Environmental Pollution 256 (2020) 113445

Contents lists available at ScienceDirect

Environmental Pollution

journal homepage: www.elsevier.com/locate/envpol

Advances and challenges of microplastic pollution in freshwater ecosystems: A UK perspective☆

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ARTICLE INFO

Article history:
Received 27 February 2019
Received in revised form
27 September 2019
Accepted 19 October 2019
Available online 21 October 2019

Keywords:
Microplastics
Freshwater
Pollution
UK

ABSTRACT

Microplastics have been increasingly documented in freshwater ecosystems in recent years, and growing concerns have been raised about their potential environmental health risks. To assess the current state of knowledge, with a focus on the UK, a literature review of existing freshwater microplastics studies was conducted. Sampling and analytical methodologies currently used to detect, characterise and quantify microplastics were assessed and microplastic types, sources, occurrence, transport and fate, and microplastic-biota interactions in the UK's freshwater environments were examined. Just 32% of published microplastics studies in the UK have focused on freshwater environments. These papers cover microplastic contamination of sediments, water and biota via a range of methods, rendering comparisons difficult. However, secondary microplastics are the most common type, and there are point (e.g. effluent) and diffuse (non-point, e.g. sludge) sources. Microplastic transport over a range of spatial scales and with different residence times will be influenced by particle characteristics, external forces (e.g. flow regimes), physical site characteristics (e.g. bottom topography), the degree of biofouling, and anthropogenic activity (e.g. dam release), however, there is a lack of data on this. It is predicted that impacts on biota will mirror that of the marine environment. There are many important gaps in current knowledge; field data on the transport of microplastics from diffuse sources are less available, especially in England. We provide recommendations for future research to further our understanding of microplastics in the environment and their impacts on freshwater biota in the UK.

1. Introduction

Due to their lightweight, durable, strong, corrosion-resistant and low electrical and thermal conductivity, plastics have become universal materials (Rillig, 2012; Lambert et al., 2014; Wagner et al., 2014). Although synthetic polymers were first developed in 1907, everyday plastic items have been mass produced since the 1940s and 1950s (Thompson et al., 2009). At that time, less than one million tons were produced annually in the world, but there has since been a steady increasing trend in plastic production and use. Such that in 2016 plastic production reached 60 million tons in the European Union (EU) and 335 million tons worldwide (Plastics Europe, 2017). A significant proportion of this production volume (mostly non-biodegradable) may reach the environment. Consequently, plastic debris is accumulating in both terrestrial and aquatic (marine and freshwater) systems globally (Lambert et al., 2014; Ivleva et al., 2017) and, as it is very slow to degrade, remains long after the plastic products' life-span (Lithner et al., 2011; Wright and Kelly, 2017). Plastic debris has even been distributed to remote regions such as the Arctic (trapped inside sea ice) and the deep sea (Obbard et al., 2014; Amelineau et al., 2016; Bergmann et al., 2017; Kanhai et al., 2018; Peeken et al., 2018).

Plastic debris enters natural ecosystems in all shapes and sizes, but, until recently, it is the larger plastic items which have caused serious public concern. Many researchers have reported problems, such as entangled seabirds (Moser and Lee, 1992; Buxton et al., 2013), and lethality in sea turtles following ingestion (Wilcox et al., 2018). The larger pieces of plastic, however, are progressively degraded until 'microplastics' are emitted, defined as having a maximum dimension <1 mm, as recommended by Hartmann et al. (2019). In addition to these microplastics (being called

☆ This paper has been recommended for acceptance by Maria Cristina Fossi.

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https://doi.org/10.1016/j.envpol.2019.113445
0269-7491/© 2019 Published by Elsevier Ltd.

secondary microplastics), another type of microplastic, beads and pellets (called primary microplastics), are manufactured in order to be used, for example, as exfoliants in personal care products or as blast media in industrial cleaning. Unlike larger plastic debris in the environment, microplastics are relatively invisible and cannot be removed from habitats for recycling.

Given that microplastics are recognized as ubiquitous anthropogenic pollutants in environments worldwide, their potential environmental health impacts are of increasing concern. The potential hazards to organisms include inflammation of the digestive system (Von Moos et al., 2012), reduced nutrient uptake (Hurley et al., 2017) and reduced growth and reproduction (Sussarellu et al., 2016). There is considerable evidence of the omnipresence of microplastics in marine and freshwater ecosystems across the world. Until recently research has mainly focused on marine environments, with relatively few studies conducted in the freshwater habitats. A literature search on the database Web of Science returns 2708 publications (Topic = microplastic OR microbead), of which a subset of 264 papers (9.7%) focused on the freshwater environment. Not surprisingly, there have been calls for more research into the risks of microplastics in freshwater ecosystems (Anderson et al., 2017; Hurley et al., 2017). Increasing concern over the detrimental impacts of microplastics has promoted some countries, such as the UK, US and Canada to act. These campaigns have mainly focused on banning the use of microbeads from various items, for example, cosmetic and personal care products.

To provide a better understanding of microplastic pollution in freshwater habitats in the UK, the aims of this review are to: (1) systematically review the state of sampling and analytical methodologies currently used to detect, characterise and quantify microplastics in the UK's freshwater environments; (2) assess the types of MPs prevalent in the UK's freshwater environments and examine their transportation, fate and biological effects; and (3) identify current knowledge gaps and provide recommendations for future research to further our understanding of microplastics in the environment and their impact on freshwater biota in the UK.

