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Review

Microplastics in freshwater systems: A review of the emerging threats, identification of knowledge gaps and prioritisation of research needs

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ABSTRACT

Plastic contamination is an increasing environmental problem in marine systems where it has spread globally to even the most remote habitats. Plastic pieces in smaller size scales, microplastics (particles <5 mm), have reached high densities (e.g., 100 000 items per m^3) in waters and sediments, and are interacting with organisms and the environment in a variety of ways. Early investigations of freshwater systems suggest microplastic presence and interactions are equally as far reaching as are being observed in marine systems. Microplastics are being detected in freshwaters of Europe, North America, and Asia, and the first organismal studies are finding that freshwater fauna across a range of feeding guilds ingest microplastics.

Drawing from the marine literature and these initial freshwater studies, we review the issue of microplastics in freshwater systems to summarise current understanding, identify knowledge gaps and suggest future research priorities. Evidence suggests that freshwater systems may share similarities to marine systems in the types of forces that transport microplastics (e.g. surface currents); the prevalence of microplastics (e.g. numerically abundant and ubiquitous); the approaches used for detection, identification and quantification (e.g. density separation, filtration, sieving and infrared spectroscopy); and the potential impacts (e.g. physical damage to organisms that ingest them, chemical transfer of toxicants). Differences between freshwater and marine systems include the closer proximity to point sources in freshwaters, the typically smaller sizes of freshwater systems, and spatial and temporal differences in the mixing/transport of particles by physical forces. These differences between marine and freshwater systems may lead to differences in the type of microplastics present. For example, rivers may show a predictable pattern in microplastic characteristics (size, shape, relative abundance) based on waste sources (e.g. household vs. industrial) adjacent to the river, and distance downstream from a point source.

Given that the study of microplastics in freshwaters has only arisen in the last few years, we are still limited in our understanding of 1) their presence and distribution in the

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environment; 2) their transport pathways and factors that affect distributions; 3) methods for their accurate detection and quantification; 4) the extent and relevance of their impacts on aquatic life. We also do not know how microplastics might transfer from freshwater to terrestrial ecosystems, and we do not know if and how they may affect human health. This is concerning because human populations have a high dependency on freshwaters for drinking water and for food resources. Increasing the level of understanding in these areas is essential if we are to develop appropriate policy and management tools to address this emerging issue.

Contents

1. Introduction

Marine debris has been identified as a factor contributing to biodiversity loss (Gall and Thompson, 2015), and poses a potential threat to human health and activities (Coe and Rogers, 1997; Derraik, 2002; Thompson et al., 2009). Marine debris is mainly comprised of plastic, with 75% of shoreline debris recorded worldwide as being plastic (see reviews by Gregory and Ryan, 1997; Derraik, 2002). Plastic debris is considered a top environmental problem (UNEP, 2005; Gorycka, 2009), and is identified alongside climate change as an emerging issue that might affect human ability to conserve biological diversity in the near to medium-term future (Sutherland et al., 2010).

Plastic debris items, ranging in size from the microscopic to items metres in size, are found in benthic and pelagic habitats in all oceans, and in remote locations such as the Arctic, Southern Ocean and the deep sea (Barnes et al., 2009, 2010; Browne et al., 2011; Van Cauwenberghe et al., 2013; Obbard et al., 2014). Impacts on marine life are influenced by debris size. Large plastic items, such as discarded fishing rope and nets, commonly cause entanglement of invertebrates, birds, mammals, and turtles (Carr, 1987; Fowler, 1987; Laist, 1997; Gall and Thompson, 2015). Smaller plastic items, such as bottle caps, cigarette lighters, and plastic pellets, can be ingested, leading to obstruction of the gut and there is concern about the potential for uptake of chemicals from the plastic (Fry et al., 1987; Laist, 1997; Gall and Thompson,2015; Law and Thompson, 2014). Microplastics (particles <5 mm, Thompson et al., 2009) with maximum estimated densities in the thousands to 100 000 of items per m^3 in surface waters and in the range of 100 000 items per m on shorelines have been recorded (Gregory, 1978; Norén, 2007; Desforges et al., 2014). These particles are ingested by a variety of marine organisms from invertebrates to fish with various consequences (e.g., Thompson et al., 2004; Lusher et al., 2013) and there is evidence that particles smaller than the current level of detection in the environment are also ingested by aquatic invertebrates (Rosenkranz et al., 2009).

The origins of microplastics include primary and secondary sources. Primary sources include manufactured plastic products such as scrubbers in cleaning and cosmetic products, as well as manufactured pellets used in feedstock or plastic production (Gregory, 1996; Fendall and Sewell, 2009; Cole et al., 2011). Manufactured pellets may be especially common in the environment near plastic processing plants whereas scrubbers or microbeads may be present in industrial and domestic wastewater, where they enter the system via rivers and estuaries (Colton, 1974; Hidalgo-Ruz et al., 2012). Manufactured pellets have also been found in beaches distant from pellet processing plants suggesting potential for their long-range marine transport (Costa et al., 2010). Secondary sources of microplastics include fibres or fragments resulting from the breakdown of larger plastic items (Browne et al., 2011; Cole et al., 2011). These fragments can originate from fishing nets, line fibres, films, industrial raw materials, consumer products and household items, and pellets or polymer fragments from degradable plastic, which are designed to fragment in the environment (Hidalgo-Ruz et al., 2012; Free et al., 2014). Microplastics from secondary sources may be associated with sites of higher population densities, though understanding of drivers for microplastic distributions is limited (Browne et al., 2011; Doyle et al., 2011; Ballent et al., 2012; Desforges et al., 2014). Secondary sources are believed to be the main origin of most microplastics in marine environments (Hidalgo-Ruz et al., 2012) although our knowledge about the relative importance of various inputs is incomplete (Law and Thompson, 2014).

It is not viable to remove microplastics from habitats due to their small size and their continuous evolution via the breakdown of larger items. Hence measures focused on reducing inputs are widely recognized as being the most effective. However, even if we were able to completely stop inputs of debris to the environment, the quantity of microplastics would likely increase because of fragmentation of larger plastic items already in the environment – legacy inputs of microplastic. We have a poor understanding of degradation rates and of fragmentation, and this is of concern because the spread and abundance of microplastics is increasing (Browne et al., 2011; Law and Thompson, 2014).

Global plastic production has increased exponentially since the 1960s, with production in 2013 at 299 million tonnes (Rochman et al., 2013a; PlasticsEurope, 2014). Despite wide research efforts investigating plastics in oceans, little research has focused on freshwater and terrestrial systems (Thompson et al., 2009; House of Commons, 2013; Wagner et al., 2014) and there are very few studies of microplastic in freshwaters. Given this paucity of information about microplastics in freshwater systems, the current paper focuses on four topics:

1) **A review of current knowledge on the presence and distribution of microplastics in freshwaters;**
2) **How the presence of microplastics in freshwater systems may be detected;**
3) **How the presence of microplastics in freshwater systems may impact aquatic organisms;**
4) **What might be done to better understand and manage this emerging problem.**

Using understanding of relevant marine literature, and initial studies of microplastics in freshwater systems, we compare and contrast the various factors surrounding the topic (e.g. distributions, methods of quantification, impacts). We draw attention to the opportunities that an increased understanding of microplastics in freshwater systems may bring for management of plastic contamination.

2. Microplastics in the environment

2.1. *Microplastic presence in freshwater systems*

The body of knowledge on the accumulation and effects of plastics in freshwater and terrestrial systems is much less than in marine systems (Thompson et al., 2009; House of Commons, 2013; Wagner et al., 2014). In oceans, the small size and low density of microplastics contributes to their widespread transport across large distances particularly by ocean currents (Cole et al., 2011; Ballent et al., 2012; Eriksson et al., 2013). Their presence has been noted on coastlines of all continents (e.g. Browne et al., 2011; Zurcher, 2009; Ivar do Sul and Costa, 2007), in remote locations such as mid-Atlantic archipelago islands (Ivar do Sul et al., 2009; Ivar do Sul et al., 2013), sub Antarctic islands (Eriksson et al., 2013), the Arctic (Obbard et al., 2014), and even in deep-sea habitats (Van Cauwenberghe et al., 2013).

