keep moving up and down in the water column. The density of seawater generally increases with depth, so that oscillations of micro particles in water column can be explained by dynamics based on the density differences between seawater and the plastic particle. Microplastic sinking and bioavailability may also differ among species (Long et al., 2015; Katija et al., 2017; Long et al., 2017). For example, Long et al. (2015) found that diatom aggregates sunk faster than cryptophyte aggregates, which is explained by the frustule made of biogenic silica that is denser than the organic matter. Fractal and porous particles impact the sinking rate of aggregates (Li and Yuan, 2002; Xiao et al., 2012), in which water can flow through easily, increasing the settling velocity (Li and Yuan, 2002; Long et al., 2015). Kowalski et al. (2016) also revealed that weathering and biofouling may alter the sinking behavior of microplastics considerably, yet the particle shape strongly affects the sinking velocity, as well. Since deep sediments are considered as an ultimate sink for marine microplastics (Woodall et al., 2014), similar studies addressing vertical transport of microplastics are encountered in the literature (Enders et al., 2015; Bagaev et al., 2017; Katija et al., 2017; Näkki et al., 2017). Näkki et al. (2017) found that bioturbation, which covers all the actions of benthic fauna changing the sediment structure, transports microplastic particles deeper within marine sediments. Katija et al. (2017) also revealed that giant larvaceans *Bathochordaeus stygius* can contribute to the vertical transport of microplastics by packaging microplastic particles into their fecal pellets. Physical, chemical, and biological factors likely to affect the settling of microplastics in marine waters are given in Table 1.

Table 1
Factors driving microplastic settling (Modified and adapted from Harrison et al. (2018)).

Governing Issues	Effective Factors
Surface chemistry and structure	Polymer type
	Adsorbed and leaching chemicals
	Age/weathering
	Particle size
Biological interactions	Pioneer colonizers
	Successional stage
	Competition
	Ingestion
Local environmental conditions	Temperature
	Oxygen
	Nutrients
	Light
	Salinity
	Pressure
	Presence of other pollutants
Movement and transport between habitats	Buoyancy
	Flocculation
	Particle spiraling
	Flooding
	Currents
Biogeography	Geographic location

In the context of the review, the studies investigated the role of microplastics on fate and transport of toxic chemicals in the marine and coastal environments are also included in this section. Several studies have shown that microplastics act as a vector in the transport of POPs; such as hexabromocyclododecane (HBCD) (Jang et al., 2017), PAHs (Mendoza and Jones, 2015; Lee et al., 2017), PCBs (Mendoza and Jones, 2015), dichlorodiphenyltrichloroethane (DDT), phenanthrene (Phe) and bis-2-ethylhexyl phthalate (DEHP) (Bakir et al., 2016; Zhang et al., 2018) from seawater to marine organisms. However, investigations of the mechanisms for the adsorption of heavy metals to microplastic particles remain relatively unexamined (Brennecke et al., 2016). There are a few studies investigating the interaction between microplastics and heavy metals in the marine system (Brennecke et al., 2016; Massos and Turner, 2017).

3.1.4. Existing research

Summary of microplastic studies in the marine environment is given in Table 2.

3.2. Estuaries

According to our viewpoint, majority of the microplastic studies in the estuarine environment are categorized under occurrence and characterization (36 articles), followed by uptake by and effects in organisms (20 articles) and fate and transport (3 articles), which are given in detail below.

3.2.1. Occurrence and characterization

Accumulation of microplastics in estuaries is high due to input of anthropogenic debris from freshwater systems and beaches and

Table 2
Microplastic research in the marine environment.

Study	Focus of the Study	Sample Type	Governing Process
Ng and Obbard (2006)	Abundance, spatial distribution and composition of microplastics in the marine compartments	Beach sediments, surface and subsurface water	Occurrence and transport
Claessens et al. (2011), Vianello et al. (2013), Qiu et al. (2015), Alomar et al. (2016), Fastelli et al. (2016), Bergmann et al. (2017), Cannas et al. (2017), Mistri et al. (2017), Naji et al. (2017a), Naji et al. (2017b)	Abundance, spatial distribution and composition of microplastics in the marine compartments	Sediment	Occurrence and transport
Liebezeit and Dubaish (2012), Heo et al. (2013), Hidalgo-Ruz and Thiel (2013), Jayasiri et al. (2013), Dekiff et al. (2014), Laglbauer et al. (2014), Fauziah et al. (2015), de Carvalho and Baptista Neto (2016), Retama et al. (2016), Esiukova (2017), Hengstmann et al. (2017), Lots et al. (2017), Munari et al. (2017), Ruiz-Compean et al. (2017), Eo et al. (2018), Yu et al. (2018)	Abundance, spatial distribution and composition of microplastics in the marine compartments	Beach sediments	Occurrence and transport
Dubaish and Liebezeit (2013), de Lucia et al. (2014), Desforges et al. (2014), Lusher et al. (2014), Chae et al. (2015), Faure et al. (2015b), Gago et al. (2015), Isobe et al. (2015), Lusher et al. (2015b), Song et al. (2015), Gajšt et al. (2016), Isobe (2016), Pedrotti et al. (2016), Ruiz-Orejón et al. (2016), Suaria et al. (2016), Anderson et al. (2017), Gewert et al. (2017), Gündoğdu and Çevik (2017), Kanhai et al. (2017), van der Hal et al. (2017), Zhang et al. (2017a,b), Cai et al. (2018), Schmidt et al. (2018)	Abundance, spatial distribution and composition of microplastics in the marine compartments	Surface and/or subsurface water	Occurrence and transport
Frère et al. (2017)	Abundance, spatial distribution and composition of microplastics in the marine compartments	Surface water and sediment	Occurrence and transport
Kooi et al. (2016), Öztekin and Bat (2017)	Abundance, spatial distribution and composition of microplastics in the marine compartments	Water column	Occurrence and vertical transport
Courtene-Jones et al. (2017)	Abundance, spatial distribution and composition of microplastics in the marine compartments	Deep sea water and benthic macroinvertebrates	Vertical transport and ingestion
Goldstein et al. (2013)	Abundance, composition and size characteristics of microplastics	Surface and subsurface water	Occurrence and transport
Ioakeimidis et al. (2014), Abidli et al. (2017), Blašković et al. (2017)		Sediment	
Lee et al. (2015)		Beach sediments	
Abayomi et al. (2017)		Beach sediment and sea surface	
Frias et al. (2016), Yu et al. (2016), Nelms et al. (2017), Lo et al. (2018)	Sources and sinks, abundance and composition of microplastics	Coastal sediment	Occurrence, fragmentation and transport
Castillo et al. (2016), Díaz-Torres et al. (2017)		Surface water	Occurrence and transport
Graca et al. (2017)		Deep and beach sediment	
Martin et al. (2017)		Sediment and bottom water	
Nel et al. (2017)		Beach sediment and surface water	
Panti et al. (2015), Aytan et al. (2016), Di Mauro et al. (2017)	The abundance and distribution of microplastics relative to the zooplankton	Surface water plankton samples	Occurrence and transport
Ivar do Sul et al. (2013), Ivar do Sul et al. (2014)	The presence of microplastics in zooplankton	Subsurface plankton samples	Ingestion
Desforges et al. (2015), Sun et al. (2017)		Calanoid copepod (*Neocalanus cristatus*) and the euphausiid (*Euphausia pacifia*), chaetognaths, jellyfish, shrimps, and fish larvae	
Long et al. (2015), Long et al. (2017)	Interactions between microplastics and phytoplankton	PS and microalgae species (Cryptophyceae, bacillariophyceae, prymnesiophycea, dinoflagellate and diatom)	Aggregation, vertical transport
Browne et al. (2008), Hamer et al. (2014), Brennecke et al. (2015), Hall et al. (2015), Li et al. (2016), Reichert et al. (2018)	The presence of microplastics in shellfish	Mussel (*Mytilus edulis*), isopod *(Idotea emarginata)*, fiddler crab (*Uca rapax*), and stony corals, (*Acropora, Pocillopora*, and *Porites)*	Ingestion and translocation
Lusher et al. (2013), Neves et al. (2015), Bellas et al. (2016), Nadal et al. (2016), Güven et al. (2017), Hermsen et al. (2017), Alomar et al. (2017), Collard et al. (2017), Jabeen et al. (2017), Ory et al. (2017), Akhbarizadeh et al. (2018), Ory et al. (2018)	The presence of microplastics in fish	Benthic, pelagic and zooplanktivorous fish species	Ingestion and translocation

Table 2 (*continued*)