2. Methodology

2.1. Search strategy

A two-tier approach was adopted for this review: first, in order to gain a general overview of the relative scientific activity in freshwater habitats compared to marine, a simple search was undertaken in one database, using restrictive keywords. Following this initial scoping exercise, a focused, exhaustive search was performed to gain a representative synthesis of the state of knowledge concerning microplastic pollution in the UK's freshwater environments. This used a wide range of databases and keywords to retrieve as many publications as possible with respect to microplastic pollution in the UK's freshwater environment. When specific data from UK's habitats were not available, information from studies in other geographical locations were included.

Covering more than 70% of the Earth's surface, marine and freshwater ecosystems are the two main categories of aquatic environments. Freshwater ecosystems include rivers, streams, lakes and ponds, channels, reservoirs and wetlands. In addition to the two main aquatic environments, there is another minor type of aquatic ecosystem, estuaries; the transitionary zone between freshwater and marine ecosystems, generally found where rivers meet the sea. Therefore, in this review, aquatic environments have been considered in three main categories: freshwater, marine, and estuary habitats.

2.2. Search terms

Search terms were selected to retrieve as many articles as possible. They were categorized into relevant subjects, by area (i.e. habitat) and intervention (i.e. microplastics). This was further refined to be UK-specific:

General search: *microplastic(s), microbead(s); freshwater, surface water, river, lake, reservoir, stream, brook, pond, estuary (estuarine), wetland, sewage, marine, ocean, and sea.*

Focused search: *microplastic(s), microbead(s); freshwater, surface water, river, lake, reservoir, stream, brook, pond, estuary (estuarine), wetland, sewage, marine, ocean, sea; United Kingdom (UK), Britain, England, Scotland, Wales, and Northern Ireland.*

2.3. Databases

For the general search, the database ISI Web of Science was exclusively used. The focused search used the following databases:

- ISI Web of Science, Scopus, SpringerLink, EI Compendex, Elsevier-ScienceDirect Online and Google Scholar.

The nature of the search was by topic (title, abstract and keywords) and no document type restrictions were enforced at this stage. The search was restricted to articles published up to November 2018.

2.4. Inclusion criteria

Inclusion criteria were 'relevance of subject(s)/discipline', i.e. environmental sciences, water resources, and 'type of study'; only peer-reviewed primary studies with field data were included for the general search to indicate trends. For the focused search, this was expanded to include experimental and exposure laboratory studies for a comprehensive overview. Article types such as reviews, editorial materials, news items, book chapters and notes were removed. Many publications were removed as they were not relevant because of their discipline, e.g., polymer science, cell biology, electrochemistry, educational scientific disciplines, toxicology and telecommunications. Articles were screened based on their title, abstract and then full text, depending on how clearly they conformed to the inclusion criteria.

3. Results and discussion

3.1. Description of studies

The number of studies on microplastic pollution in all three aquatic ecosystems (fresh, marine and estuarine) has surged, especially since 2012. Although a total of 716 papers were found in the general search, only 396 papers of them were publications reporting field data. These 396 articles were used to scope the relative scientific activity in aquatic microplastic contamination. Approximately 77.5% of microplastics research (307 papers) was concerned with marine ecosystems, with less attention having been made to the other two aquatic environments; only 16.2% (64 papers) and 7.6% (30 papers) for freshwaters and estuaries, respectively (Fig. 1c and d). While research activity in all three habitats has increased worldwide, the number of annual publications differs substantially. After 2012, there was an average growth of about 23, 5 and 2 publications per year for marine, freshwater and estuarine environments, respectively.

The focused search strategy resulted in a total of 50 papers concerning microplastic research in the UK, most of which were recently conducted in the last 5 years (76%, 38 papers). As found in