Until recently the distribution of microplastics in freshwater systems as in marine systems was unknown. Even large plastic items (e.g., fragments >5 mm, line, films, and polystyrene) have only recently been recorded in lakes (Faure et al., 2012), rivers (e.g., Williams and Simmons, 1996; Moore et al., 2011) and estuaries (e.g., Morritt et al., 2014; Sadri and Thompson, 2014). In the last few years, studies have been identifying *micro*plastics in varied freshwater systems across continents (Table 1). Microplastics have been found in: North America, in the Los Angeles basin (Moore et al., 2011), the North Shore Channel of Chicago (Hoellein et al., 2014), the St. Lawrence River (Castañeda et al., 2014) and the Great Lakes (Zbyszewski and Corcoran, 2011; Zbyszewski et al., 2014; Eriksen et al., 2013); in Europe, in Lake Geneva (Faure et al., 2012), the Italian Lake Garda (Imhof et al., 2013), the Austrian Danube river (Lechner et al., 2014), the German Elbe, Mosel, Neckar, and Rhine rivers (Wagner et al., 2014), and the UK Tamar estuary (Sadri and Thompson, 2014); and in Asia, in Lake Hovsgol, Mongolia (Free et al., 2014). The microplastics detected in these studies are of varied origins including primary and secondary sources and are of different compositions (Table 1).

2.2. *Microplastic sources*

Authors have suggested that primary source microplastics entering marine systems include polyethylene, polypropylene, and polystyrene particles in cleaning and cosmetic products, which enter the aquatic system through household sewage discharge (Zitko and Hanlon, 1991; Gregory, 1996; Fendall and Sewell, 2009). Other primary microplastics suggested to enter aquatic systems include those of industrial origin in spillage of plastic resin powders or pellets used for airblasting (Gregory, 1978, 1996), and feedstocks used to manufacture plastic products (Lechner et al., 2014; Zbyszewski et al., 2014). Secondary microplastics originate from the breakdown of larger plastic items. Breakdown may occur before microplastics enter the environment, e.g.

Table 1 – Studies detecting microplastics in freshwaters. Table entries are ordered alphabetically by continent and then study authors.

Water body name & location	Study authors	What was sampled	Size classes, and Sampling mesh size for water samples (where reported)	Maximum abundance, and Mean abundance (where reported)
Lake Hovsgol, Mongolia, Asia	Free et al., 2014	Surface water	Size classes: 0.355–0.999 mm, 1.00–4.749 mm, and >4.75 mm Sampling mesh: 333 μm	Max: 44 435 items km^{-2}, Mean: 20 264 items km^{-2} Abundances include all particles, of which 81% represents size <4.75 mm
Lake Geneva, Europe	Faure et al., 2012	Sediment & Surface water	Size classes: <2 mm, <5 mm (sediments) <5 mm, >5 mm (water) Sampling mesh: 300 μm	Max: 9 items per sample (sediment), 48 146 items km^{-2} (water) Mean: not indicated Item size class: <5 mm
Lake Garda, Italy, Europe	Imhof et al., 2013	Sediment	Size classes: 9–500 μm, 500 μm–1 mm, 1–5 mm, >5 mm	Max: 1108 ± 983 items m^{-2} Mean: not indicated Item size class: <5 mm
Danube river, Austria, Europe	Lechner et al., 2014	Surface water	Sizes classes: <2 mm, 2–20 mm Sampling mesh: 500 μm	Max: 141 647.7 items 1000 m^{-3}, Mean: 316.8 (±4664.6) items 1000 m^{-3} Abundances include all particles, of which 73.9% represent spherules (~3 mm)
Tamar estuary, UK, Europe	Sadri and Thompson, 2014	Surface water	Size classes: <1 mm, 1–3 mm, 3–5 mm, >5 mm Sampling mesh: 300 μm	Max: 204 pieces of suspected plastic Mean: 0.028 items m^{-3} Abundances include all plastic particles, of which 82% represents size <5 mm
Elbe, Mosel, Neckar, and Rhine rivers, Germany, Europe	Wagner et al., 2014	Sediment	Size classes: <5 mm	Max: 64 items kg^{-1} dry weight Mean: not indicated Item size class: <5 mm
St. Lawrence River, Canada/USA, North America	Castañeda et al., 2014	Sediment	Size classes: not indicated. Sampling mesh: 500 μm.	Max: not indicated Mean: 13 759 (±13 685) items m^{-2} Highest mean site density: 136 926 (±83 947) items m^{-2} Items size range: 0.4 to 2.16 mm
Lakes Superior, Huron, and Erie, Canada/USA, North America	Eriksen et al., 2013	Surface water	Size classes: 0.355–0.999 mm, 1.00–4.749 mm, >4.75 mm Sampling mesh: 333 μm	Max: 463 423 items km^{-2} Mean: 43 157 items km^{-2} Abundances include all particles, of which 98% represents size <4.75 mm
North Shore Channel of Chicago, USA, North America	Hoellein et al., 2014, Abstract	Not indicated	Microplastics defined as 0.3–5 mm	Higher microplastic counts downstream of a wastewater treatment plant (WWTP) than upstream of the WWTP Max and Mean: not indicated
Los Angeles River, San Gabriel River, Coyote Creek, USA, North America	Moore et al., 2011	Surface, mid, and near-bottom water	Size classes: >=1.0 and <4.75 mm, >=4.75 mm Sampling mesh: 333, 500, and 800 μm	Max: 12 932 items m^{-3} Mean 24-h particle counts on date of greatest abundance: Coyote creek: 4999.71 items m^{-3} San Gabriel river: 51 603.00 items m^{-3} Los Angeles River: 1 146 418.36 items m^{-3} Item size class: 1.0–4.75 mm

Lake Huron, Canada/USA, North America	Zbyszewski and Corcoran 2011	Sediment	Size classes: <5 mm plastic pellets, >5 mm broken plastic, polystyrene	Lake Huron total pieces: 3209, represented by 2984 pellets, 108 fragments, and 117 pieces of Styrofoam Mean: not indicated
Lakes Erie and St. Clair, Canada/USA, North America	Zbyszewski et al., 2014	Sediment	Size classes: styrofoam, pellets, plastic fragments (<2 cm), intact or near-intact debris	Lake Erie total pieces: 1576, represented by 603 pellets, 934 plastic fragments, and 39 pieces of Styrofoam Lake St. Clair total pieces: 817, represented by 110 pellets, 192 plastic fragments, 234 pieces of styrofoam, and 281 intact or near-intact debris Mean: not indicated

synthetic fibres from the washing of clothes (Browne et al., 2011), or after due to environmental weathering of plastic items (Andrady, 1994, 1998). Secondary microplastics arising as fibres from washing clothes, are mainly made of polyester, acrylic, and polyamide, and may reach more than 100 fibres per litre of effluent (Habib et al., 1998; Browne et al., 2011). Fibres similar to those in household sewage effluent have been found to be dominant at sewage disposal sites and exhibit long residence times. These secondary source microplastics are therefore also likely to have long residence times in freshwater systems (Zubris and Richards, 2005; Browne et al., 2011), whether they be natural water bodies (rivers and lakes), modified water bodies (e.g. dammed reservoirs), or artificial water bodies (artificial lake).

Primary and secondary microplastics have been detected by initial freshwater studies across varied systems (Table 1). Primary microplastics of household origin, of a similar size, shape, colour and elemental composition as microbeads from commercial facial cleansers, have been confirmed in samples from North American Great Lakes (Eriksen et al., 2013). Primary microplastics of industrial origins have been detected in rivers and lakes. Pre-production plastic resin pellets were the second most dominant debris in rivers from the Los Angeles basin (Moore et al., 2011) and the most dominant debris in Lake Huron (Zbyszewski and Corcoran, 2011). Authors suggested the plastic raw materials in samples from the Danube River, Lake Huron, and Lake Erie likely were released from plastic production sites (Zbyszewski and Corcoran, 2011; Zbyszewski et al., 2014; Lechner et al., 2014). Secondary microplastics have been found in Lake Hovsgol, Mongolia, and in Lake Garda, Italy, where fragments were the dominant form of microplastic (Imhof et al., 2013; Free et al., 2014). In both studies, the authors suggested these secondary microplastics came from degradation and breakdown of larger plastic items of household origin (Imhof et al., 2013; Free et al., 2014).

These studies indicate spatial associations between the types of microplastics found and human activities. The sources of microplastics can often be identified by either the nature, or relative abundance of the microplastic material. For example, raw plastic (pellets and flakes) were found in the Danube, a river that has plastic production sites adjacent to it (Lechner et al., 2014); resin pellets and microbeads were most abundant in the industrial region of Lake Huron and the densely populated and industrial lake Erie (Zbyszewski and Corcoran, 2011; Eriksen et al., 2013); the lack of primary pellets but an abundance of secondary fragments in the shores of the sparsely populated mountain lakes (Garda and Hovsgol) suggested an origin from the breakdown of household items (Imhof et al., 2013; Free et al., 2014).