Study	Focus of the Study	Sample Type	Governing Process
Fossi et al. (2012), Lusher et al. (2015a), Fossi et al. (2016)	The presence of microplastics in baleen whales	Seawater, planktonic samples, fin whales (*Balaenoptera physalus*), True's beaked whale (*Mesoplodon mirus)*	Ingestion
Setälä et al. (2014), Santana et al. (2017)	Transfer of microplastics along the food chain	Zooplankton, mussel (*Perna perna*), crab (*Callinectes ornatus*) and the puffer fish (*Spheoeroides greeleyi*)	Ingestion, biogenic transport and bioaccumulation
Lebreton et al. (2012), Cozar et al. (2014), Eriksen et al. (2014), Law et al. (2014)	Modeling distribution of floating macro- and microplastics on the surface of the open ocean	–	Transport
von Moos et al. (2012)	Effect of HDPE on blue mussel cells and tissue	Blue mussel (*Mytilus edulis* L.)	Ingestion
Oliveira et al. (2013)	Effect of PE on short-term toxicity of the PAH	Common goby (*Pomatoschistos microps*)	Sorption
Chua et al. (2014), Avio et al. (2015)	Transfer of PBDEs and PAHs to marine organisms by microplastics	Amphipods (*Allorchestes compressa*) and Mussel (*Mytilus galloprovincialis*)	Sorption and ingestion
Isobe et al. (2014)	Modeling distribution of microplastics in coastal waters	–	Transport
Cole and Galloway (2015), Cole et al. (2015), Gandara E Silva et al. (2016), Sussarellu et al. (2016), Tosetto et al. (2016), Jeong et al. (2017)	Toxic effects of microplastics to marine and beach organisms	Copepod (*Calanus helgolandicus, Paracyclopina nana*), mussel (*Perna perna*), oyster (*Crassostrea gigas*), beachhoppers (*Platorchestia smithi*)	Ingestion
Davarpanah and Guilhermino (2015)	Effect of virgin or Cu-loaded microplastics on marine microalgae	Microalgae (*Tetraselmis chuff*)	Sorption and ingestion
Enders et al. (2015)	The abundance and characteristics of microplastics and modeling vertical distribution	Subsurface water	Occurrence and transport
Mendoza and Jones (2015)	The presence of POPs on marine microplastics	Seawater column	Sorption
Reisser et al. (2015)	Vertical distribution of microplastics at sea	Surface and subsurface water	Transport
Van Cauwenberghe et al. (2015)	Effects of microplastics on energy metabolism of mussels and lugworms	Mussel (*Mytilus edulis*) and lugworm (*Arenicola marina*)	Ingestion and translocation
van Sebille et al. (2015)	Modeling spatial distribution of microplastics using global dataset of marine microplastics and ocean circulation models	Surface plankton samples	Transport
Bakir et al. (2016)	Modeling transfer of POPs to benthic invertebrate, a fish and a seabird via microplastics	–	Sorption and transport
Brennecke et al. (2016)	Interaction between microplastics (PS and PVC) and heavy metals (Cu and Zn)	Synthetic sample with seawater, microplastics and leached heavy metals	Sorption
Cole et al. (2016)	The effect of microplastics on fate of zooplankton faecal pellets	Zooplankton (*Calanus helgolandicus* and *Centropages typicus*)	Ingestion, egestion and sinking
Ferreira et al. (2016b)	The effects of gold nanoparticles (Au-NP), microplastics and temperature increase on common goby	Common goby (*Pomatoschistus microps*)	Ingestion
Fonte et al. (2016)	Influence of microplastics on toxicity of the antibiotic cephalexin to common goby	Common goby (*Pomatoschistus microps*)	Sorption and ingestion
Green et al. (2016)	The effects of microplastics on biological activity of lugworms and on nitrogen cycling and primary productivity of the sediment	Sediment and lugworm (*Arenicola marina*)	Ingestion
Green (2016)	The effects of microplastics on biological activity of European flat oysters and on the structure of associated macrofaunal assemblages	European flat oysters (*Ostrea edulis*), periwinkles (*Littorina* sp.), isopod (Idotea balthica) and shell clam (*Scrobicularia plana*)	Ingestion
Gutow et al. (2016)	Adsorption of microplastics to seaweeds and transfer of them to marine food web	Seaweed *(Fucus vesiculosus*) and periwinkle *(Littorina littorea*)	Sorption and ingestion
Paul-Pont et al. (2016)	Effect of microplastics alone or in combination with fluoranthene	Mussel (*Mytilus edulis*)	Sorption and ingestion
Pedà et al. (2016)	The intestinal responses of European sea bass exposed to microplastics	European sea bass (*Dicentrarchus labrax*)	Ingestion
ter Halle et al. (2016)	Fragmentation pattern of microplastics	Surface water	Photodegradation

(*continued on next page*)

Table 2 (*continued*)

Study	Focus of the Study	Sample Type	Governing Process
Akhbarizadeh et al. (2017)	Risk posed by microplastics and toxic elements, abundance and source identification	Coastal sediments	Relationship between microplastic quantities and heavy metal contamination
Avio et al. (2017)	The presence of microplastics in different benthic fish and macroinvertebrates	Fish and macroinvertebrates (*Ophiomusium lymani*, *Hymenaster pellucidus* and *Colus jeffreysianus*)	Ingestion
Bagaev et al. (2017)	Abundance, distribution and type of microplastics. Numerical modeling of transport.	Water column	Transport, sinking, re-suspension
Chubarenko and Stepanova (2017)	Migration of microplastics in sea coastal zone	Surface water	Transport
Debroas et al. (2017)	Microbial communities on the surface of the plastic compounds	Surface water	Biofouling
Devriese et al. (2017)	Effect of virgin or PCB-loaded microplastics on marine biota	Norway lobsters (*Nephrops norvegicus*)	Sorption, ingestion and bioaccumulation
Hinata et al. (2017)	Relationship between residence time of beached microplastics and their upward terminal velocities	Mark-recapture experiments on plastic floats	Backwash flux and onshore-offshore diffusivities
Iwasaki et al. (2017)	Numerical modeling transport of microplastics	–	Stokes drift and surface ocean currents
Jang et al. (2017)	The role of expanded polystyrene (EPS) on transport of hexabromocyclododecane (HBCD)	Beach sediment	Sorption and transport
Karlsson et al. (2017)	The abundance of microplastics in field collected samples and biota	Sediment, surface water, invertebrates, fish (*Salmo trutta*) and mussels (*Mytilus* edulis)	Occurrence, transport and ingestion
Katija et al. (2017)	The role of giant larvacean on transport of microplastics	Giant larvacean (*Bathochordaeus stygius*) (In-situ feeding experiments)	Ingestion, egestion and sinking with fecal pellets
Kooi et al. (2017)	Modeling vertical transport of microplastics by biofouling	–	Biofilm growth and sinking
Lee et al. (2017)	Fugacity analysis of PAHs between microplastics and seawater	PAH and PE	Sorption and partition
Martínez-Gómez et al. (2017), Ribeiro et al. (2017), Jovanović et al. (2018), Lo and Chan (2018)	Toxic effects of virgin microplastics (PS and HDPE)	Sea urchin (*Paracentrotus lividus*), clam (*Scrobicularia plana*), fish (Sparus *aurata*), and sea snail (*Crepidula onyx*)	Ingestion
Massos and Turner (2017)	The presence of heavy metals (Cd and Pb) and halogen (Bromine (Br)) in beached microplastics	Beach sediment	Sorption
Näkki et al. (2017)	Vertical transport of microplastics via bioturbation	Sediment and benthic invertebrates	Biogenic transport, ingestion and sinking
Paço et al. (2017)	Biodegradation of microplastics by marine fungus	PE and fungi (*Zalerion maritimum*)	Biodegradation
ter Halle et al. (2017)	Weathering of plastic debris	Collected PE from surface water	Degradation
Viršek et al. (2017)	Transport of the bacterial fish pathogen species *Aeromonas salmonicida* by microplastics	Surface water	Biofouling
Vroom et al. (2017)	Effect of aging microplastics on their ingestion by zooplankton	Pristine and aged PS, copepods (*Calanus finmarchicus* and *Acartia longiremis*)	Weathering and biofouling
Barboza et al. (2018)	Toxic effects of microplastics with mercury on fish	European seabass (*Dicentrarchus labrax*)	Sorption, ingestion and bioaccumulation
Kolandhasamy et al. (2018)	Uptake pathways of microplastics	Mussel	Adherence and ingestion
Zhang et al. (2018)	The presence of organophosphorus esters (OPEs) and phthalic acid esters (PAEs) in the beached microplastics	Beach sediment	Sorption

from wash-back by marine surface currents (Besseling et al., 2018). Rech et al. (2014) observed similar composition of anthropogenic litter between riversides and coastal beaches of each river near the estuarine areas. In addition, direct littering of plastic debris from coastal areas is a significant source of microplastic pollution. According to the scientific research, between 4.8 and 12.7 million tonnes of plastic enters the oceans every year from coastal regions (Jambeck et al., 2015), the highest concentrations of microplastics are often being detected in estuaries and coastal areas (Antunes et al., 2018). Numerous studies have investigated the abundance and distribution of microplastics in various estuarine environments including Portugal coasts (Antunes et al., 2013; Frias et al., 2014;