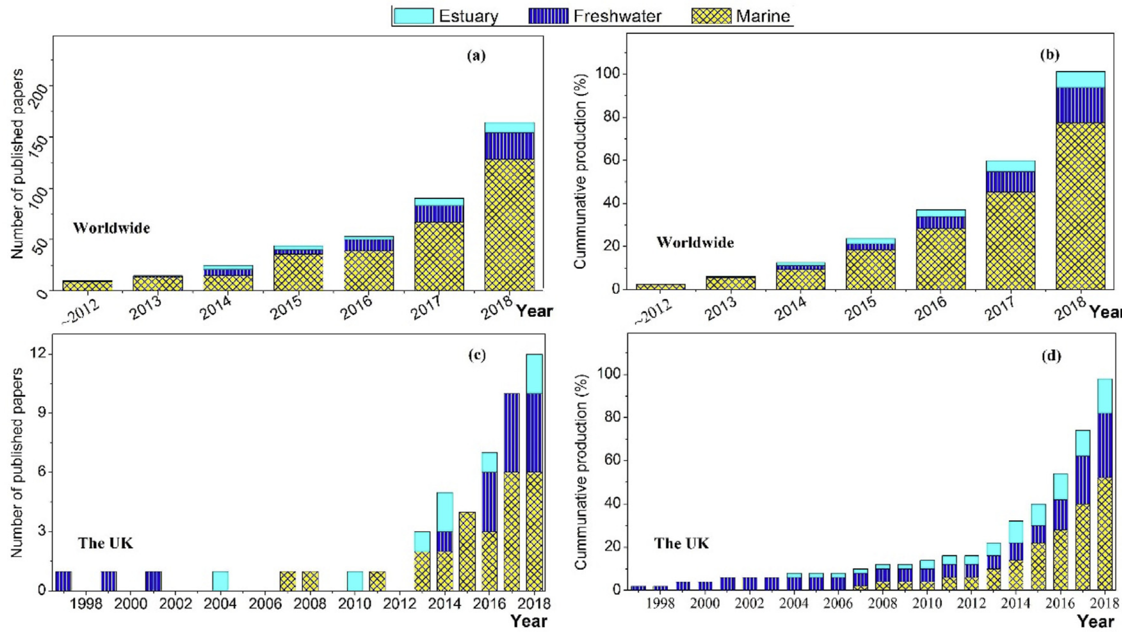

Fig. 1. Comparison between microplastic pollution publications on marine, freshwater and estuarine environments. (a) Total number of scientific publications in aquatic systems across the world each year; (b) Cumulative number of scientific publications (%) in aquatic systems across the world; (c) Total number of scientific publications on the UK's aquatic systems each year; (d) Cumulative number of scientific publications (%) in the UK's aquatic systems.

the general search, scientific efforts are less focused on microplastics in freshwater systems, with studies comprising 32% (16 papers), 52% (26 papers) and 16% (8 papers) for freshwater, marine and estuarine environments, respectively (Fig. 1a and b).

For the 16 freshwater studies, only 8 papers, which were mainly performed in England, involved field sampling (Fig. 2, Table 1), while the other 8 papers focused on related topics, such as the release of microplastics from domestic washing and wastewater treatment works, and microplastics in riverine litter. Although freshwater habitats cover only 12% of land in the UK, they provide some of the most valuable resources for British people and wildlife. These include water for drinking, agricultural and industrial production purposes, water abstraction, transport, and recreational activities. The UK's freshwater economic value based on its services (fish capture, water abstraction, peat extraction, pollution removal and recreation) could be in excess of £39.5 billion annually (Office for National Statistics, 2017). Given that there is no microplastics field data available for many freshwater habitats, especially in Scotland, Wales and Northern Ireland (Fig. 2, Table 1), more research is required.

The UK's freshwater systems consist mostly of wetlands and open waters, which comprise standing waters (such as lakes, reservoirs, canals, and ponds) and flowing waters (such as rivers and streams). It is worth noting that the observed habitat categories were disproportionally distributed in the UK's freshwater microplastic research. Most of the studies were conducted in rivers (12 papers), one study was performed in a lake, and the other three studies are indirectly related to microplastic pollution in freshwater environments (Fig. 3b). Thus, several freshwater habitats in the UK, such as wetlands, lakes, reservoirs, canals and ponds currently lack research focus regarding possible microplastic contamination. Groundwater is a vital component of ecologically important flows to some freshwaters, such as rivers and wetland, across the world. Moreover, groundwater accounts for around 30% of the drinking water supply in England (Drinking Water, 2017), yet no studies were found which address potential microplastic pollution in UK groundwater.

Although freshwater habitats are less extensive than seawater habitats, they are important focuses of biodiversity which support a broad range of plants and animals (Dudgeon et al., 2006). In the UK freshwater microplastic studies, five papers were found to focus on biota impact/interaction, five papers considered sediments, three papers were related to riverine litter, two papers referred to wastewater treatment plants, one paper analysed laundry emissions, and one paper focused on surface water (Fig. 3a). This highlights a need to expand knowledge of freshwater microplastics, especially in unsampled habitats in the UK.

Eighteen of the 26 identified marine studies (i.e. 69%) are concerned with microplastic-biota interactions, whereas from the 16 recorded freshwater studies, only 5 papers focused on similar interactions, and 2 of the 8 estuarine studies considered this interaction (Fig. 3c). Fig. 3d displays the number of UK aquatic studies classified by biotic group. Attention has been paid to a relatively wide set of marine species, including fish, decapods, amphipods, molluscs and mammals. However, to-date, just a few studies have assessed impacts on freshwater fauna, including fish (roach), oligochaeta (worm), Ephemeroptera (may fly), trichopteran (caddis fly) and anthoathecata (hydra). Compared with the abundance of different species in UK freshwater ecosystems (Harding and Bell, 2001), the organisms used in these studies are still very limited.

3.2. Freshwater microplastics sampling and detection techniques

Research on the distribution, impact, and fate of microplastic debris is dependent on the use of appropriate analytical methods. Microplastics are heterogeneously distributed in freshwater systems. Large variations in sediment concentrations ranging between 56 and 2543 particles kg^{-1} were measured in a major urban waterbody, Irwell, Manchester, UK (Hurley et al., 2017). It is therefore difficult but essential to collect representative samples (water, sediment and biota) to assess the diversity and distribution of microplastics in freshwater environments. It is clear that the