Differences between freshwater and marine systems in generation of secondary source microplastics from environmental weathering are not known. Even for marine systems, fragmentation and degradation rates of microplastics are unknown (Law and Thompson, 2014). There may be differing degrees of physical forces, such as storms and wave action in marine systems, but plastics in freshwater systems still experience physical and chemical degradation (Andrady, 2011). Free et al. (2014) investigating microplastics in Lake Hovsgol suggested that particles may experience relatively high levels of weathering due to increased UV light penetration and reduced biofouling in oligotrophic lake waters (Free et al., 2014).

Freshwater studies employing scanning electron microscopy to examine the surface of microplastics (Zbyszewski and Corcoran, 2011; Imhof et al., 2013) have reported degradation patterns (cracks, pits and adhering particles) similar to those observed in plastics from marine beaches (Gregory, 1978; Corcoran et al., 2009). Observing degradation in surface characteristics of microplastics can be useful in tracing a particle's history. Surface characteristics can reveal whether the particle experienced mechanical degradation (e.g. wave action, sand friction, Zbyszewski et al., 2014), oxidative weathering (e.g. photo-oxidation from UV-B exposure, Zbyszewski et al., 2014), or potentially biological degradation (e.g., hydrocarbon degrading microbes, Zettler et al., 2013) and can provide insights into depositional environments (e.g. sandy beaches vs. muddy organic-rich shorelines) the particles came from Zbyszewski et al. (2014). Degradation patterns are important to consider, as the shape, size, density, and texture of microplastics contributes to the way particles interact with factors that affect their presence in the environment (section 2.3), and the physical forces that drive their transport (section 2.4; Ballent et al., 2012).

2.3. *Factors affecting quantity of microplastics in the environment*

A number of factors have been suggested to affect the quantity of microplastics present in freshwater environments. These, in addition to physical forces (section 2.4), include human population density proximal to the water body, proximity to urban centers, water residence time, size of the water body, the type of waste management used, and amount of sewage overflow (Moore et al., 2011; Zbyszewski and Corcoran, 2011; Eriksen et al., 2013; Free et al., 2014). In the Great Lakes of North America, pelagic microplastic counts reached up to 1101 particles in a tow of 3.87 km (466 305 particles km^{-2}) in the highly populated Lake Erie, while particle counts for the less populated Lakes Huron and Superior reached 15 particles in a tow of 3.76 km (6541 particles km^{-2}) and 15 particles in a tow of 1.94 km (12 645 particles km^{-2}) respectively (Eriksen et al., 2013). Greater microplastic densities were detected in the southern parts of Lake Huron, North America, and Lake Hovsgol, Mongolia, where the lakes experience industrial activity and tourism respectively (Zbyszewski and Corcoran, 2011; Free et al., 2014). However, even in Lake Hovsgol, a remote area with low population densities, the estimated pelagic microplastic densities reached 44 435 particles km^{-2} (Free et al., 2014). The authors suggested that high pelagic particle counts in this less populated lake might be a result of the long water residence time and small lake size concentrating particles. They suggested such patterns might also explain why the larger Lakes Huron and Superior had low pelagic microplastic particle counts (Eriksen et al., 2013) relative to the high microplastic densities of the relatively smaller Lake Geneva (Faure et al., 2012).

With regards to the relationship between microplastic presence and wastewater treatment, authors suggest that population uses of certain products, e.g. microbeads in cosmetic/cleaning products, in conjunction with wastewater treatments which are unable to capture floating microplastics, contributes to the presence of microplastics in freshwater bodies (Eriksen et al., 2013). These authors also suggest that combined sewage overflow employed in the Great Lakes contributed to presence of microbeads in samples. Microplastic concentrations may also vary with proximity to wastewater treatment facilities. In the North Shore Channel of Chicago microplastic densities were higher downstream from a wastewater treatment plant than upstream of the plant (Hoellein et al., 2014). This sampling design that included sites upstream and downstream from a wastewater plant, highlights the importance of sampling design in influencing observed patterns of microplastic presence (more in section 3.2).

2.4. *Factors involved in dispersal*

Microplastic distributions in marine environments are still not fully known, but key for estimating global distributions is an understanding of the external forces that drive their movements. Quantitative and modelling approaches point to the role of varied physical forces influencing transport and dispersal at a range of spatial scales. An observational and modelling study showed that large-scale forces such as wind driven surface currents and geostrophic circulation drive dispersal patterns of microplastics in the western North Atlantic Ocean and Caribbean Sea (Law et al., 2010). Meanwhile at smaller scales, experimental and field evidence points to wind driven turbulence influencing vertical position of neustonic particles (Ballent et al., 2012; Kukulka et al., 2012), while models show that turbulent flows, from tides or waves, can lead to resuspension of benthic particles (Ballent et al., 2012, 2013). Physical forces even play a role in position of particles within marine sediments. An evaluation of the three dimensional position of microplastics within marine sediments in Santos Bay, Brazil, provided evidence that deposition of particles might be related to high energy oceanographic events like sea storms (Turra et al., 2014).

External forces that drive dispersal interact with properties of the particles themselves (e.g. density, shape, and size) and other properties of the environment such as seawater density, seabed topography, and pressure (Ballent et al., 2012, 2013). Particle density frequently shows up as a factor influencing transport and dispersal in marine studies (Law et al., 2010; Morét-Ferguson et al., 2010; Ballent et al., 2012, 2013). Common consumer plastics range in density from 0.85 to 1.41 g ml^{-1}, where polypropylene and low/high density polyethylene (LDPE, HDPE) plastics have densities lower than 1 g ml^{-1}, and polystyrene, nylon 6, polyvinyl chloride (PVC), and polyethylene terephthalate (PET) have densities higher than 1 g ml^{-1}. Sources for fibres and fragments of low-density plastics include bags, rope, netting, and milk/juice jugs, and sources for high-density particles include food containers, beverage bottles, and films (Andrady, 2011). Since this range includes material of lower, equal, or higher density than water, microplastics can be distributed throughout the water column (Morét-Ferguson et al., 2010). Thus, particle density can determine whether a particle occupies a pelagic versus benthic transport route; low-density plastics occupy the surface and neustonic environment, while high-density plastics are found at depth and on the benthos (Morét-Ferguson et al., 2010). Degradation through biological and physical processes and fouling by a succession of epibionts can affect particle dispersal by changing the size and molecular weight of plastics (Morét-Ferguson et al., 2010). Particles may cycle through the marine water column if they undergo cycles of fouling and defouling (Andrady, 2011; Lobelle and Cunliffe, 2011).

Initial freshwater studies are finding that similar physical forces to those suggested for marine systems contribute to microplastic transport and dispersal. In Lake Hovsgol, Mongolia, wave energy was a significant predictor of microplastic distributions. A south-to-north decrease in microplastic presence observed in the study was suggested to arise from: 1) entry of plastics at the more urbanised southwestern shore, 2) northward transport by southwesterly winds, and 3) southerly concentration of particles by the lake's drainage through the Eg River to the south. The study authors also suggested the degree of fouling might affect particle presence on the lake surface where wave energy acts on particles (Free et al., 2014). Similarly, southerly winds leading to surface circulation and a rotating eddy at the northern tip of Lake Garda, Italy, was suggested to explain patterns of microplastic distribution (Imhof et al., 2013), and in Lake Erie patterns of particle density were explained by converging currents near the sample sites (Eriksen et al., 2013). In the Los Angeles River, USA, microplastic density was highest in samples collected in the wet season, mid channel, and near the surface rather than samples collected in the dry season, mid-column or near the bottom of the water column, or near the river bank (Moore et al., 2011).

Based on studies of suspended sediments, other physical factors that might influence particle transport in freshwater include flow velocity, water depth, substrate type, bottom topography, and seasonal variability of water flows (Simpson et al., 2005). Factors that may have a temporal aspect include: tidal cycle (only in estuaries), storms, floods, or anthropogenic activity (e.g. dam release) (Moatar et al., 2006; Kessarkar et al., 2010). A range in transport distances might arise from physical forces interacting with particle characteristics (density, size and charge). An example is variability in sediment flux as a river runs to an estuary. Particles of high density may occupy the benthic transport route as bedload and be deposited in the lower reaches of the river, while particles of fine-size fractions and low density may occupy the pelagic transport route in suspension and be carried into estuaries and beyond into the sea (Eisma and Cadeé, 1991). On reaching an estuary, turbulence and salinity can interact with particle density, size, and charge, leading to increased flocculation and particle deposition (Kranck, 1975; Olsen et al., 1982; Eisma and Cadeé, 1991). These interactions may similarly occur in microplastics, leading to increased deposition where fresh and saline waters meet. These various transport patterns may be affected at larger temporal scales by seasonal variations in river discharge (Eisma and Cadeé, 1991; Moatar et al., 2006; Kessarkar et al., 2010).