Lourenço et al., 2017), coastal beaches in Korea (Lee et al., 2013; Kim et al., 2015), the Goiania Estuary (Lima et al., 2015), the Paranaguá Estuary (Moreira et al., 2016) and the Jurujuba Cove (Castro et al., 2016) in Brazil, Tamar (Sadri and Thompson, 2014) and Solent Estuaries (Gallagher et al., 2016) in the UK, various estuaries in South Africa (Naidoo et al., 2015), German Baltic Coast (Stolte et al., 2015), Gulf of Mexico Estuary (Wessel et al., 2016), Khark Island in Iran (Akhbarizadeh et al., 2017), Atrato Delta in Colombia (Correa-Herrera et al., 2017), Dutch River Delta in Netherlands (Leslie et al., 2017), south-eastern coast of Australia (Ling et al., 2017) and various estuaries in China (Zhao et al., 2014; Fok and Cheung, 2015; Zhao et al., 2015; Cheung et al., 2016; Fok et al., 2017; Peng et al., 2017; Tsang et al., 2017; Cheung et al., 2018; Xu et al., 2018). These studies generally aim to demonstrate the presence of microplastics in estuarine surface waters (e.g. Sadri and Thompson, 2014; Tsang et al., 2017; Cheung et al., 2018) and in sediments (e.g. Lee et al., 2013; Kim et al., 2015; Moreira et al., 2016; Tang et al., 2018) and to investigate the most abundant type of microplastics found in a particular estuarine media by classifying these microparticles according to their sizes, shapes, colors or polymer types. According to our search results, majority of the relevant studies focus on microplastic pollution on coastal beach sediments. Kim et al. (2015) examined the most dominant microplastic type on high-tidal coastal beaches in Korea and defined the wind and currents as the driving forces influencing the spatial distribution of microplastics. A similar study carried out by Wessel et al. (2016) reported PP and PE as the most abundant polymer types in Gulf of Mexico Estuary, and revealed that the locations directly exposed to marine currents and tides have higher microplastic abundance and diversity. In addition to spatial variation, the studies that investigated the temporal variation of microplastic abundance in estuarine environments are also encountered (Lima et al., 2014; Castro et al., 2016; Cheung et al., 2016; Correa-Herrera et al., 2017; Tsang et al., 2017). Cheung et al. (2016) investigated the seasonal variation of microplastic abundance in 25 estuarine beaches in South China to understand the influence of wet and dry seasons on river inputs. Tsang et al. (2017) implemented a similar study for surface water and sediments to observe the spatio-temporal variation of microplastics in Hong Kong coastal regions. The studies reveal that the highest amount of plastic particles is observed during the wet seasons as a result of increased river flow, which accelerate the transport of these particles to the lower estuaries (Lima et al., 2014; Cheung et al., 2016).

3.2.2. Uptake by and effects in organisms

Microplastics are transferred from lower to higher levels in the food web. Presence of microplastics in variety of estuarine organisms were proved by several studies (e.g. Phillips and Bonner, 2015; Ferreira et al., 2016a; Pazos et al., 2017; Ferreira et al., 2018; Hu et al., 2018; Li et al., 2018). High concentrations of microplastics in coastal waters have led to the investigation of the occurrence of microplastics in zooplankton samples (Frias et al., 2014; Lima et al., 2014; Lima et al., 2015). The variety of studies have examined the seasonal variation of microplastics and ichthyoplankton and interactions between them in estuarine environments to understand the bioavailability of these microparticles to the marine biota (Lima et al., 2014; Lima et al., 2015; Correa-Herrera et al., 2017). Lima et al. (2014) also demonstrated that the amount of microplastics may surpass that of ichthyoplankton, which determine their bioavailability and distribution patterns within planktonic organisms. Santana et al. (2016) reported microplastic pollution in the Santos Estuary, Brazil by investigating the presence of microplastics in brown mussels (*Perna perna*). Ingestion of microplastics by the Chinese mitten crab (*Eriocheir sinensis*) from the Baltic coastal waters (Poland) and the Tagus Estuary (Portugal) were also studied based on stomach content analysis (Wójcik-Fudalewska et al., 2016). Phillips and Bonner (2015) examined 535 fish to analyze the occurrence and type of microplastics ingested by freshwater and estuarine fish in the Gulf of Mexico. According to the results, 8% of the freshwater fish and 10% of the marine fish had microplastic in their digestive tract. Ferreira et al. (2016a) detected microplastic particles in the digestive tract of weakfish (*Cynoscion acoupa*) collected from different habitats of the Goiana Estuary, Brazil and revealed that plastic particles were more frequently ingested than any other food item. Their results also indicate that microplastics were ingested in all areas and seasons. Lourenço et al. (2017) detected microfibers in large proportion of sediment samples (91%), macroinvertebrates (60%) and shorebird faeces (49%) from three important wetlands along the Eastern Atlantic. Their analysis results indicated only 52% of recorded microfibers were composed of synthetic polymers including polyacrylonitrile (PAN), polyethylene terephthalate (PET), PE, PP, PS, and nylon; while 28% were cellulose-based fibers and the remaining 20% were composed of proteins. The presence of microplastics were also investigated in gut contents of 11 fish species of the Rio de la Plata Estuary, microplastic concentrations were reported as significantly being higher closer to the sewage discharge (Pazos et al., 2017). Furthermore, a study by Watts et al. (2014) indicated that the shore crab (*Carcinus maenas*) can uptake microplastics through the inspiration across the gills. Microplastic particles and their adsorptive capacity may lead to growth delay, decrease in reproductivity, and mortality increase of estuarine organisms (de Sá et al., 2015). A study undertaken by Luís et al. (2015) revealed toxicological interactions between microplastics and Cr(VI) on goby fish. A similar study investigated the predatory performance of goby fish (*Pomatoschistus microps*) in the presence of microplastics combined with prey (*Artemia* nauplii) in two estuaries located in NW Iberian coast, USA under different environmental conditions (de Sá et al., 2015). The authors observed a significant reduction in predatory performance (65%) of fish from one estuary, while the fish from other estuary were not affected significantly, suggesting that developmental conditions may influence the prey selection capability of fish.

3.2.3. Fate and transport

Marine microplastics predominantly originate from coastal regions. They may remain nearshore for a long time or reach the oceans depending on the intensity and direction of surface currents and wind waves. However, there is little knowledge on the fate and transport of microplastics in coastal zones and estuaries. Yonkos et al. (2014) investigated the impact of urban development and storm events on microplastic quantities in four estuarine tributaries within the Chesapeake Bay, USA, and found that the greatest microplastic concentrations occurred shortly after major rainfall events. Liubartseva et al. (2016) developed the Markov chain model to simulate the macroplastic abundance at the sea surface and fluxes onto the coastline, and to identify source - receptor relationships among the subregions of the Adriatic Basin, between the years 2009–2015. According to the model outcomes, the coastline was the main sink of floating plastic debris. One other study implemented by Kaiser et al. (2017) examined how biofouling changes the sinking behavior of microplastics in estuarine and marine waters. After six weeks of incubation, the results revealed that despite the higher density of marine water, sinking velocity of PS particles incubated in ocean water increased more rapidly for particles sinking in the estuarine water. This was attributed to biofouling to be the major control over the sinking of microplastics.

Table 3
Microplastic research in estuaries.