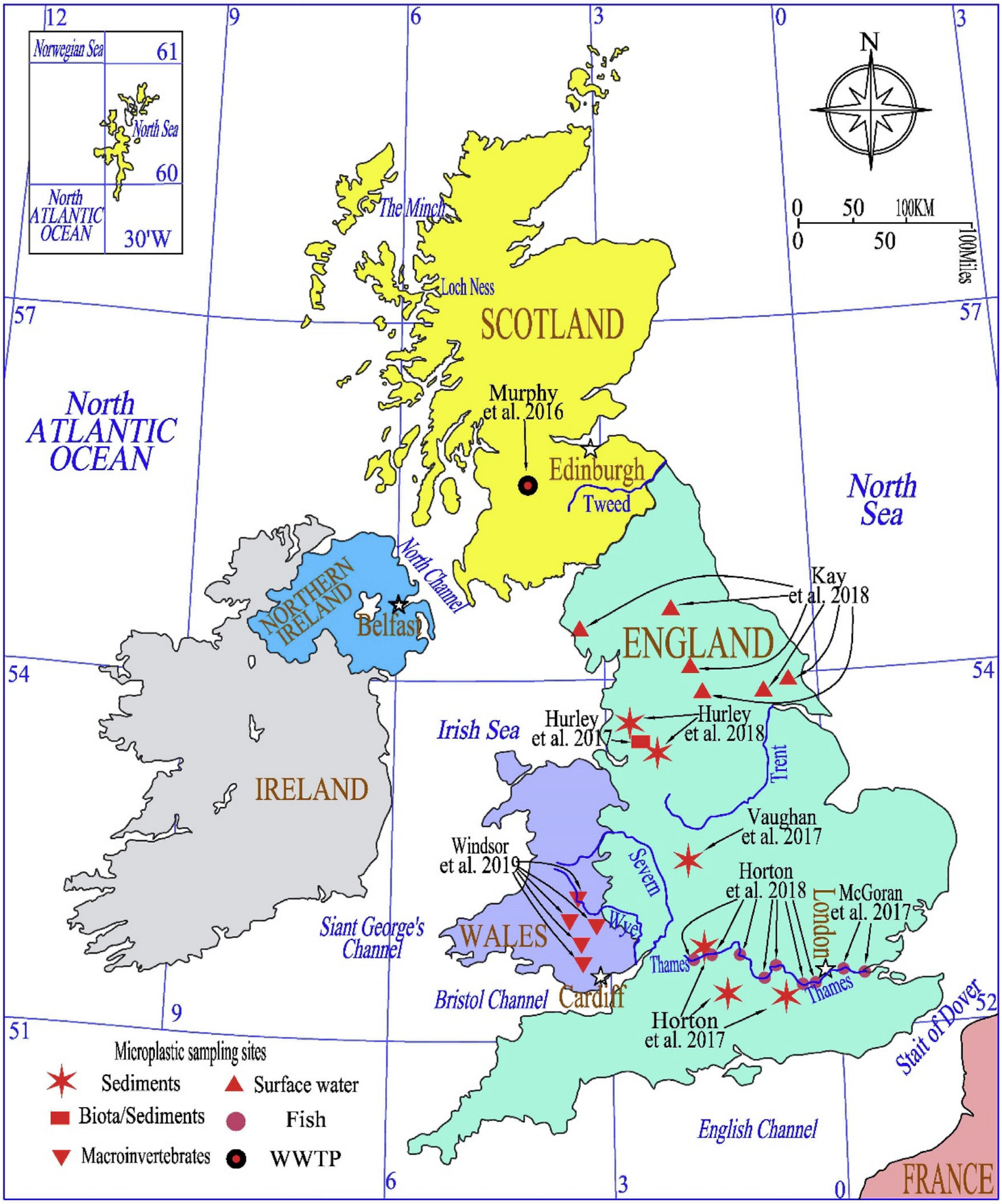

Fig. 2. A map showing the location of microplastic field studies performed in UK freshwater environments. WWTP = wastewater treatment plant.

parameter of particle size is of huge importance for microplastic research in aquatic environment. For instance, the sizes of microplastics may play a vital role in regulating the interactions between microplastics and freshwater communities. Moreover, the minimum size of the plastic particles detected could be largely decided by the mesh size. Therefore, harsh sample processing is not recommended to avoid the loss of particles in the lower size range, which may result in underestimating the microplastic concentrations. Furthermore, visual inspection of microplastics may be influenced by the high load of naturally derived particles in the sample matrix.

3.3. Sampling microplastics

Microplastics have been found in virtually all forms of freshwater habitats from the surface water (e.g. Kay et al., 2018) to bottom sediments (e.g. Horton et al., 2017; Vaughan et al., 2017; Hurley et al., 2018) to aquatic organisms (e.g. Hurley et al., 2017; McGoran et al., 2017; Horton et al., 2018; Windsor et al., 2019). It is hence important to monitor them in both abiotic and biotic matrixes.

The most common methods to sample surface water can be categorized into two distinct classes, namely bulk sampling and volume-reduced sampling that is usually equipped with net-based

Table 1
Studies on microplastic contamination in the UK's freshwater habitats. 'Not performed' means it was explicitly stated that purification was not performed; 'Not mentioned' means the purification process cannot be found in the paper; 'MP' = microplastics.