2.5. *Freshwater systems as contributors to microplastics in oceans*

Whether rivers are major sources of microplastics to the ocean has yet to be established. Microplastics are present in sewage discharge (Browne et al., 2011), in effluent from plastic manufacturing plants (Hays and Cormons, 1974), in urban runoff (Lattin et al., 2004), and in rivers (Moore et al., 2011; Hoellein et al., 2014; Lechner et al., 2014; Wagner et al., 2014). In the Danube River microplastic litter was numerous with industrial raw materials accounting 79% of plastics (Lechner et al., 2014), and in the Los Angeles River microplastics were the most dominant size range of plastic items caught in sampling nets (Moore et al., 2011, Table 1). Therefore, the role of freshwater systems as transport routes for microplastics to oceans needs to be considered.

The link between marine pollution and rivers is clear for other types of pollutants from municipal discharges, sewage, urban runoff and stormwater (Olsen et al., 1982; Abril et al., 2002; U.S. EPA, 2009; EEA, 2012). Legal frameworks set up across international boundaries, such as the European Union's Water Framework Directive (Directive, 2000/60/EC) and Marine Strategy Framework Directive (Directive, 2008/56/EC), promote integrated management of freshwaters and marine waters, and part of this management involves addressing pollution including materials in suspension (EC, 2010) and microplastics (MSFD; Galgani et al., 2010). One of the few studies looking at fluxes of plastics in and out of an estuary suggests that the Tamar River, UK, in late spring and in summer was neither a source nor a sink, with as many microplastic particles entering the estuary as leaving it (Sadri and Thompson, 2014). It is notable that the Tamar estuary is not highly populated, and therefore estuaries receiving inputs from highly industrialized or populated catchments might be expected to make greater contributions of microplastics to the ocean. For other pollutants, population density, land use, and the level of sewage treatment are all correlated with pollutant inputs into rivers and estuaries (Abril et al., 2002).

3. Detecting and monitoring microplastics

3.1. *Sampling and identification*

Despite an increasing understanding of microplastic presence across marine geographic locations and habitats, the cost and difficulties of sampling microplastics from benthic and pelagic habitats limit present knowledge of spatial and temporal distributions (Hidalgo-Ruz et al., 2012; Galgani et al., 2013; NOAA Office of Response and Restoration, 2013); techniques are generally time consuming and unable to identify all particles (Galgani et al., 2013). Challenges of detecting microplastics include: 1) the ability to capture plastic particles from a sample of water or sediment; 2) separating the plastic fragments from other particles in the sample; and 3) identifying the types of plastics present and dealing with the difficulties of identification from processes such as discolouration by biofilms on microplastics (Hidalgo-Ruz et al., 2012; Eriksen et al., 2013).

In marine investigations, the techniques for sampling microplastics vary, with approaches differing in collection method, identification, and enumeration (Hidalgo-Ruz et al., 2012). They include selective sampling and bulk or volume-reduced sampling. Selective sampling has been applied to surface sediments, while bulk or volume-reduced sampling has been used in sampling sediments or water parcels. Once samples are obtained, plastics are separated from the sample by density separation, filtration, sieving, and/or visual sorting. Characterisation of particles has used morphological descriptions, source, type, shape, colour, chemical composition, and degradation stage of particles. The most reliable method of identification has been infrared spectroscopy, which reveals chemical composition (Hidalgo-Ruz et al., 2012). The importance of using a reliable identification method is illustrated by Eriksen et al. (2013), who studied the elemental composition of particles that were visually identified as microplastics. They found that many particles initially identified as plastic were actually aluminium silicates and these in some replicates made up 20% of the 0.355–1 mm size fraction of particles (Eriksen et al., 2013).

Sampling methods similar to those used in marine systems (e.g., Thompson et al., 2004), are used to detect microplastics in freshwater systems (e.g., Eriksen et al., 2013; Imhof et al., 2013). Methods need fine enough filters and the addition of a substance to the water or slurry to increase the water density sufficiently to float the plastics (Hidalgo-Ruz et al., 2012; Imhof et al., 2012, 2013). A challenge of investigations is to separate low-density materials and to extract and identify microplastics <500 μm, but continued method development is improving researcher's ability to do this (Imhof et al., 2012, 2013). One recent method is the Munich Plastic Sediment Separator (Imhof et al., 2012), which, by applying a higher density of separation fluid, can separate plastic particles in a range of sizes: mesoplastic and large microplastic particles in the range of 20–5 mm and 5–1 mm, as well as small microplastic particles (<1 mm). The approach, which reliably separates plastics of all polymer types, in different size classes and with varying physical properties (Imhof et al., 2012), was applied in a recent freshwater study of Lake Garda Italy, and succeeded in extracting and identifying particles down to 9 μm (Imhof et al., 2013). Another recent method, developed by Claessens et al. (2013), applies elutriation to separate microplastics from sediments with high extraction efficiencies (93–98%). This group has also developed a technique to extract microplastics from biota with similarly high extraction efficiencies (Claessens et al., 2013).

3.2. *Considerations for method development*

The emergence of methods that are better able to separate size ranges and polymer types is improving our ability to measure and detect microplastics, however, it is too early to select a unified approach. Method development needs to involve discussion of how to: 1) keep methods simple to ensure sufficient replication to account for natural variability, 2) keep costs low enough to enable method accessibility, 3) have methods that are precise and accurate, and 4) have methods that minimize contamination. Microplastics are not regularly monitored so there is no available baseline information at present (Galgani et al., 2010, 2013). As there is still a lack of understanding on the potential for microplastics to cause harm, it might be premature to standardize monitoring approaches without knowing what spectrum, size ranges and types, of microplastics are of interest.

Discussion of the cost/benefit of a monitoring approach, and the time requirements of processing, might be especially important in scenarios where regular monitoring is needed to determine geographic origins of waste (Galgani et al., 2013). In these cases, an inexpensive, simple to use, safe, and quick method may be most desirable. Another scenario where an inexpensive and easy to use method might be especially desirable is in monitoring efforts by developing countries where environmental policies operate on a limited budget (Free et al., 2014). In such cases, density separation by the NaCl method (Thompson et al., 2004), which may be less complete in its extraction efficiency, but is simple, inexpensive, rapid and does not use hazardous chemicals, may be most appropriate.

Monitoring efforts also need to be context dependent, taking into account the site-specific physical and biological drivers that might affect microplastic distributions and concentrations. For example, both advective (influenced by velocity field) and diffusive/dispersive (influenced by turbulence) transport may affect distributions, and both processes would vary with the nature of the water body, depending on factors such as geology (including substrate type) and relief (Whitehead and Lack, 1982; Moatar et al., 2006). Illustrations of physically influenced particle distributions include: 1) Lake Erie sampling stations with anomalously high particle counts occurred at a site of converging currents (Eriksen et al., 2013); 2) timed sampling in mid-channel surface river waters resulted in higher particle counts than samples collected at the river bank or in bottom waters (Moore et al., 2011); and 3) a dispersion gradient from shoreline sources was likely reflected in higher particle counts at lake shore samples than samples collected further from shore (Eriksen et al., 2013). Another consideration for monitoring efforts is the residence time of a water body. High particle abundances might be related to residence time of lake waters (e.g., Lake Hovsgol, Mongolia, Free et al., 2014) or to the amount of seasonally driven runoff in a river (e.g., LA basin rivers, USA, Moore et al., 2011). Vertical variations in particle abundances are influenced by wind-driven vertical mixing (Kukulka et al., 2012); for monitoring purposes the most reliable concentrations would be measured under no wind conditions. Thus, within a water body, physically driven spatial patterns and temporal patterns can affect observed distributions and abundance patterns. Whether monitoring is to occur in rivers, lakes, estuaries, marine coastlines, or other aquatic habitats, the hydrodynamic characteristics of the site, in space and in time, as well as the prevailing weather (wind, rainfall) need to be considered.

Development of methods to detect, identify, measure, and monitor microplastics can benefit from studies under way for marine and freshwater systems. As nations are increasingly focused on monitoring and achieving good water quality and ecosystem health (e.g. Europe's Directive, 2000/60/EC and Directive, 2008/56/EC), the timing is right to invest research efforts in method development and between laboratory intercomparability.

4. Potential impacts

4.1. Which biota interact with microplastics?

Initial freshwater field and laboratory studies have demonstrated that five species of freshwater invertebrates, one species of freshwater fish, nine species of brackish fish, and one species of amphidromous fish can ingest microplastics (Table 2 and references therein). In the freshwater invertebrate study between 32 and 100% of exposed individuals ingested microplastics (Imhof et al., 2013). The only freshwater river field study to date shows that gobies collected from 7 out of 11 French streams contained microplastics (Sanchez et al., 2014). In the marine field more research on organismal impacts has been carried out, showing that a wide array of animals ingest microplastics (Table 3).