Study	Focus of the Study	Sample Type	Governing Process
Rech et al. (2014), Moreira et al. (2016), Wessel et al. (2016)	Abundance, spatial distribution and composition of microplastics in the estuarine compartments: sources and sinks	Beach sediment	Occurrence and transport
Ling et al. (2017), Peng et al. (2017)		Estuarine sediment	Occurrence and transport
Sadri and Thompson (2014), Kang et al. (2015), Zhao et al. (2014), Zhao et al. (2015), Gallagher et al. (2016)		Surface water and/or subsurface water	Occurrence and transport
Leslie et al. (2017)		Sewage influent, effluent and sludge; surface water; sediment	Occurrence and transport
Naidoo et al. (2015), Tsang et al. (2017), Tang et al. (2018)		Surface water, sediment and/or beach sediment	Occurrence and transport
Hu et al. (2018)		Surface water, sediment and tadpoles	Occurrence and ingestion
Browne et al. (2010)	Effect of wind and depositional regime on distribution of microplastics	Water column	Transport
Antunes et al. (2013), Frias et al. (2010)	POPs in microplastics collected from beaches	Beach sediment	Sorption
Lee et al. (2013), Fok et al. (2017)	Abundance and size distribution of microplastics	Beach sediment	Occurrence and fragmentation
Frias et al. (2014), Kang et al. (2015)	The presence of microplastics in zooplankton	Plankton samples	Ingestion
Lima et al. (2014), Correa-Herrera et al. (2017)	Seasonal and spatial variation of microplastics relative to the zooplankton	Plankton samples and surface water	Surface runoff transport, ingestion
Holmes et al. (2014)	The role of plastic pellets on transport of trace metals	Virgin and beached pellets, river and sea water	Sorption
Yonkos et al. (2014)	Effect of land use and weather conditions on microplastic abundance	Surface water	Occurrence and transport
de Sá et al. (2015), Luís et al. (2015)	Effect of microplastics on the acute toxicity of Cr(VI) and predatory performance of common goby	Common goby (*Pomatoschistus microps*)	Sorption and ingestion
Kim et al. (2015)	Driving forces influencing the spatial variation of microplastics	Beach sediment	Occurrence and transport
Phillips and Bonner (2015), Pazos et al. (2017), Vendel et al. (2017), Ferreira et al. (2018), Pegado et al. (2018)	The presence of microplastics in fish	Freshwater and estuarine fish	Ingestion
Stolte et al. (2015), Cheung et al. (2016)	Abundance and seasonal variation of microplastics	Beach sediment	Occurrence, surface runoff and transport
Castro et al. (2016), Cheung et al. (2018)		Surface water	
Ferreira et al. (2016a)	Seasonal and spatial variation of microplastics in Acoupa weakfish	Fish	Ingestion
Liubartseva et al. (2016)	Modeling transport of floating plastic debris	–	Transport by ocean currents and winds
Watts et al. (2014), Santana et al. (2016), Wójcik-Fudalewska et al. (2016), Li et al. (2018), Waite et al. (2018)	The presence of microplastics in shellfish	Mussel *(Perna perna)*, Crabs (*Carcinus maenas, Eriocheir sinensis, Panopeus herbstii*) and oysters (*Saccostrea cucullata, Crassostrea virginica*)	Ingestion
Akhbarizadeh et al. (2017)	Ecological risk assessment of microplastics, heavy metals and PAHs	Beach sediment	Sorption
Kaiser et al. (2017)	Sinking behavior of microplastics in estuarine and coastal waters	Surface water, lab-based samples	Biofouling
Lourenço et al. (2017)	Transfer of microplastics along the food chain	Sediment, macroinvertebrates, shorebird faeces	Ingestion and bioaccumulation
Xu et al. (2018)	Risk assessment of microplastics in the Changjiang Estuary, China	Surface water	Occurrence and transport

3.2.4. *Existing research*

Microplastic studies in the estuarine environment are given in Table 3.

3.3. *Rivers*

Rivers act as a major transport pathway for plastic debris. It has been estimated that between 70 and 80% of marine plastics are transported to the seas through the conduits provided by rivers (Bowmer and Kershaw, 2010). Moore et al. (2011) revealed that total weight of microplastic particles that were released from the two urban rivers, Los Angeles and San Gabriel, into the marine environment was 30 metric tons over a three-day period. However, research in rivers has been relatively limited, only started gaining attention around 2014. In this review, a total of 34 papers, published between the years 2014 and 2018, related to microplastic research in rivers are identified in Web of Science Core Collection database. While 20 articles of total are evaluated under the subcategory occurrence and characterization, only a few studies are revised under uptake by and effects in organisms (nine articles), and fate and transport (five articles).

3.3.1. *Occurrence and characterization*

Considering the origin and risk posed by microplastics, the studies investigating the occurrence and abundance in freshwaters have increased, many of which have focused on rivers (e.g. Mani et al., 2015; Baldwin et al., 2016; Kapp and Yeatman, 2018; Nel et al., 2018). The level of microplastic pollution in riverine waters was first reported in California, in 2011 (Moore et al., 2011). In recent years, microplastics were also found in Rhine River, Germany (Klein et al., 2015; Mani et al., 2015); 29 Great Lakes tributaries, US (Baldwin et al., 2016); Ombrone River, Italy (Guerranti et al., 2017); Thames River, UK (Horton et al., 2017a); Columbia Rivers (Kapp and Yeatman, 2018) and Shanghai Rivers, China (Peng et al., 2018). In addition, Lechner et al. (2014) estimated meso and microplastic (5–20 mm) discharge via Danube River, Austria into the Black Sea as 4.2 tons per day using stationary driftnets over a two-year period (2010–2012). Gasperi et al. (2014) examined the quantity and type of floating macroplastics flowing down the River Seine within the Paris metropolitan area. The authors revealed that between 22 and 36 tons of floating macroplastic debris were intercepted annually, and most of them were made of PP, PE and, to a lesser extent, PET. The influence of municipal and industrial effluents on microplastic concentration in surface water and sediment have been investigated for a variety of rivers, such as St. Lawrence (Castaneda et al., 2014) and Ottawa, Canada (Vermaire et al., 2017), Seine in Paris (Dris et al., 2015), the Rhine-Main area in Germany (Klein et al., 2015), Raritan in New Jersey (Estahbanati and Fahrenfeld, 2016), and Saigon in Vietnam (Lahens et al., 2018). Dris et al. (2015) carried out the first study in the Seine River to investigate microplastic pollution in urban compartments including wastewater, atmospheric fallout and rivers. Vermaire et al. (2017) examined microplastic abundance in surface waters and sediments of the Ottawa River, Canada, and revealed that microplastic concentrations were significantly higher downstream of the wastewater treatment plant compared with the upstream of the effluent output. In China, the effect of anthropogenic factors on the abundance of microplastics in surface water of 20 urban lakes and urban reaches of the Hanjiang River and Yangtze River of Wuhan were studied (Wang et al., 2017b). The authors found fibers as the most dominant microplastic shape and they identified PET and PP as the dominant polymer types of microplastics analyzed. Similar studies were also carried out for Yangtze (Zhang et al., 2015) and Xiangxi Rivers (Zhang et al., 2017a,b), two typical tributaries of the Three Gorges Reservoir in China. Horton et al. (2017a) investigated the abundance and sources of microplastics in sediments of the Thames River. The authors found significantly higher amount of plastic particles at downstream of a storm drain outfall receiving urban runoff that contains fragments derived from road marking paints. Nel et al. (2018) assessed the microplastic pollution dynamics in an urban river sediment experiencing temporal differences in river flow and found that microplastic amount in winter was approximately 25 times higher than in summer, likely due to the increased sedimentation associated with reduced river flows during the winter season.

Microplastic in sediment samples is usually difficult to detect, due to the nature and dark color of the sediment, and high percentage of colored organic matter it contains. For this reason, the potential use of deposit feeders as an indicator of microplastic pollution has gained attention in recent years. To date, due to being intermediary trophic link of the food web and their relative abundance in freshwaters, some studies have focused on deposit feeders, such as chironomid larvae (Nel et al., 2018) and Asian clam (Su et al., 2018). These studies indicated a relationship between microplastic abundance in sediment and deposit feeders suggesting chironomid larvae and Asian clam could serve as indicators of microplastic pollution in freshwater sediments (Nel et al., 2018; Su et al., 2018).

3.3.2. Uptake by and effects in organisms

Ingestion of microplastics by freshwater fauna has only been studied for invertebrates and fish (Sanchez et al., 2014; Peters and Bratton, 2016; Ma et al., 2016; Campbell et al., 2017; Hurley et al., 2017; McGoran et al., 2017; Silva-Cavalcanti et al., 2017; Horton et al., 2018; McNeish et al., 2018). Microplastics were detected in the digestive tracts of bluegill (*Lepomis macrochirus*) and longear (*Lepomis megalotis*) sunfish from the Brazos River in USA (Peters and Bratton, 2016), gudgeons (*Gobio gobio*) from French rivers (Sanchez et al., 2014), European flounder (*Platichthys flesus*), European smelt (*Osmerus eperlanus*) and roach (*Rutilus rutilus*) from the Thames River, UK (McGoran et al., 2017; Horton et al., 2018) and five different fish species including northern pike (*Esox lucius*), white sucker (*Catostomus commersoni*), emerald shiner (*Notropis atherinoides*), fathead minnow (*Pimephales promelas*), and fivespine stickleback (*Eucalia inconstans*) from an urban creek immediately downstream of Regina, Canada (Campbell et al., 2017). Zhang et al. (2017a,b) found PE and nylon in the digestion tracts of 25.7% of fish samples collected from the Xiangxi River. Hurley et al. (2017) investigated the ingestion of microplastics by Tubifex worms from bottom sediments in a major urban waterbody fed by the Irwell River, Manchester, UK and estimated the mean concentration of ingested microplastics as 129 ± 65.4 particles per gram tissue. Silva-Cavalcanti et al. (2017) detected microplastics in guts of *Hoplosternum littorale*, a freshwater fish heavily consumed in semi-arid regions of South America, fibers being the most frequent form (46.6%).