Location	Water body type	Sample type	Sample collection description	Separation solution/MP extraction	Purification	Identification	Reference
The upper Mersey and Irwel catchments	River	Sediment	Cylinder resuspension technique to a depth of 100 mm	Three density solutions were used: NaCl solution (1.025 g cm^{-3}), NaCl solution (1.2 g cm^{-3}) and NaI solution (1.8 g cm^{-3})	Not performed	Visual identification + FTIR spectrometer	Hurley et al. (2018)
Edgbaston Pool, 3 km from the centre of Birmingham	Lake	Sediment	HTH gravity corer to a depth of 100 mm	A combination of size- and density separation, a 1 mm and a 500 mm sieve. Density separated using water.	Not performed	Binocular microscope (x40)	Vaughan et al. (2017)
Three tributaries of the Thames	River	Sediment	stainless steel scoop to a depth of 100 mm	ZnCl2 solution (1.7–1.8 kg L−1)	Not mentioned	Visual inspection using a binocular light microscope + Raman spectroscopy (RS)	Horton et al. (2017)
Up- and downstream of six wastewater treatment plants (WWTPs) selected across the north of England	River	Surface water	300-µm mesh size net	Not mentioned	Not mentioned	Visual identification under a stereomicroscope	Kay et al. (2018)
The River Clyde, Glasgow	River	Influent and effluent of a WWTP	Steel buckets (10L)	Not mentioned	Not mentioned	Fourier-transform infrared spectroscopy (FT-IR)	Murphy et al. (2016)
Plymouth, south coast of Devon, England	(the laundering of clothes)	laundering wastewater	Nylon CellMicroSieve™ (Fisher Scientific) with 25 µm pores	Not mentioned	Not mentioned	FT-IR Microscope	Napper and Thompson (2016)
The Salford Quays basin, located on the edge of Manchester city centre	River	Sediments, *Tubifex* worms	Bottom sediments: using a UWITEC gravity corer. Tubifex worms were abundant in the surface sediment layer.	Three extracts were used: 1.025 g cm−3 NaCl, 1.2 g cm−3 NaCl, and 1.8 g cm−3 NaI.	10% KOH at 60 °C.	Visual identification using a microscope+the hot needle test + FT-IR spectroscopy	Hurley et al. (2017)
The South Wales valleys	River	Heptageniidae, Baetidae and Hydropsychidae	Intensive kick sampling and hand-searching	Hypersaline solution (1.2 g cm−3).	15% H_2O_2 solution	A tandem microscopy technique: Light microscopy used initially + light microscopy, bright- and dark-field spectroscopy	Windsor et al. (2019)
The main body of the River Thames between 36 km and 239 km from the source	River	*Rutilus rutilus* (roach)	Electrofishing techniques	Not mentioned	Not performed	A binocular microscope + RS	Horton et al. (2018)
The River Thames	River	*Platichthys flesus* (European flounder) and *Osmerus eperlanus* (European smelt)	Fyke nets	Not mentioned	Not mentioned	Dissecting microscope + FT-IR Microscope	McGoran et al. (2017)
Paisley, Scotland	/ (Laboratory experiment)	*Hydra attenuate* (Freshwater cnidarian)	Cultured in glass bowls	Not mentioned	Not mentioned	FT-IR Microscope	Murphy and Quinn (2018)

devices (neuston or plankton nets) or a sieve requiring no specialized equipment. Huge volumes of water can be filtered using the latter method and is recommended for water sampling in lakes and large rivers, but it would not include microplastics smaller than the net mesh size; by contrast, the former method can sample all size ranges of microplastics and should be implemented to complement the latter. The most popular types of reported mesh sizes globally are 300 µm (e.g. Fischer et al., 2016; Sighicelli et al., 2018; Kay et al., 2018) and 333 µm (e.g. Eriksen et al., 2013; Su et al., 2016; Anderson et al., 2017), while other mesh sizes such as 112 µm (Zhang et al., 2015), 153 µm (Estahbanati and Fahrenfeld, 2016) and 500 µm (Lechner et al., 2014) were also used. Samples collected along the water column have been reported for freshwater habitats, but only infrequently in comparison to samples collected from surface waters (Pico et al., 2019). Studies that take a bulk sampling approach usually give the results in terms of sample volume, whereas studies that adopt net-based devices report the results per surface sampled (mostly km^2). For example, Anderson et al. (2017) used a Manta trawl to collect surface water samples from Lake Winnipeg, Canada, and found that maximum concentrations reached about 7.48×10^5 microplastics km^{-2} (approximately 4.16 microplastics m^{-3} when factoring in the height of the manta net aperture (0.18 m)). In another study, large numbers of microplastics were found in bulk water samples in China's Taihu Lake, with a maximum concentration of 2.58×10^4 microplastics m^{-3} (Su et al., 2016). The use of different and often large units of measurement makes it difficult to infer biological impacts, as the inflated values lose context on a larger scale. Working towards standardization, it would be beneficial to provide concentrations in meaningful units (e.g. particles m^{-2} rather than particles km^{-2}).