Marine animals ingesting microplastics include benthic and pelagic organisms, possessing varied feeding strategies and occupying different trophic levels. Benthic marine invertebrates that ingest microplastics include sea cucumbers (Graham and Thompson, 2009), mussels (Browne et al., 2008; Farrell and Nelson, 2013), lobsters (Murray and Cowie, 2011), amphipods, lugworms, and barnacles (Thompson et al., 2004; Browne et al., 2013; Wright et al., 2013a). Some invertebrates preferentially select plastic particles; deposit and suspension feeding sea cucumbers from benthic habitats ingest a disproportionately high number of plastic fragments and fibres from a given ratio of plastic to sand (Graham and Thompson, 2009). In pelagic marine habitats, microplastics are ingested by a range of zooplankton taxa (Cole et al., 2013; Setälä et al., 2014) and by adult and larval fish (Carpenter et al., 1972; Browne et al., 2013; Lusher et al., 2013; Rochman et al., 2013b). The first freshwater investigation of ingestion by an array of invertebrates shows that, as in marine studies, animals across habitats, feeding guilds, and trophic levels ingest microplastics (Table 2; Imhof et al., 2013). Even at the most basic organismal level, diverse microbial communities that include heterotrophs, autotrophs, predators and symbionts, associate with microplastics (Zettler et al., 2013).

At higher trophic levels, seabirds ingest microplastics directly as well as indirectly, via fish that have consumed microplastics (Hays and Cormons, 1974; Ryan et al., 1988; Tanaka et al., 2013). Ingestion of microplastics by fur seals and sea lions in sub Antarctic islands is evidence of microplastics reaching the highest trophic levels of a marine foodweb even in remote locations (McMahon et al., 1999; Eriksson and Burton, 2003). These large marine mammals most probably obtain microplastics through trophic transfer via their ingestion of fish; an analysis of sea lion scats identified 1 mm plastic fragments only when otoliths from the fish *Electrona subaspera* were present (McMahon et al., 1999). Microplastics can have average densities of 1–1.9 pieces per fish (Carpenter et al., 1972; Lusher et al., 2013), but magnification through the food web suggests a concentration factor of between 22 and 160 times in seals (Eriksson and Burton, 2003). It is possible large vertebrates associated with freshwaters, e.g., waterfowl, may ingest microplastics, either directly or through ingestion of other organisms. In freshwaters, waterfowl, upland game birds (e.g. Ring-necked Pheasants *Phasianus colchicus*, Gray Partridge *Perdix perdix*), and shorebirds ingest lead shot, which poses a problem due to storage of particles in bird gizzards (Scheuhammer and Norris, 1995). Microplastics may also be ingested by freshwater birds and stored in gizzards.

4.2. How do microplastics affect organisms?

In marine organisms, ingestion of large plastic items may cause choking, internal or external wounds, ulcerating sores, blocked digestive tracts, false sense of satiation, impaired feeding capacity, starvation, debilitation, limited predator avoidance, or death (Gregory, 2009; Gall and Thompson, 2015). The impacts on marine organisms of ingesting *micro*plastic-sized particles are largely unknown (Wright et al., 2013b; Law and Thompson, 2014), but initial investigations provide evidence of physical impacts (Table 3). Evidence for impacts of microplastic ingestion on freshwater taxa is much more limited, both in the number of studies conducted and in the number of taxa investigated. The few freshwater studies to date, however, may be suggestive of physical impacts being similar to those in marine studies (Table 3).

In laboratory experiments with the marine *Nephrops* lobster, plastic fragments (5 mm) were not readily excreted, and observations of field specimens show that plastic fibres can form filament balls in the stomach, presumably through churning activity (Murray and Cowie, 2011). Plastic particles may be differentially retained based on size and density (Table 3). When fed plastic beads of different sizes and densities, the sea scallop *Placopecten magellanicus* retained larger (20 μm) and lighter (1.05 g ml^{-1}) particles longer than smaller (5 μm) and denser (2.5 g ml^{-1}) particles (Brillant and MacDonald, 2000). Such differential retention of microplastic, which lacks in nutrition value, may affect the nutritional gain of the sea scallop in environments of microplastic presence. Reduced energy reserves may be the result of inflammatory responses of tissues to microplastics (e.g., in the marine lugworm, *Arenicola marina*) or of a reduction in feeding or false satiation from particle accumulation in digestive cavities (e.g., in *A. marina*) (Wright et al., 2013a). Similarly in field collected estuarine *Eugerres brasilianus* fish, adults that ingested plastic fragments (<5 mm) had lower mean total weight of gut contents potentially indicating reduction in feeding or false satiation (Ramos et al., 2012). In freshwater taxa, particle (size: 20 and 1000 nm) accumulation and retention has been observed in the freshwater water flea, *Daphnia magna* (Rosenkranz et al., 2009).

Studies also show potential microplastic effects at the tissue and cellular level (Table 3). In *Mytilus edulis*, ingested microplastics (size: >0–80 μm) can cause an inflammatory response in tissues and reduced membrane stability in cells of the digestive system (von Moos et al., 2012). Particles (sizes: 3 and 9.6 μm) are also translocated from the digestive system into the circulatory system of *M. edulis*, where they can persist for more than 48 days (Browne et al., 2008). In the freshwater *Daphnia*, ingested microplastics (size: 20 and 1000 nm) have been shown to cross over into cells and translocate to oil storage droplets (Rosenkranz et al., 2009). Japanese medaka fish, *Oryzias latipes*, fed virgin and marine polyethylene fragments (size: <0.5 mm) exhibit bioaccumulation, liver stress response (glycogen depletion, fatty vacuolation and single cell

Table 2 – Freshwater field and laboratory investigations of microplastic and organism interactions.

Study authors, field/lab study	Particle size, composition	Study aim	Taxa	Microplastic uptake? Yes/No/NA	Additional results
Dantas et al., 2012, field study	Size not indicated, nylon fragments	To determine plastic ingestion in two drum species in relation to varying season, habitat, and size-class.	Drum, juvenile, sub-adult, and adult, *Stellifer brasiliensis* and *Stellifer stellifer* (found in estuaries)	Yes	Between 6.9 and 9.2 % of individuals across all species ingested plastic. All size classes ingested plastic. Plastic ingestion differed by season, habitat and size class: Adults in the late rainy season in the middle estuary had the highest number of ingested fragments in their guts.
Hoellein et al., 2014 (conference abstract), field study	Not indicated	To detect microplastic sources, abundance, and effects in rivers.	Bacterial community (sequencing ongoing)	NA	Dense bacterial biofilms on microplastic.
Imhof et al., 2013, lab study	29.5 ± 26 μm (mean ± SD), polymethyl methacrylat	To measure microplastic uptake by freshwater fauna.	Cladoceran freshwater water flea, *Daphnia magna*	Yes	100% of individuals ingested microplastics
			Amphipod crustacean, *Gammarus pulex*	Yes	96 ± 0.03% (mean ± SE) of the faeces contained microplastic
			Clitellate worm, *Lumbriculus variegatus*	Yes	93 ± 0.07% (mean ± SE) of individuals ingested microplastics
			Ostracod, *Notodromas monacha*	Yes,	32.4 ± 3.8% (mean ± SE) of exposed individuals ingested microplastics
			Gastropod freshwater snail, *Potamopyrgus antipodarum*	Yes	87.8 ± 1.9% (mean ± SE) of the faeces contained microplastic
Oliveira et al., 2013, lab study	1 and 5 μm, polyethylene	To determine if microplastics modulate short-term toxicity of contaminants (pyrene).	Common goby, *Pomatoschistus microps* (found in estuaries)	Not indicated	Fish exposed to pyrene had delayed mortality when microplastics were present. Microplastics presence also led to increased pyrene metabolites.
Possatto et al., 2011, field study	Millimetre scale, nylon fragments and hard plastic	To determine ingestion of plastic debris by three catfish species at three size classes.	Catfish, juvenile, sub-adult, and adult, *Cathorops spixii*, *Cathorops agassizii*, *Sciades herzbergii* (found in estuaries)	Yes	Between 17 and 33 % of individuals across all species ingested plastic. All size classes ingested plastic. Size classes differed in number of ingested fragments.
Ramos et al., 2012, field study	1–5 mm, blue nylon fragments	To determine ingestion of plastic debris by 3 gerreid species at three size classes in the Goiana estuary.	Gerreidae fish, juvenile, sub-adult, and adult, *Eugerres brasilianus*, *Eucinostomus melanopterus* and *Diapterus rhombeus* (found in estuaries and mangroves)	Yes	Between 4.9 and 33.4 % of individuals across all species ingested plastic. All size classes (except *D. rhombeus* juveniles) ingested plastic. Species differed in the number and weight of ingested fragments. Size classes differed in number of ingested fragments. Adults of *E. brasilianus* that ingested fragments had lower mean total weight of gut contents.
Rochman et al., 2013b, lab study	3 mm LDPE pellets (virgin or marine treated)	To determine risk from chemicals sorbed on microplastics.	Japanese medaka, *Oryzias latipes* (amphidromous, found in fresh, brackish and marine waters)	Yes	Fish bioaccumulate pollutants sorbed on microplastics and experience liver toxicity.