3.3.3. Fate and transport

Although microplastic research in freshwaters has gained increased attention over the last years, there is still insufficient knowledge on the fate and transport mechanisms of microplastics in rivers. Similar to marine environment, density and particle size affect retention of microplastics in river sediments, but they are not solely the dominant factors determining the distribution of these micro particles in rivers (Nizzetto et al., 2016a; Besseling et al., 2017). Nizzetto et al. (2016a) upgraded a mathematical model of catchment hydrology, soil erosion and sediment budgets for theoretical assessment of microplastic transport in river catchments. The authors revealed that microplastics that have densities higher than water could be retained in the sediment; however, high flow periods could remobilize this pool, meaning sediments in low flow river segments are likely hotspots for deposition of microplastics. Besseling et al. (2017) implemented scenario studies on the fate and transport of nano- and microplastics in rivers by constructing a model that accounts for advective transport, homo and heteroaggregation, sedimentation-resuspension, polymer degradation, presence of biofilm and burial. According to the model outcomes, particle size affects retention and positioning of the accumulation hotspots in the sediment along the river, yet aggregation and further sedimentation act as major processes controlling the fate and retention of microplastics along the river, due to excess mass of suspended solids that form heteroaggregates with microplastics, which overwhelm the modelled effects of polymer density and biofilm formation.

A global model of macro and microplastic discharge from rivers into oceans based on waste management, population density and hydrological information was developed by Lebreton et al. (2017). The authors estimated that between 1.15 and 2.41 million tonnes of plastic waste enters the ocean every year from rivers, over 74% of emissions occurring between May and October. The study also reveal that the top 20 polluting rivers are mostly located in Asia and accounted for 67% of the global annual discharge. One other modeling approach was developed using an existing global model for nutrients, the Global NEWS (Nutrient Export from WaterSheds) to analyze the composition and quantity of point-source microplastic fluxes from European rivers to the seas and to explore future trends up to the year 2050 (Siegfried et al., 2017). The model results indicated that about two-thirds of the microplastics modelled flow into the Mediterranean and Black Sea, and most of these particles were synthetic polymers from TRWP (42%) and plastic-based textiles abraded during laundry (29%).

Research indicates that there is a need for the investigation of microplastic transport processes between terrestrial and freshwater systems. A recent study investigated the transport of TRWP in the Seine Watershed (France) by developing a comprehensive watershed-scale modeling methodology. The authors studied both terrestrial transport to soil, air, and roadways, as well as freshwater transport processes including heteroaggregation, degradation and sedimentation. The model estimated that the per capita mass release of TRWP in the Seine Watershed was 1.8 kg per inhabitant in a year, 18% of which were transported to the freshwater, whereas 2% were exported to the estuary (Unice et al., 2019a). The model also confirmed the significance of particle size and density on estuarial transport of TRWP (Unice et al., 2019b).

3.3.4. Existing research

Microplastic research in rivers from past to present are summarized in Table 4.

3.4. Lakes

Microplastic research in lakes has gained attention since 2013. 30 articles are found in Web of Science Core Collection database. 20 of articles are evaluated under the subcategory occurrence and characterization, while 11 and two of these are examined under uptake by and effects in organisms and fate and transport, respectively.

3.4.1. Occurrence and characterization

Occurrence and characterization of microplastics in lakes have been studied over the last few years (e.g. Eriksen et al., 2013; Ballent et al., 2016; Su et al., 2016; Wang et al., 2017b). Most of

Table 4
Microplastic research in rivers.

Study	Focus of the Study	Sample Type	Governing Process
Castaneda et al. (2014)	Abundance and size distribution of microplastics	Sediment	Occurrence and transport
Lechner et al. (2014)	Abundance and mass of microplastics: microplastic input from river into sea	Surface water	Occurrence and transport
Gasperi et al. (2014), Mani et al. (2015), Zhang et al. (2015), Wang et al. (2017b), Kapp and Yeatman (2018), Lahens et al. (2018)	Abundance, spatial distribution and composition of microplastics in the riverine compartments: sources and sinks	Surface water	Occurrence and transport
Klein et al. (2015), Ballent et al. (2016), Guerranti et al. (2017), Horton et al. (2017a,b), Peng et al. (2018)		Sediment	Occurrence and transport
Wang et al. (2017a)		Surface sediment	Occurrence, transport and sorption
Zhang et al. (2017a,b)		Surface water, sediment and fish	Occurrence, transport and ingestion
Sanchez et al. (2014), Peters and Bratton (2016), McGoran et al. (2017), Silva-Cavalcanti et al. (2017), Horton et al. (2018), McNeish et al. (2018)	The presence of microplastics in freshwater fish	Fish samples (*Gobio gobio*, *Lepomis macrochirus, Lepomis megalotis*, *Platichthys flesus*, *Osmerus eperlanus*, *Hoplosternum littorale*, *Rutilus rutilus)*	Ingestion
Dris et al. (2015)	The abundance, type and size distribution of microplastics	Wastewater, surface water and atmospheric fallout	Occurrence and transport
Baldwin et al. (2016)	Impact of land use, wastewater effluents and hydrologic conditions on microplastic abundance and characteristics	Surface water	Occurrence and transport
Estahbanati and Fahrenfeld (2016)	Impact of wastewater effluents on microplastic abundance and type	Surface water	Occurrence and transport
Ma et al. (2016)	Toxicity of microplastics with Phe	*Daphnia magna*	Ingestion, sorption and bioaccumulation
Nizzetto et al. (2016a), Besseling et al. (2017)	Modeling fate and transport of microplastics in rivers	–	Surface runoff, advective transport, sinking-resuspension, aggregation and degradation
Campbell et al. (2017)	Impact of wastewater effluents on microplastic abundance and type in river and fish samples	Surface water, fish	Occurrence and ingestion
Hurley et al. (2017)	The presence of microplastics in Tubifex worms from bottom sediment	Sediment *Tubifex tubifex*	Ingestion
Lebreton et al. (2017), Siegfried et al. (2017)	Sources of microplastics and global modeling of inputs from rivers into seas	Field measurements data	Waste management, population density, hydrology and transport
Schmidt et al. (2017)	Sources of microplastics and inputs from rivers into seas	River water column data	Transport
Nel et al. (2018)	Seasonal variation of microplastics and potential use of chironomids as indicators of microplastic pollution	Sediment, *Chironomus* spp	Occurrence, transport and ingestion
Unice et al. (2019a), Unice et al. (2019b)	Modeling fate and transport of (TRWP) in watershed	–	Terrestrial and freshwater transport

this research have focused on US Great Lakes. Microplastic pollution in Great Lakes of North America has been reported for surface waters (Eriksen et al., 2013; Mason et al., 2016; Hendrickson et al., 2018), along shorelines (Zbyszewski et al., 2014; Corcoran et al., 2015) and in offshore lake bottom sediments (Corcoran et al., 2015). Ballent et al. (2016) investigated the abundance and depositional patterns of microplastics in nearshore, tributary and beach sediments along the Canadian shoreline of Lake Ontario. Anderson et al. (2017) reported microplastic pollution in Lake Winnipeg, and formally compared the results to reported concentrations in other large North American Lakes (Eriksen et al., 2013). Faure et al. (2015a) assessed plastic abundance in Swiss lakes, identified the composition of these particles and examined the hydrophobic micropollutants adsorbed onto the microplastics. They finally assessed the potential exposure of fish and water birds to both microplastics and adsorbed micropollutants. Biginagwa et al. (2016) reported the presence of microplastics in the African Great Lakes. Another study was carried out in Italy by Fischer et al. (2016) assessing the abundance and spatial distribution of microplastics within surface waters and shoreline sediments of the lakes Bolsena and Chiusi. The authors also investigated the effects of heavy winds on particle distribution and revealed that an event of heavy winds and moderate rainfall prior to the sampling led to an increase in concentrations, which is most probable related to lateral land-based and sewage effluent inputs. Microplastic pollution levels in surface water and sediments in Taihu Lake (Su et al., 2016), Three Gorges Reservoir (Di and Wang, 2018) and Qinghai Lake, China (Xiong et al., 2018) were reported. Similar studies were carried out for microplastics in the sediments of Tibetan Plateau lakes, China (Zhang et al., 2016), Vembanad Lake, India (Sruthy and Ramasamy, 2017) and Edgbaston Pool, Birmingham, UK (Vaughan et al., 2017) as well as surface water of Lake Hovsgol, Mongolia (Free et al., 2014), and 20 urban lakes of Wuhan, the largest city in central China (Wang et al., 2017b).