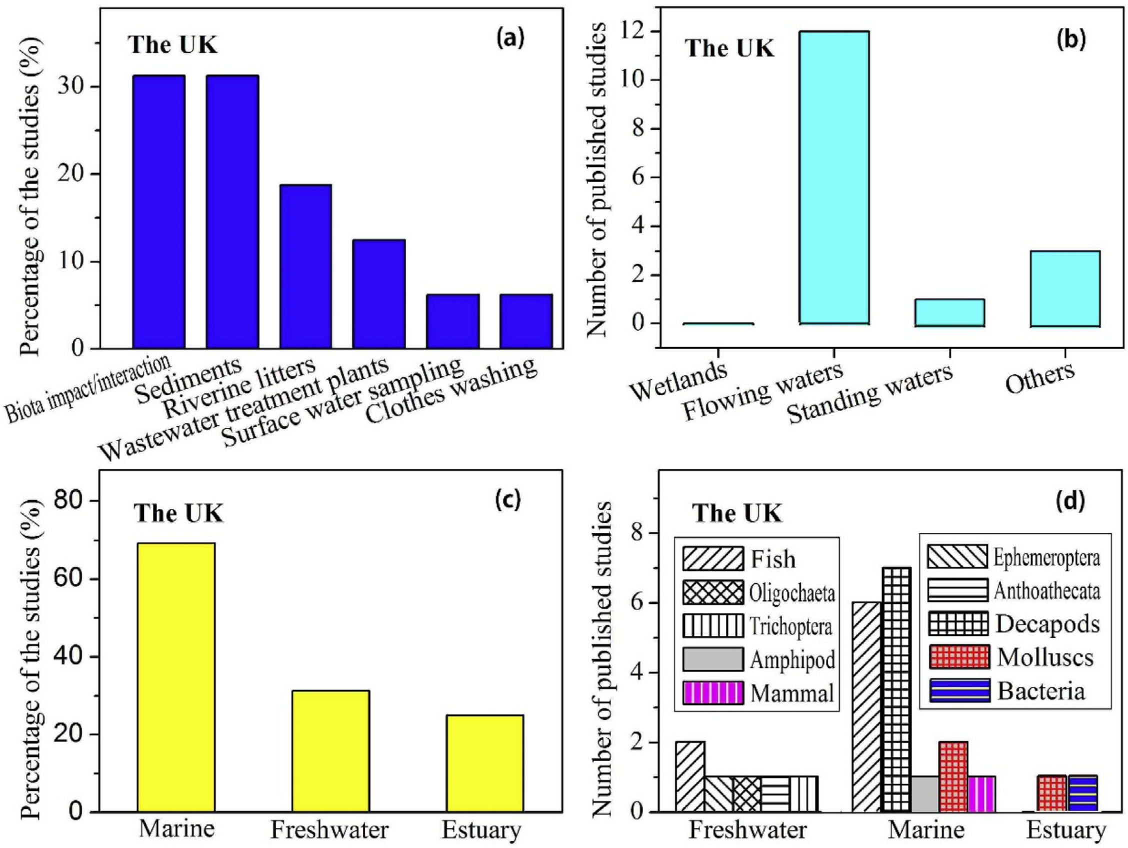

Fig. 3. Microplastic research biases in the UK's aquatic environments. (a) Percentage of studies classified according to the paper's focus; (b) number of studies classified according to the freshwater environment studied; (c) percentage of studies about microplastic interactions with organisms; (d) number of studies classified according to biotic groups impacted by microplastics.

Relative to sediments, only a few water samples from UK's freshwater systems have been collected for microplastic analysis. Kay et al. (2018) used a 300 μm mesh size net in England to collect river water samples. The net, attached to a wooden pole, was placed in the water for sampling durations of 15 min.

For sediment samples in freshwater systems, the most commonly used sampling instruments are sediment grabs (e.g. stainless-steel spoons, trowels, or shovels), which are used to collect the upper sediment layer (~100 mm). It is good practice to avoid contact with plastic equipment during the sampling and sample preparation. Studies that sample the sediment generally report the results in terms of the volume (m^3) or mass (kg) of the sediment samples. Monitoring studies have quantified microplastics from sediments in the UK's freshwater habitats, including rivers and lakes. Riverbed sediment samples were gathered by inserting a large cylinder (diameter: 420 mm; height: 690 mm) into bed sediment to a depth of 10 cm (the biologically active zone) in northwest England (Hurley et al., 2018). In another study, a stainless-steel scoop was used to sample the sediment surface to about 10 cm depth from three tributaries of the River Thames, UK (Horton et al., 2017). Apart from the river sediment samples, sediments have also been sampled in a UK lake habitat. In a study conducted in central Birmingham, UK, researchers collected the top 10 cm of surface sediments of Edgbaston Pool from a boat using an HTH gravity corer equipped with a tube having an internal diameter of 78 mm (Vaughan et al., 2017).

To collect samples of freshwater biota, researchers usually either catch aquatic organisms using effective tools (e.g. net, trawl, electrofishing, or hook and line) from the wild or acquire biota samples from farmed specimens. It's good practice to collect biota based on representative sampling, while, for some organisms such as top predators, it is a challenge to collect enough biota. This in turn will affect the assessment accuracy of the state of microplastic pollution in those organisms. Recently, a small number of studies have been carried out in the UK to detect the presence of microplastics in tissues of freshwater organisms. Most of these biota are sampled from the environment, including (1) Tubifex worms collected from undisturbed sediment samples (Hurley et al., 2017); (2) Heptageniidae (mayflies), Baetidae (mayflies) and Hydropsychidae (caddisflies) collected using a method of rigorous kick sampling and hand-searching (Windsor et al., 2019), (3) roach (*Rutilus rutilus*) using electrofishing techniques (Horton et al., 2018), (4) European flounder and European smelt using fyke nets (McGoran et al., 2017), and (5) in one study, researchers cultured freshwater cnidaria (*Hydra attenuate*) in glass bowls containing 700 mL of Hydra medium (Murphy and Quinn, 2018).

Sampling at wastewater treatment plants has also been carried out in the UK. Murphy et al. (2016) detected microplastics at a wastewater treatment plant located on the River Clyde, Glasgow. An on-site technician used steel buckets (10 L) for sample collection at four stages of the treatment process: influent, grit and grease effluent, primary effluent, and the final effluent before it is released to the surrounding environment (Murphy et al., 2016). The results from the above studies in the UK are discussed in further detail below.