Rosenkranz et al., 2009, lab study	20-nm and 1000-nm carboxylated polystyrene	To determine uptake, accumulation and depuration of microplastics.	Cladoceran freshwater water flea, *Daphnia magna*	Yes	Evidence of particles crossing the gut epithelial layer. Depuration was faster for large beads.
Sanchez et al., 2014, field study	Micrometre to millimetre scale, fibres and pellets	To detect microplastic presence in wild gudgeons collected from 11 French streams.	Gudgeons, *Gobio gobio* (found in freshwater)	Yes	12% of collected fish had ingested microplastics. Fish from 7 of 11 sampled streams contained microplastics.

necrosis), and early tumour formation (Rochman et al., 2013b). The latter laboratory study used brackish conditions (water pH = 7.8, alkalinity = 100 mg/$CaCO_3$; Ohrel Jr. and Register, 2006) and an adult species of fish (*O. latipes*) that is amphidromous and migrates between both marine and freshwater habitats (Rochman et al., 2013b). This study may indicate that a stress induced response to microplastic ingestion could occur in marine and freshwater fish.

In addition to direct physical impacts from the microplastic itself, ingested plastic debris may act as a medium to concentrate and transfer chemicals and persistent, bioaccumulative, and toxic substances (PBTs), such as polychlorinated biphenyls, PCBs, to organisms (Table 3) (Teuten et al., 2007, 2009; Engler, 2012; Browne et al., 2013). Microplastics may be carriers of a) chemicals that are sorbed onto their surface from their environment (e.g., PCBs or Dichlorodiphenyldichloroethylene, DDEs), or b) chemicals that are added to the plastic (e.g., plasticizers) in the plastic production process (Mato et al., 2001; Talsness et al., 2009). There is potential for both of these types of chemicals to be transferred to organisms. Marine studies investigating transport of hydrophobic contaminants (e.g., phenanthrene) by plastic have found that contaminants sorb to plastics more easily than they do to some natural sediments and that microplastics can consequently transfer contaminants to organisms (Teuten et al., 2007). For example, plastic was shown to facilitate the transport of contaminants to the sediment-dwelling lugworm, *A. marina* and to the amphidromous Medaka fish, *O. latipes* (Teuten et al., 2007; Rochman et al., 2013b). In other experiments with *A. marina*, accumulated nonylphenol and triclosan from polyvinyl chloride (PVC) led to impaired immune functions and physiological stress and mortality, however the quantity of plastic used was relatively high (Browne et al., 2013). Experiments also show that microplastics modulate contaminant toxicity (Table 3). In experiments with *O. latipes*, a greater percentage of fish exposed to a diet with plastic and sorbed chemicals exhibited signs of liver stress, than fish exposed to a diet with plastic but without sorbed chemicals (Rochman et al., 2013b). The freshwater goby, *Pomatoschistus microps*, exposed to microplastics with sorbed pyrene, exhibited greater pyrene metabolite accumulation and altered mortality than fish exposed to pyrene alone and no microplastics (Oliveira et al., 2013). Such variety of laboratory studies provide evidence for potential effects of microplastics on organisms. However, it's important to test impacts in the field and using laboratory scenarios that mimic likely field exposures. In the absence of such data it is difficult to infer the extent of effects in natural environments where understanding of exposure is still limited.

At higher marine trophic levels, there is correlative evidence for potential transfer of adhered contaminants in seabirds, Great Shearwaters *Puffinus gravis*, and short-tailed shearwaters *Puffinus tenuirostris*, which have shown positive correlations between PCB and ingested plastics (Ryan et al., 1988; Tanaka et al., 2013). Studies with large filter feeding vertebrates, suggest that these animals might also ingest microplastics. Fossi et al. (2014) suggested that the presence of chemicals, phthalates and organochlorines, in basking sharks and fin whales might be evidence of microplastic ingestion. As contaminants are ubiquitous in the environment, without

Table 3 – Example microplastic encounters with biota in marine and freshwater organisms.

Impact	Examples from the marine literature: organism, lab/field study, reference	Examples from the freshwater literature: organism, lab/field study, reference
Ingestion	Fish, field, Lusher et al., 2013; fur seals, field, Eriksson and Burton, 2003; Lobster, field and lab, Murray and Cowie, 2011; mussel and oysters, field, Van Cauwenberghe and Janssen, 2014; planktonic invertebrates, lab, Setälä et al., 2014; zooplankton, lab, Cole et al., 2013;	Benthic and planktonic invertebrates (see Table 2), lab, Imhof et al., 2013; Fish, field, Sanchez et al., 2014
Differential ingestion of microplastic relative to natural particles	Sea cucumber, lab, Graham and Thompson, 2009	No evidence
Differential ingestion relative to organism life stage	Brachyuran larvae, lab, Cole et al., 2013	No evidence
Microplastics crossing into/out of cells or epithelia	Mussel, lab, Browne et al., 2008; Mussel and crab, lab, Farrell and Nelson, 2013; mussel, lab, von Moos et al., 2012	*Daphia*, lab, Rosenkranz et al., 2009
Retention/accumulation of microplastics in the organism, particle size-based feeding selectivity; differential rates of depuration based on particle size	Mussel, lab, Browne et al., 2008; Lobster, field and lab, Murray and Cowie, 2011; scallop, lab, Brillant and MacDonald, 2000; zooplankton, lab, Cole et al., 2013	*Daphia*, lab, Rosenkranz et al., 2009
Injury, disrupted feeding/swimming	Lugworm, lab, Besseling et al., 2012; Lugworm, lab, Browne et al., 2013; Lugworm, lab, Wright et al., 2013a; zooplankton, lab, Cole et al., 2013	No evidence
Stress, immune response, altered metabolic function, toxicity	Lugworm, lab, Browne et al., 2013; lugworm, lab, Wright et al., 2013a; Medaka fish,[b] lab, Rochman et al., 2013b; mussel, lab, von Moos et al., 2012	Medaka fish, lab, Rochman et al., 2013b
Contaminant bioaccumulation[a] (chemicals inherent in plastic)	No evidence Note: there is evidence that a plastic treatment diet has increased contaminant levels relative to the negative control diet, but no significant evidence of transfer to the organism (Rochman et al., 2013b)	No evidence Note: there is evidence that a plastic treatment diet has increased contaminant levels relative to the negative control diet, but no significant evidence of transfer to the organism (Rochman et al., 2013b)
Tumour formation	Medaka fish, lab, Rochman et al., 2013b	Medaka fish, lab, Rochman et al., 2013b
Altered mortality	Lugworm, lab, Besseling et al., 2012 (suggested based on microplastic presence in dead organisms, but not a significant evidence)	No evidence
Adsorption of chemicals, transfer of chemicals to organism	Lugworm, lab, Browne et al., 2013; Medaka fish, lab, Rochman et al., 2013b Seabird, field, Tanaka et al., 2013 (suggested by correlation)	Medaka fish, lab, Rochman et al., 2013b
Contaminant bioaccumulation[a] (chemicals sorbed on plastic)	Lugworm, lab, Besseling et al., 2012; Lugworm, lab, Browne et al., 2013; Medaka fish, lab, Rochman et al., 2013b	Medaka fish, lab, Rochman et al., 2013b
Disrupted feeding/swimming	Lugworm, lab, Browne et al., 2013;	No evidence
Modulation of contaminant toxicity -> Stress, immune response, altered metabolic function, toxicity	Lugworm, lab, Browne et al., 2013; Medaka fish, lab, Rochman et al., 2013b	Goby fish, lab, Oliveira et al., 2013 Medaka fish, lab, Rochman et al., 2013b
Modulation of contaminant toxicity -> Altered mortality	Lugworm, lab, Browne et al., 2013;	Goby fish, lab, Oliveira et al., 2013;
Dietary energy gain/nutritional condition	Lugworm, lab, Besseling et al., 2012; Lugworm, lab, Wright et al., 2013a (suggested impact)	No evidence

Trophic food-web transfer	Mussel and crab, lab, Farrell and Nelson, 2013; Zooplankton, lab, Setälä et al., 2014	No evidence
Substrate for rafting communities	bacteria, diatoms, dinoflagellates, coccolithophores, and radiolarians, field, Carson et al., 2013 pelagic insect, field, Goldstein et al., 2012	No evidence

[a] The term "bioaccumulation" is defined as "The biological sequestering of a substance at a higher concentration than that at which it occurs in the surrounding environment or medium" (U.S. Geological Survey, 2007).
[b] Rochman et al. (2013b) used an adult species of fish (*Oryzias latipes*) that is amphidromous and migrates between both marine and freshwater habitats.

evidence of plastic ingestion, it may be difficult to identify causality in cases of contaminant presence in animal tissues. In freshwater systems no studies exist with evidence of microplastic contaminant transfer to birds.