3.4.2. Uptake by and effects in organisms

Several studies in the literature investigates the ingestion of microplastics by freshwater invertebrates (Ma et al., 2016; Rehse et al., 2016; Su et al., 2016; Scherer et al., 2017; Redondo-Hasselerharm et al., 2018), and fish (Batel et al., 2016; Biginagwa et al., 2016; Karami et al., 2016; Wen et al., 2018). Su et al. (2016) analyzed the abundance of microplastics in plankton net samples and in Asian clams (*Corbicula fluminea*) in Taihu Lake, China. Biginagwa et al. (2016) reported the presence of microplastics in the gastrointestinal tracts of local fish species (*Lates niloticus* and *Oreochromis niloticus*) of the African Great Lakes. Rehse et al. (2016) analyzed the adverse effects of microplastics on limnic zooplankton (*Daphnia magna*) on laboratory conditions for 96 h and revealed that ingestion of 1 μm particles led to immobilisation increasing with dose and time, while 100 μm particles that could not be ingested by *Daphnia magna* did not cause any observable physical harm. In addition, a study by Scherer et al. (2017) demonstrated that the quantity of microplastic uptake by invertebrates, such as *Daphnia magna*, *Chironomus riparius* and *Gammarus pulex* depends on their feeding type and morphology as well as on the availability of microplastics. Exposure experiments were pursued for freshwater amphipod (*Gammarus fossarum*), which resulted in a significant decrease of the assimilation efficiency (Blarer and Burkhardt-Holm, 2016; Straub et al., 2017).

Toxic effects of microplastics interacted with hydrophobic pollutants have also been subject of research for freshwater fauna. Ma et al. (2016) studied the interaction of phenanthrene (Phe), with different size of microplastics (from 50 nm to 10 μm) and their associated toxicity to *Daphnia magna*. According to the results, during a 14-day incubation, 50 nm nanoplastics showed significant toxicity and physical damage to *Daphnia magna*, while 10 μm microplastics did not show significant effects on the bioaccumulation, dissipation, and transformation of Phe. Another study examined the exposure of juvenile African catfish (*Clarias gariepinus*) to virgin microplastics alone or in combination with Phe for 96 h (Karami et al., 2016). At the end of the experiment, substantial histopathological changes were observed in the gill and liver, also virgin or Phe-loaded microplastics altered blood biochemical parameters and the transcription of some of the reproductive-axis genes of the fish. A study carried by Batel et al. (2016) investigated the transfer of microplastic particles and associated benzo[*a*]pyrene (BaP) along a simple artificial food chain between brine shrimp (*Artemia* nauplii) and zebrafish (*Danio rerio*). According to their research, small (1–20 μm) polymer particles accumulated in brine shrimp, and were subsequently transferred to zebrafish. However, virgin microplastic particles alone had no observable effect on the intestinal tracts of fish. Similar results were also obtained from a study by Chen et al. (2017), which revealed microplastics, other than nanoplastics alone exhibited no significant effect on zebrafish mobility, yet when co-exposed with a certain concentration of 17 α-ethynylestradiol (EE2) (20 μg/L), both plastics lead to hypoactivity. Rochman et al. (2017) developed a model to investigate the differences in bioaccumulation of polychlorinated biphenyls (PCBs) among polymers including PET, PE, polyvinylchloride (PVC) or PS. The authors also examined the associated influence of different types of microplastics with PCBs on Asian clams (*Corbicula fluminea*). According to the model results, the concentrations of PCBs in clams exposed to PE and PS would be greater than PET and PVC. Additionally, at the end of the 28-day exposure, the authors observed the greatest histological abnormalities in clams fed with PCBs/PVC or PCBs/PS.

3.4.3. Fate and transport

Only two studies investigating fate and transport of microplastics in lakes were identified through the literature search. Hankett et al. (2016) developed a model system to investigate the deposition/desorption behaviors of endocrine disrupting toxins on microplastics for different environments, such as land and freshwater surface. According to their model outcomes, nonylphenols (NPs) deposits differently on both PS and PET under humid and dry environments, which is attributed to surface energy changes from increased water content that provides a favourable environment for NP deposition. Hoffman and Hittinger (2017) estimated coastal plastic input into the Great Lakes and then used currents from a numerical hydrodynamic model to calculate microplastic inventory and transport, as well as to map spatial distribution of plastic debris throughout the Great Lakes.

3.4.4. Existing research

Microplastic research in lakes is summarized in Table 5.

3.5. Soil

Most of the plastic litter arrives to the oceans or disposed on the land. Geyer et al. (2017) estimated that approximately 6300 million tons of plastic waste were generated between the years 1950 and 2015, 4977 million tons of which accumulated in landfills and the natural environment. In addition, it is estimated that between 125 and 850 tons of microplastics per million inhabitants are added annually to European agricultural soils through the application of sewage sludge (Nizzetto et al., 2016b). Light macro and microplastics may be transported across land by wind, whereas denser plastic items are more likely to be buried in deeper layers and remain in the soil (Horton et al., 2017b). There is evidence that low temperatures, low oxygen levels and coverage with water,

Table 5
Microplastic research in lakes.

Study	Focus of the Study	Sample Type	Governing Process
Eriksen et al. (2013), Free et al. (2014), Mason et al. (2016), Anderson et al. (2017), Wang et al. (2017b), Hendrickson et al. (2018)	Abundance and composition of microplastics in lakes	Surface water	Occurrence and transport
Di and Wang (2018)		Surface water and sediment	Occurrence and transport
Sruthy and Ramasamy (2017), Vaughan et al. (2017)		Sediment	Occurrence and transport
Zhang et al. (2016)		Sediment	Occurrence, transport and weathering
Su et al. (2016)		Surface water, sediment and Asian clam (*Corbicula fluminea*)	Occurrence, transport and ingestion
Zbyszewski et al. (2014), Corcoran et al. (2015), Sighicelli et al. (2018)	Abundance, spatial distribution and composition of microplastics in lakes: sources and sinks	Surface water	Occurrence and transport
Fischer et al. (2016)		Surface water and sediment	
Xiong et al. (2018)		Surface water, sediment and fish	
Ballent et al. (2016)		Sediment	Occurrence, transport and sinking-resuspension
Faure et al. (2015a)	The presence of microplastics in Lakes Geneva and potential ingestion by fish and birds	Surface water and beach sediment	Sorption and ingestion
Batel et al. (2016)	Transfer of microplastics and associated POPs along the food chain	*Artemia* nauplii and zebrafish (*Danio rerio*)	Sorption, ingestion and bioaccumulation
Rochman et al. (2017)		Asian clam (*Corbicula fluminea*) and white sturgeon (*Acipenser transmontanus*)	
Biginagwa et al. (2016)	The presence of microplastics in freshwater fish	Fish samples (*Lates niloticus, Oreochromis niloticus*)	Ingestion
Blarer and Burkhardt-Holm (2016), Scherer et al. (2017), Straub et al. (2017)	Uptake of microplastics by freshwater invertebrates	Invertebrate samples (*Gammarus fossarum, Daphnia magna, Chironomus riparius, Physella acuta, Gammarus pulex, Lumbriculus variegatus*)	Ingestion
Hankett et al. (2016)	Effect of humidity on endocrine disrupting toxin deposition on microplastics	NP, PS and PET	Sorption and surface energy
Imhof et al. (2016)	Abundance and size distribution of very small microplastics (1−50 μm) and associated heavy metals	Beach sediment	Occurrence and sorption
Karami et al. (2016), Ma et al. (2016), Chen et al. (2017)	The effect of virgin or hydrophobic pollutant-loaded microplastics on limnic organisms	Fish samples (*Clarias gariepinus, Danio rerio*), zooplankton (*Daphnia magna*)	Sorption and ingestion
Rehse et al. (2016), Redondo-Hasselerharm et al. (2018)	Toxic effects of microplastics on limnic organisms	Zooplankton (*Daphnia magna*), invertebrates (*Gammarus pulex, Hyalella azteca, Asellus aquaticus, Sphaerium corneum, Tubifex* spp., *Lumbriculus variegatus*)	Ingestion
Hoffman and Hittinger (2017)	Modeling spatial distribution of microplastics in the Great Lakes	Surface water (literature data)	Occurrence and transport
Wen et al. (2018)	Effect of microplastics under elevated temperatures on fish	Discus fish (*Symphysodon aequifasciatus*)	Ingestion