3.4. Sample preparation and microplastic identification

Once the samples have been collected, the next stage is the separation of microplastics from the matrix. In contrast to sediment and sewage samples, it is relatively easy to remove the polymer particles from water samples, which can be filtered using glass fiber filters or stainless-steel sieves at the scene and/or in the laboratory. A commonly reported technique for the separation of microplastics from collected samples is density separation. Frequently used liquid solutions include saturated NaCl, $ZnCl_2$, $KHCO_2$, $CaCl_2$ and NaI (Liebezeit and Dubaish, 2012; Fries et al., 2013; Nuelle et al., 2014; Stolte et al., 2015). Saturated NaCl, with a density of about 1.2 g cm^{-3}, is the most preferred solution since it is inexpensive and environmentally friendly. Other types of solutions have higher densities and may improve extraction of microplastic with a higher density (>1.2 g cm^{-3}), but the cost and toxicity of the solution should be considered prior to use. In UK studies, in order to separate microplastics from sediment particles, researchers have used the density separation approaches, which include using three density solutions (NaCl solution (1.025 g cm^{-3}), NaCl solution (1.2 g cm^{-3}) and NaI solution (1.8 g cm^{-3}) (Hurley et al., 2017; Hurley et al., 2018), a combination of size- and density separation (Vaughan et al., 2017), and $ZnCl_2$ solution (1.7–1.8 g cm^{-3}) (Horton et al., 2017).

Undesirable particles (e.g. organic matter) may still be present in the processed sample and can easily be confused with microplastics during quantification. Therefore, it may be necessary to remove such potentially interfering substances, which can be achieved by chemical or enzymatically-catalyzed reactions. Chemical destruction can be achieved by treating the sample with different chemicals: H_2O_2 (hydrogen peroxide) (Nuelle et al., 2014), a mixture of H_2O_2 and H_2SO_4 (sulphuric acid) (e.g. Imhof et al., 2013), or Fenton's Reagent; (Tagg et al., 2017). The efficacy of peroxide digestions has been questioned (Cole et al., 2014). Alternatively, enzymatic treatments involving the action of enzymes (biological catalysts) can be employed (Cole et al., 2014), but the application has been perceived as a time-consuming and costly undertaking. Several new studies prove the efficiency and the cost-efficiency of enzymatic protocols (e.g. Löder et al., 2017). In UK studies, organic matter digestion was not performed for some sediment studies (Hurley et al., 2018; Vaughan et al., 2017; Horton et al., 2017), while, in the case of biota samples, organic matter oxidation was performed using 15% hydrogen peroxide (H_2O_2) solution (Windsor et al., 2019) and 10% KOH (Hurley et al., 2017). However, some digestions such as H_2SO_4 and HNO_3 with high temperature and/or pressure could degrade microplastics with a low pH tolerance (e.g. polyamide, polystyrene). Hence, assessing digestion methods on a range of polymer types and sizes to validate their efficacies is needed.

Background microplastic contamination is commonplace in the laboratory due to the ubiquity of synthetic fibres indoors. Therefore, rigorous blanks controls should be included during sampling and sample preparation. The blank samples should be analysed using the same procedures as for the samples. Typically, an average blank baseline is deducted from the total microplastic count, however, to improve accuracy, these should be deducted respective to the different polymer types and shapes too.

After the above-mentioned separation and purification processes, the microplastics need to be identified. Microplastic identification depends on the physical and chemical properties of particles. In most cases, visual inspection is first adopted before identification of polymer type is conducted. Large particles can be seen with unassisted vision (the naked eye), whereas small particles must be identified by means of microscopy. Visual identification alone may lead to an overestimation of microplastics due to the presence of interfering inorganic and organic particles. Furthermore, visual identification becomes increasingly difficult the smaller the particle due to a lack of obvious morphological characteristics (J. Lee et al., 2013). Hence, additional analytical approaches are necessary to ensure accurate and robust results. These techniques include Fourier transform-infrared spectroscopy (FTIR) (Löder et al., 2015; Tagg et al., 2015; Cincinelli et al., 2017; Simon et al., 2018), Raman spectroscopy (RS) (Cole et al., 2013; Collard et al., 2015; Imhof et al., 2016; Araujo et al., 2018), FTIR microscopy, Raman microscopy (Kappler et al., 2015), pyrolysis gas chromatography-mass spectrometry (Fabbri, D., 2001; Fries et al., 2013; McCormick et al., 2016; Hendrickson et al., 2018), thermogravimetry coupled to differential scanning calorimetry (Majewsky et al., 2016), and liquid chromatography (Bejgarn et al., 2015; Hintersteiner et al., 2015). Among these techniques the chemical identification of microplastics is most commonly achieved with FT-IR and RS, which both use energy shifts in characteristic functional groups of the sample in identification. Their advantages and limitations have been previously discussed (e.g. Shim et al., 2017): Summarising, (1) both methods are suitable to identify microplastics in environmental samples; (2) compared to RS, FT-IR can result in significant underestimation of microplastics because it is not efficient at detecting small-sized microplastics, especially in the size range <20 μm; (3) RS requires little to no sample preparation, while pretreatment of samples, such as using hydrogen peroxide (H_2O_2) to digest the organic matter, is usually fulfilled when using FT-IR methods; (4) RS is sensitive to fluorescence interference, which is not a problem for FT-IR. In addition, one of the current greatest challenges in microplastics research is to detect very small polymeric particles (e.g. nanoplastics). Atomic force microscopy (AFM) combined with either RS or IR may enable researchers to carry out nanoplastic analysis.