Marine field studies confirm the presence of sorbed environmental contaminants on microplastics (Mato et al., 2001; Rochman et al., 2013b), and laboratory evidence suggests that sorbed contaminants can be transferred to marine fish and invertebrates (Besseling et al., 2012; Browne et al., 2013; Rochman et al., 2013b). Since chemicals are present in water entering treatment plants, in treated effluent, and in drinking water (Morasch et al., 2010; Brausch and Rand, 2011), there could be concern that freshwater systems close to industrial and population centers may have both a greater microplastic presence, and greater concentrations of chemicals and contaminants, and that biota in these regions may therefore experience greater exposure. Such concerns are valid, but more research is needed, as interactions (chemical sorption/desorption to plastic and transfer to biota) are complex and not yet fully predictable. Chemical transfer depends on the plastic, the contaminant, the surrounding environment, and the organism that ingests the plastic. For example, the sorption capacity varies between plastics, e.g., polyethylene sorbs greater concentrations of contaminants than other polymers (Rochman et al., 2013c), and the release of contaminants from plastics is facilitated by increased temperature and low pH equivalent, resembling conditions in a warm-blooded animal (Bakir et al., 2014a).

4.3. *Potential for wider environmental impacts of microplastics*

In addition to having direct interactions with organisms, microplastics in aquatic habitats may have wider impacts by interacting with the abiotic environment or by having indirect effects on biotic communities or ecosystems (Fig. 1). A potential physically driven transport potential is a regionalised concentration of chemicals in the environment as microplastics respond to transport by physical forces (Bakir et al., 2014b). Recent research has found that sorption and desorption rates of chemicals are dominated by the ambient concentrations of contaminants and residence time of particles. For example, it is suggested microplastics are more likely to sorb contaminants in estuaries where there are higher reported concentrations of contaminants and long particle residence and potential storage in sediments (Bakir et al., 2014b).

Other than affecting the distribution of chemicals in the environment, microplastics may directly or indirectly affect abiotic qualities of the environment. Authors suggest microplastic accumulation in pelagic and benthic habitats might alter light penetration into the water column or change sediment characteristics, and in turn these changes could affect biogeochemical cycles (Arthur and Baker, 2011). Physical and chemical properties of sediment, which are important to an ecosystem include grain size, pore size, and sediment binding capacity to chemicals (Simpson et al., 2005). While no evidence yet exists for abiotic effects of microplastic in marine or freshwater systems, there is evidence of microplastic accumulation in marine sediments, and suggestions that its

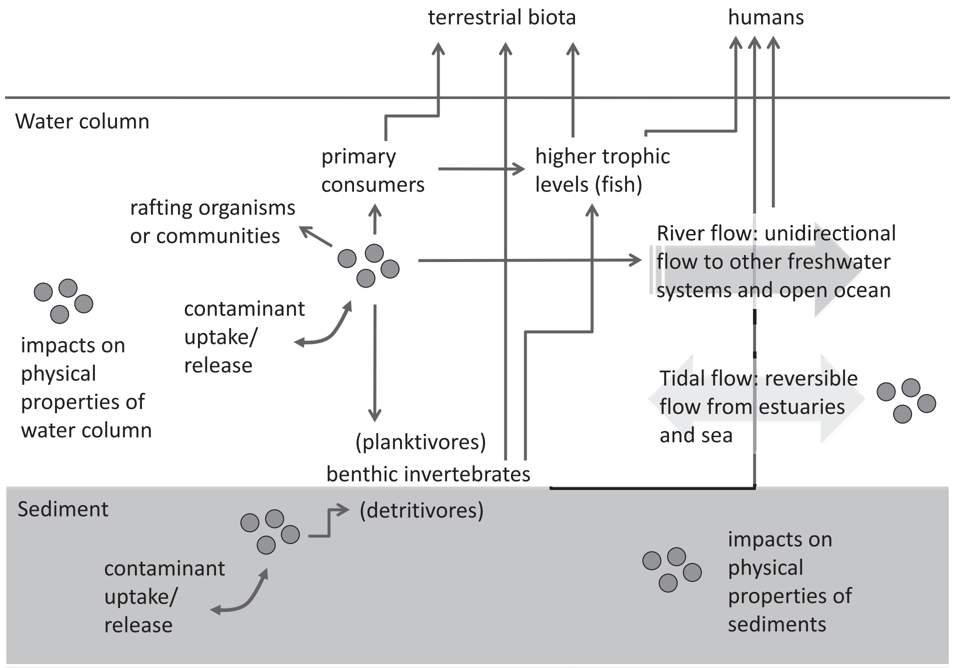

Fig. 1 – Diagram showing the potential transfer pathways of microplastics in freshwater systems.

presence may alter the behaviour of benthic ecosystem engineers. Claessens et al. (2011) using sediment cores estimated significant increases in microplastic accumulation in beach sediment from the Belgian coast over an estimated 16 years. Wright et al. (2013a) suggested that if there was 6.34% microplastics by volume in sediments of the Wadden Sea, there could be 130 m^2 less sediment being reworked by the lugworm *A. marina* annually. Wright et al. (2013a) speculate the potential for cascading effects from microplastic ingestion by marine benthic species. Similarly, accumulation of microplastics in freshwater sediments and ingestion by freshwater benthic fauna might have cascading effects with trophic and ecosystem consequences (e.g., impacts on community structure). Microplastic ingestion by benthic freshwater invertebrates could impact sediment bioturbation, or since benthic biota form a large component of some fish diets (e.g., contributing up to 90% of fish prey biomass in some cases, Schindler and Scheuerell, 2002), microplastic impacts on benthic organisms could affect higher trophic levels (e.g., trophic energy transfer or trophic interactions). Similar impacts may also occur in pelagic habitats where microplastics can reach densities higher than naturally occurring planktonic organisms (Lechner et al., 2014).

The effects of microplastic may also transfer between habitats. For example, in marine systems, transfer of microplastics from marine to terrestrial habitats is documented in the sub Antarctic islands, where seals and sea lions consumed fish suspected of containing microplastics, and deposited scats on land (McMahon et al., 1999; Eriksson and Burton, 2003). Microplastics in freshwater may have carry-over effects to terrestrial systems, as many freshwater organisms are prey to terrestrial insects, amphibians, reptiles, and birds (Polis et al., 1997). Some forest birds receive up to 98% of their resources from aquatic prey (Nakano and Murakami, 2001). Potential exists for microplastic transfer across habitats via animal migrations, much the way anadromous fish transfer marine nutrients to freshwater systems (Polis et al., 1997). Other habitat related effects of microplastics includes their role as a substrate for egg laying organisms or as habitat for encrusting organisms, rafting communities and microbial communities (Gregory, 1978; Goldstein et al., 2012; Carson et al., 2013; Zettler et al., 2013). Microplastics serve as novel ecological habitats for microbes and may provide substrate for opportunistic pathogens (Zettler et al., 2013).

Differential impacts of ingestion by life-stage have not been examined. However, across habitats, early life stages are considered to have heightened sensitivity to environmental conditions; environmental impacts on early life stages can transfer to later life stages, leading to reduced developmental potential, fitness, and survivorship (Pechenik, 2006). A valuable research avenue may be testing the potential for microplastics to cause differential impacts by life-stage of aquatic animals. For instance, is it possible that earlier fish stages (i.e., embryos) are more sensitive to microplastic exposure than later stages (i.e., juvenile fish), and exposure of embryos in rivers beds to adsorbed microplastic contaminants could have consequences for juvenile growth rates or survival. Such scenarios are observed for other contaminants; exposure of pink salmon, *Oncorhynchus gorbuscha*, embryos to crude oil led to carry-over effects in growth of juveniles and in survival of the marine stages (Heintz et al., 2000). Since various terrestrial and aquatic vertebrates and invertebrates have early life stages that develop in freshwater systems, it may be important to study the potential for early life stages to interact with microplastics and/or their associated contaminants.