sediment, or soil reduce exposure to UV radiation, hampering plastic fragmentation. Hence, photo-oxidative fragmentation is extremely slow for macro and microplastics buried in the soil; thus, soils have been considered as a sink of microplastics (Duis and Coors, 2016). Although microplastics have potential to alter the geochemistry of soils and interact with soil biota (Machado et al., 2018), their probable impacts on terrestrial environment remain poorly understood, yet has gained attention, particularly since 2016. Most of these studies have generally focused on the ingestion of microplastics by soil biota, such as earthworms and soil collembolans (Huerta Lwanga et al., 2016; Hodson et al., 2017) and role of these terrestrial organisms on transport of microplastics in soil (Huerta Lwanga et al., 2017a; Maaβ et al., 2017; Rillig et al., 2017; Zhu et al., 2018). Huerta Lwanga et al. (2016) investigated the survival and fitness of the earthworm *Lumbricus terrestris* exposed to different concentrations of PE, whereas Hodson et al. (2017) studied the interactions between PS particles and zinc (Zn) to understand the effect of microplastics on metal bioavailability to earthworms. Liu et al. (2017) examined the effect of microplastics on soil dissolved organic matter (DOM) by analyzing the soil contents of dissolved organic carbon (DOC), dissolved organic nitrogen (DON), ammonium (NH_4^+), dissolved organic phosphorus (DOP), and phosphate (PO_4^{3-}). The authors also investigated the change in Phenol Oxidase (PO) and Fluorescein Diacetate hydrolase (FDAse) enzyme activities of bacterial communities. According to their outcomes, microplastic addition facilitated the accumulation of high-molecular weight humic-like material and stimulated enzymatic activity. Fate and transport of microplastics in soil have been studied investigating ingestion of these particles by earthworms (Huerta Lwanga et al., 2017a; Rillig et al., 2017), collembolan species (Maaβ et al., 2017) and mites (Zhu et al., 2018). In addition to occurrence and biogenic transport, researchers have also investigated microbial decay of microplastics in soil as a potential soil restoration method (Huerta Lwanga et al., 2018). Previous studies revealed that PE particles can be biodegraded by bacteria isolated from the gut of the larvae of meal moth (*Plodia interpunctella*) (Yang et al., 2014) and PE size can decrease as it passes through the gut of earthworm *Lumbricus terrestris* (Huerta Lwanga et al., 2016). Huerta Lwanga et al. (2018) examined the effect of bacteria extracted from the gut of *Lumbricus terrestris* on low-density polyethylene (LDPE)

Table 6
Microplastic research in soil.

Study	Focus of the Study	Sample Type	Governing Processes
Huerta Lwanga et al. (2016)	Exposure of the earthworm *Lumbricus terrestris* to microplastics	*Lumbricus terrestris*	Ingestion
Nizzetto et al. (2016a)	Microplastic transport in river basins	–	Surface runoff, soil erosion and sediment transport
Hodson et al. (2017)	Metal exposure in earthworms via microplastics	Synthetic soil mixture, *Lumbricus terrestris*	Sorption and ingestion
Liu et al. (2017)	Effects of microplastics on soil DOM and enzyme activities of bacterial communities	Synthetic soil mixture, lab-based FDAse and PO	Sorption
Huerta Lwanga et al. (2017a)	Microplastic transport in earthworm *Lumbricus terrestris* burrows and risk assessment	Synthetic soil mixture, *Lumbricus terrestris*	Biogenic transport
Huerta Lwanga et al. (2017b)	Transfer of micro- and macroplastics from soil to chickens	Soil, earthworm casts chicken faeces and gizzards	Ingestion and biomagnification
Maaβ et al. (2017)	Microplastic transport in soil by collembolan species	Synthetic soil mixture, *Folsomia candida* and *Proisotoma minuta*	Ingestion and biogenic transport
Rillig et al. (2017)	Microplastic transport in soil by earthworm *Lumbricus terrestris*	Synthetic soil mixture and *Lumbricus terrestris*	Ingestion and biogenic transport
Huerta Lwanga et al. (2018)	Low-density polyethylene (LDPE) decay by earthworm gut bacteria	Synthetic soil mixture, *Lumbricus terrestris*	Biodegradation
Zhang and Liu (2018)	Abundance and distribution of microplastics in soil aggregates	Soil	Occurrence
Zhu et al. (2018)	Predator (mite)-prey (collembolan) relationship in the transport of microplastics in soil	*Hypoaspis aculeifer* (mite), *Folsomia candida* (collembolan)	Ingestion and biomagnification

decay. The authors observed that LDPE particle size was significantly reduced, and 60% of microplastics content was decayed with isolated bacteria.

Microplastic research in the terrestrial environment has generally been investigated through laboratory experiments by adding microplastic particles and soil organisms to the soil samples. Yet, field-based experiments are scarce. A study undertaken by Huerta Lwanga et al. (2017b) assessed macro and microplastics in soil, earthworm casts, chicken faeces, crops and gizzards collected from the traditional Mayan home gardens of Southeast Mexico to indicate the biomagnification of microplastics through food chain. Another study revealed the abundance and distribution of microplastics in soil aggregates collected from Dian Lake Basin, China (Zhang and Liu, 2018). The studies related to microplastics in soil are given in Table 6.

3.6. *Atmosphere*

Atmospheric deposition as a source of microplastics remains poorly investigated. There is evidence that microplastic particles, in the form of fibers, can become airborne. Microplastics have been measured in atmospheric fallout in Paris (Dris et al., 2016) and in China (Cai et al., 2017), with greater concentrations in urban areas. Dris et al. (2016) estimated that between three to ten tons of synthetic fibers were deposited on Paris agglomeration (area around 2500 km^2) by atmospheric fallout every year. Thus, there is potential for atmospheric deposition to be a source of microplastics to freshwater systems, and to marine environment. In addition, microplastic exposure via inhalation could occur, yet the adverse effects of which to human health is unknown. The inhaled or ingested microplastics may accumulate and exert localized particle toxicity by inducing or enhancing an immune response (Wright and Kelly, 2017).

In addition to outdoor air, the presence of microplastics in indoor air was also proved (Dris et al., 2017). In laboratory conditions, natural or synthetic fibers present in indoor air generally lead to background contamination of environmental samples with microplastics. For this reason, a new methodology was developed by Torre et al. (2016) and Wesch et al. (2017) to minimize airborne microplastic contamination to increase scientific results accuracy. Dris et al. (2017) observed that fibers in indoor air contained, 33% PP, and revealed that such fibers were not likely to be inhaled because of their size but the exposure may occur through dust ingestion, particularly for young children.

4. Discussion

This paper highlights the current state of knowledge and research gaps about microplastic research in different environmental compartments. Microplastics in the marine environment has long been a developing research field, exhibiting a rapid increase in the number of publications, especially over the last decade. This increasing trend can be observed, particularly between the years 2015 and 2017 (Fig. 2b). Microplastic studies in the marine environment have generally focused on the occurrence of these pollutants, and their ingestion by biota, as well as adsorption of different environmental contaminants, such as POPs and heavy metals onto these microparticles. Microplastic uptake by marine biota has been examined for different organisms, especially in stomachs of various fish species and invertebrates, particularly collected in the field. However, the field of studies related with highest trophic level, such as marine mammals are relatively narrow.

Research involving fate and transport mechanisms of microplastics in the marine environments are still limited. There are several studies modeling transport of floating macro and microplastic debris in the oceans; nevertheless, the publications involving vertical distribution of microplastics in the water column are scarce. Fate and transport studies in low depth coastal sediments and shallow waters are also relatively few. Owing to oceanic currents and seasonal variations, high spatial and temporal variability of microplastics in the marine environment hamper assessing the density, abundance and distribution of these particles (Doyle et al., 2011; Ryan et al., 2009; Cole et al., 2011). In this respect, more advanced studies adopting holistic approaches are required to understand the driving forces affecting both horizontal and vertical distribution of microplastics in the marine environment.