The potential routes in which microplastics may interact with freshwater environments and ecosystems are varied. As the presence of microplastics in freshwater systems begins to be documented, investigations on encounters and impacts on biotic and abiotic qualities of the ecosystem will be a necessary next step to determine potential for any wider environmental consequences.

4.4. *Suggested research on potential impacts on humans*

The impacts of microplastics (from marine or freshwaters) on humans are not well documented. In the area of food safety for example, due to limited information, literature reviews have been unable to assess the consequences of microplastics presence (Hollman et al., 2013). Microplastics are however, being documented in the tissues of commercially grown marine bivalves; concentrations of 0.36 ± 0.07SD and 0.47 ± 0.16SD particles per gram of soft tissue (wet weight) respectively were detected in mussel, *M. edulis*, acquired from a mussel farm in Germany and from the oyster, *Crassostrea gigas*, bought in a supermarket and originally reared in the Atlantic Ocean (Van Cauwenberghe and Janssen, 2014). Therefore it is important to investigate whether microplastics could have the potential to have either direct or indirect effects on human health or on economies. Specific research might investigate effects on: 1) resources directly used by humans (drinking water, bathing water, or food resources); 2) logistics of water use; and 3) ecosystem services. Research avenues might consider the following:

- Presence of microplastic.
- Transfer of chemicals to food; either chemicals inherent in microplastics or chemicals sorbed and transported by microplastics.
- Interactions of fishery/aquaculture species with microplastics and whether these interactions affect the edibility or marketability of fish/aquaculture species.
- Whether application of sewage sludge to terrestrial systems for agricultural reasons may lead to transfer of microplastics and/or chemicals to soil used in growing food. Indeed, even after secondary or tertiary wastewater treatments, effluents can contain particle loads comparable to sewage receiving preliminary treatment (Puig-Bargués et al., 2005). Therefore use of effluents in agricultural irrigation may contribute to the transfer of microplastic particles.
- Economic considerations, such as whether microplastic presence in aquaculture species could lead to loss in revenues, or the extent of costs associated with clean-up efforts.

In the water treatment literature, clogging is widely acknowledged as a major problem in screening processes where small particles may reduce the capacity of filters used in potable and wastewater treatment (Ljunggren, 2006). Clogging also poses problems when agricultural microirrigation systems use effluents (Puig-Bargués et al., 2005). At present, however, it is not clear how microplastics present an additional challenge in comparison with natural particulates. Microplastics may only constitute a small proportion of particulates so their contribution to water treatment problems may be small.

The interactions listed above are not fully known and warrant further investigation. An awareness of the extent and quantity of microplastic present in water systems will be necessary: 1) in planning new wastewater treatment plants; and 2) development of policies aimed at managing pollution and maintaining valuable ecosystems services (e.g. the European Commission's Water Framework Directive, possible legislation on the use of microbeads as abrasives in cosmetics) would benefit from greater knowledge of the role of microplastics in freshwater systems.

5. Policy development

Greater knowledge of extent and impacts of microplastic in marine waters versus freshwaters is reflected in more policy and management interest for marine systems, though even these are still in their infancy. Policy initiatives for marine litter aim at: 1) understanding presence and impacts, and 2) preventing further inputs or reducing total amounts in the environment. Examples of nation's efforts to deal with marine litter include the US Interagency Marine Debris Coordinating Committee (IMDCC), which supports the US national/international marine debris activities, and “recommends research priorities, monitoring techniques, educational programs, and regulatory action” (EPA, 2013). The European Commission's Marine Strategy Framework Directive (MSFD) has designated a Technical Subgroup on Marine Litter to provide “scientific and technical background for the implementation of MSFD requirements”, which include identification of research needs, development of monitoring protocols, preventing litter inputs and reducing litter in the marine environment. The MSFD “litter” designation includes microplastics and acknowledges a limitation in “knowledge of the accumulation, sources, sinks … environmental impacts … temporal and spatial patterns and potential physical and chemical impacts” of microplastics (Galgani et al., 2010, 2013).

Microplastic presence in freshwaters has only recently received attention, and policy initiatives are less developed than for marine systems, but could benefit from similar initiatives to those of Europe's MSFD and the activities of the US IMDCC. Authors investigating microplastics in freshwaters have noted that microplastic debris, while abundant in rivers and lakes, is not subject to regulation. In the study of US LA basin rivers, microplastic sized particles (<5 mm) were the most numerically abundant plastic in samples, but their size range did not subject them to regulation (Moore et al., 2011). Researchers suspected that the high level of microplastic contamination, characterized by a predominance of fragments from household origin, in the remote Lake Hovsgol of Mongolia, resulted from a lack of modern waste management and enforcement (Free et al., 2014). These authors note the need for policy development, as well as for legislation and enforcement, in order to address microplastic contamination in freshwaters (Moore et al., 2011; Free et al., 2014), and to help deal with the potential role of freshwater systems as pathways of transport of microplastics from land-based sources to oceans (Lechner et al., 2014).

6. Conclusions, next steps, and opportunities

Microplastics are ubiquitous in marine systems where they interact with a variety of organisms. Early investigations suggest that microplastic presence and interactions in freshwater systems are equally far reaching. Microplastics are being detected in Asia (Free et al., 2014), the EU (Faure et al., 2012; Imhof et al., 2013; Lechner et al., 2014; Wagner et al., 2014), and North America (Moore et al., 2011; Eriksen et al., 2013; Castañeda et al., 2014; Hoellein et al., 2014; Zbyszewski and Corcoran, 2011; Zbyszewski et al., 2014). They are found in remote and protected areas (e.g., Lake Hovsgol, Mongolia) and in large enough quantities to outnumber natural particles (e.g., Danube river, Austria) (Free et al., 2014; Lechner et al., 2014). They are also speculated to be a large contribution of land-based litter to oceans, e.g., Lechner et al.'s (2014) estimate that 1533 tonnes per year of plastic litter enter the Black sea from the Danube. Early studies suggest both freshwater invertebrates and fish ingest microplastics, with ingestion leading to physical effects that include physiological stress responses and even signs of tumour formation (Imhof et al., 2013; Oliveira et al., 2013; Rochman et al., 2013b). Reviewing the marine and freshwater literature we reach similar conclusions, in assessment of microplastic spread and impacts on freshwater systems, as Wagner et al.'s (2014) initial review of microplastics in freshwater systems. As research on microplastics in freshwaters is in its infancy, only arising in the last five years, many questions remain and further research is needed to: 1) develop optimal methodology for monitoring microplastics in freshwater systems; 2) quantify all aspects driving presence, abundance and distribution of microplastics in the environment; 3) understand the degradation behaviour including particle lifetimes and ultimate fate in freshwater; 4) assess the potential of rivers to be a source of microplastic to the oceans; 5) assess and understand microplastic interactions with biota; 6) assess microplastic impacts on ecosystem services; and 7) evaluate the consequences of microplastic for humans.

Globally freshwater is a dwindling natural resource and is in a fragile state. Available supplies are subject to competing pressures and impacts such as pollution threaten freshwater's uses and ecological quality. As demand continues to rise, there is a clear need for quality assessment, integrated resource management, and improved global water quality (UNEP, 2007). In the United States, 44% of assessed rivers and streams and 64% of assessed lakes and reservoirs are considered impaired (US EPA, 2009). As nearly 50% of Europe's surface water is of poor ecological quality and 40% is of unknown chemical status (Werner, 2012), the process of identifying, monitoring, and dealing with water pollution will be essential. The EU Water Framework Directive calls for control of pollutants in water bodies, including materials in suspension (Directive, 2000/60/EC). US states continue to improve water monitoring programs with the intent of meeting the Clean Water Act goals of restoring and maintaining the chemical, physical, and biological integrity of the nation's waters (Copeland, 2012). Such initiatives demonstrate the interest of nations in managing and improving quality of freshwater resources.

Attention and research similar to that recommended for microplastics in marine systems is needed for freshwater systems. Progress on this issue requires support from a solid scientific knowledge base and would benefit from cooperative efforts by the relevant statutory bodies and legislative frameworks at the international, national, and regional levels (e.g. the European Commission's WFD and MSFD, and the UK's Environmental Agency, Department for Environment Food and Rural Affairs, and the Centre for Environment, Fisheries, and Aquaculture Science). Indeed, a solid knowledge base is critical for policy makers (EEA, 2012). Concerted efforts on all fronts, including survey, monitoring, research, and policy, will be required to better understand any emergent threats posed by microplastics in freshwater systems and to develop appropriate, informed strategies for managing them.

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