Microplastic research in other environments such as freshwater, soil and air have been poorly understood. Although, rivers are the major transport routes transferring plastics to the oceans (Wagner and Lambert, 2018), in this review, only about 11% of the studies had a riverine context. Carrying out research especially in lower river reachers would provide useful insights towards understanding the mechanisms effective in microplastic transport and accumulation in the seas. In recent years, there is a rapid growth in knowledge of abundance of microplastics in freshwater systems. However, still limited research linking microplastic concentrations and their associated ecological impacts hampers assessing exact environmental risk of microplastics in these systems. Although, terrestrial environment is a considerable source of microplastics with a ~80% contribution to the coastal and marine environments (Andrady, 2011), studies involving the detection and fate of microplastics in soil and in terrestrial ecosystems are still scarce. The data for occurrence of microplastics in the terrestrial environment is almost not available, while uptake of microplastics by terrestrial organisms has so far only been investigated in few studies. According to Rillig (2012), this can be attributed to the relative ease, which microplastic filaments can be extracted and quantified from the water. However, this has started to change in the last two years, and attention is now also directed to the terrestrial environments. However, in most cases, addressing microplastic sources on a particular location is challenging due to variability of potential origins and geographical distribution of different sources. Particularly, identifying nonpoint sources of microplastics is more difficult as these sources are more diffusive and diverse (Fahrenfeld et al., 2019). Microplastics could be transported to surface waters via runoff; nevertheless, nearly no data are available on microplastic pollution in surface runoff. Despite the fact that recent studies suggest that agriculture is a significant source of microplastic pollution due to the use of plastic mulch, and the application of sewage sludge as soil amendment (Nizzetto et al., 2016b; Rodríguez-Seijo and Pereira, 2019); to the best of our knowledge, the impact of agricultural runoff on transport and load of microplastics to receiving waters have not been studied yet. Furthermore, the contribution of high population density and urban areas on microplastic pollution have been demonsrated in many studies (Jambeck et al., 2015; Lebreton et al., 2017; Wang et al., 2017b), yet there are few data on microplastic load in stormwater runoff (Fahrenfeld et al., 2019). Additionally, though previous research reported atmospheric deposition as a potential source of microplastics (Dris et al., 2016; Cai et al., 2017) little attention has been paid on the atmospheric environment. Lack of comprehensive assessment of microplastic sources and fluxes in urban areas makes difficult to understand the contribution of atmospheric fallout to microplastic pollution in the aquatic compartments, and to evaluate the climatic forces as a pathway of microplastic transport (Dris et al., 2016). In this regard, further studies should investigate the contribution of dry and wet atmospheric fallout to the microplastic load in marine and freshwater systems, by studying the depositional fluxes at the air-water interface.

It is generally assumed that majority of microplastics in the environment are derived from secondary sources via fragmentation of macroplastics. However, little information is encountered in literature on fragmentation rates of macroplastics (Duis and Coors, 2016). Although there is evidence that fragmentation of plastics is facilitated with increased temperatures and high oxygen levels, differences between marine, freshwater and terrestrial systems in generation of secondary microplastics from fragmentation are not well established. In recent years, degradation patterns of microplastics have been observed by examining surface characteristics of the plastic particles and it has been found that degradation patterns of freshwater microplastics are similar to those of microplastics from marine beaches (Zbyszewski et al., 2014; Eerkes-Medrano et al., 2015). According to Eerkes-Medrano et al. (2015), observing degradation patterns of microplastics could be useful in tracking a particle's history and identifying the origin of the plastic particle, which also provides a viewpoint to understand the interaction with physical forces affecting transport of these particles in the environment. Hence, comprehensive examination of the fate and transport mechanisms of microplastics in the environment is necessary to address point and nonpoint sources of these pollutants, as well as primary and secondary sources that contribute to the total amount of microplastics in the environment.

The studies related to microplastic uptake have mainly focused on presence of these small particles in aquatic and terrestrial organisms. Nonetheless, relatively limited number of studies related with toxic effects of microplastics on marine organisms are available in the literature. Indeed, these are fewer on freshwater and terrestrial organisms. To date, adverse consequences of microplastics alone or in combination with other emerging contaminants, and trophic transfer of these along the food chain have been examined mostly under laboratory conditions preparing a synthetic media. This kind of exposure analysis have also been carried out for freshwater and soil organisms; however, field-based examination of microplastics in these communities remain limited. Based on the current knowledge, there is not a certain evidence whether microplastics pose a significant risk or not in the real environment. Effects of altering environmental conditions, different polymer types and the associated level of toxicity in different species in the natural environments are still unknown. Due to methodological limitations, such as lack of accurate classification, only a fraction of microplastic size distribution in environmental compartments are known (Duis and Coors, 2016; Burns and Boxall, 2018). Although several studies demonsrated that particle size of microplastics and their adverse effects on species are inversely correlated (Browne et al., 2008; Ma et al., 2016; Chen et al., 2017), the knowledge on occurrence of small microplastics and nanoplastics and associated impacts on the natural environment are scarce. In order to perform a precise risk assessment of microplastics, these research gaps need to be filled in by more comprehensive monitoring schemes. From this viewpoint, more advanced analytical methods are required to succeed in determining the processes that control the fate of microplastics and their interactions with other contaminants in the environment (Rocha-Santos and Duarte, 2015).

The scientific research indicates that a standardized method for microplastic sampling and analysis is not available in literature. Microplastic research generally requires the large volume of samples due to relatively low concentrations of these particles in the environment. Surface water and water column samples are collected via plankton nets filtering the great volume of water typically with a mesh size of 330 μm. These, surface-trawling plankton nets are typically made of nylon mesh, which pose a risk to increase the contamination potential of the samples and may cause problems during analysis. In addition, the lack of standardized use of plankton nets and different mesh sizes hamper the comparability of the quantitative data on microplastic concentrations in marine and freshwater samples. Furthermore, to date, the sampling methodology used for many years for the marine environment has been applied for microplastic sampling from rivers, yet high flow conditions in rivers is challenging for the use of plankton nets in these environments and in some cases a limiting factor for the microplastic research in freshwater systems. The non-standardized detection methods in natural samples are also a limitation for the calibration and validation of the numerical models, which hinders the comparison of the model outputs with field observations (Besseling et al., 2017; Zhang, 2017). Developing standardized methods in future for collecting, processing and analyzing natural samples would allow direct comparison between studies and facilitate to assess sources, transport pathways and potential risks of microplastics (Horton et al., 2017b).

Microplastic pollution is widespread across the globe, yet potential impacts may be exacerbated, when they accumulate in specific locations. Although presence of microplastics in water, land and air has been demonstrated, these environments are commonly considered independently, but in reality are closely linked. Studying the particle behaviors and transport mechanisms of microplastics within and between the environmental compartments can aid in understanding how and where microplastics will accumulate. The occurrence of plastic debris in rivers and lakes just started to receive attention, and there is a lack of research linking the sources, such as wastewater treatment plant discharge, runoff, and sinks. This is necessary to assess the risks associated with microplastic concentrations. It is likely that microplastic abundance is correlated with proximity to particular industry, wastewater discharges and land use practices within a watershed. The review undertaken here addressed that one of the most understudied areas of microplastics is catchment based research practices. So, it would be very useful for future researchers to focus on the sources, sinks and transport of these emerging pollutants on catchment scale. Indeed, field studies in combination with integrated watershed modeling and risk mapping would provide valuable insights for understanding the issues governing microplastic accumulation in soil and water. Consequently, future studies should focus on development of novel methods and modeling approaches to assess microplastic occurrence, fate and transport in the environment. Short-term and long-term effects of microplastics should be evaluated.

5. Conclusions and recommendations

This paper reviews a vast amount of existing literature related to microplastics in the environment. Section 3 represents the summary and review of over 200 papers classified according to the environmental compartment and research issue (Fig. 1). A more comprehensive critical review is provided in Section 4. According to these, the areas that merits further investigation can be summarized as the following:

1. Developing standardized methods for specific environments to collect, process and analyze the natural samples.
2. Developing a standardized classification and quantification system for macro- and microplastics, both with respect to composition and size/shape.
3. Undertaking field of studies related with highest trophic level, such as marine mammals.
4. Investigating the vertical distribution of microplastics in the water column.
5. Carrying out research in river downstreams and transport to the seas.
6. Collecting data for occurrence of microplastics in the terrestrial environments.
7. Identifying nonpoint sources of microplastics and analysis of data in surface runoff and stormwater.
8. Investigating dry and wet atmospheric fallout, and studying the depositional fluxes at the air-water interface.
9. Determining the fragmentation rates of macroplastics.
10. Performing precise risk assessment of microplastics via comprehensive monitoring schemes.
11. Undertaking field studies in combination with integrated watershed modeling and risk mapping and development of novel methods and modeling approaches to assess microplastic occurrence, fate and transport in the environment.

Declaration

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