Among the few papers found for UK freshwater microplastics research, suspected microplastics were identified either by visual methods (microscopy) (Vaughan et al., 2017; Kay et al., 2018; Windsor et al., 2019) or by spectroscopic methods, namely FT-IR (Murphy et al., 2016; Napper and Thompson, 2016; Hurley et al., 2017; McGoran et al., 2017; Hurley et al., 2018; Murphy and Quinn, 2018) and RS (Horton et al., 2017; Horton et al., 2018). Non-standardised analytical methods hitherto make it difficult in practice to compare the results between different studies, and therefore this is seldom done. Moreover, the above techniques can be expensive and time-consuming, and therefore only a subset of particles are usually examined. The distribution of the subset across samples often goes unreported and it is unclear how these analysed particles are selected, potentially introducing further operator bias. To improve the scientific robustness and rigor of the field, a class of measurement parameters should be agreed, detailing the minimum proportion of suspected microplastics to be analysed in a sample per shape, and across the size-distribution. Standards should also be set when reporting this information in publications.

4. Microplastic sources, occurrence, transport and fate

4.1. Sources and occurrence in UK freshwater systems

Microplastics can be introduced into freshwater systems from a variety of sources through diverse routes. Identifying sources of microplastics is very important to mitigate the detrimental impacts on freshwater ecosystems. As mentioned previously, the sources can be classified into primary (designed and produced intentionally) and secondary (originating from fragmentation of large plastic items). Small plastic debris with a symmetrical shape and smooth edges/texture can be categorized as primary, otherwise they are assumed to be secondary (Auta et al., 2017; Talvitie et al., 2017). In the UK, microplastics have been observed in both standing and flowing waters.

Primary microplastics are mainly derived from industrial and domestic production, such as toothpaste, facial scrubs, and other personal care products (entering freshwater habitats via household sewage discharge), air blasting media, feedstocks (used to manufacture plastic products) and drug vectors. These products could all be important primary sources of microplastics to freshwater habitats. One important pathway for primary microplastics to enter the environment is via the application of sewage sludge. Previously, sewage sludge has been recycled to land in the UK and Europe in order to reduce the need for manmade fertilisers; around 80% of sewage sludge produced in the UK is applied to agricultural land (DEFRA, 2012). However, this varies between countries and hence the relative importance of this pathway will differ geographically. Murphy et al. (2016) found that microbeads used in personal care products were transported in raw effluent to waste water treatment works and were found in grease samples during the treatment process in a large wastewater treatment work located on the River Clyde, Glasgow, UK. A total of four stages were sampled and analysed: influent after 19 mm coarse screening (15.70 ($\pm$5.23) items L^{-1}), grit and grease effluent (8.70 ($\pm$1.56) items L^{-1}), primary effluent (3.40 ($\pm$0.28) items L^{-1}), and the final effluent (0.25 ($\pm$0.04) items L^{-1}), which represents a very high removal rate (98.4%) (Murphy et al., 2016). Similarly, high removal rates were reported by other studies, such as a 98.3% removal rate recorded at a Finland wastewater treatment plant (Lares et al., 2018). Hence, approximately 79% of the influent microplastic burden may be returned to land.

Notwithstanding the high removal rates achieved, a substantial number of microplastics will enter freshwater habitats in effluent, in view of the enormous volumes of wastewater. Furthermore, a considerable amount of microplastics may have entered the recipient freshwaters though WWTP effluent due to the fact that (1) many of the wastewater treatment plants cannot achieve this high removal rate because of a lack of crucial treatment processes like disc filters or Membrane Bio Reactors, which decrease the amount of particulate matter in the effluent water; and (2) untreated effluent could enter directly into the recipient waters (rivers or streams) when the volume of influent exceeds the treatable capacity in storm events. Upstream actions, e.g. regulating microplastic production and uses, may be effective ways to address the detrimental problem of primary microplastic emissions; rinse-off cosmetics and personal care products containing microbeads were banned in England and Scotland from June 2018.

The major type of microplastic pollution is the secondary particles (Eerkes-Medrano et al., 2015), which are usually derived from macroplastic (>5 mm) fragmentation or synthetic fabrics. It is difficult to identify the origins of secondary microplastics because of the large range of sources and pathways (Wright et al., 2013a; Lasee et al., 2017). Secondary microplastics can be produced before or after entering the environment. Release of synthetic fibres from the laundering of clothes is an example of how microplastics are formed before entering habitats. Napper and Thompson (2016) selected three synthetic fabric types in high-street retail stores near Plymouth, UK to examine the release of fibres from common man-made fabrics. Results indicated that laundering 6 kg of acrylic fabric would release on average 137,951–728,789 fibres per wash. Synthetic fibers have been found in both WWTW sludge and effluent samples (Habib et al., 1998; Zubris and Richards, 2005; Napper and Thompson, 2016). Therefore, fibres could be an important source of microplastics to freshwater habitats where they have long residence times. Since most microplastics can be removed from wastewater via sewage treatment works by retainment in sludge, the sinks of these synthetic fibers (secondary) and primary microplastics are likely similar.

In contrast to synthetic fibres, the fragmentation of macroplastics may mostly occur after entering the environment, such as litter; loss from municipal waste collection, processing and landfills; and industrial, agricultural (plastic mulches and polytunnels) and transportation (such as fragments of road-marking paints, wear and tear from tyres) sources, which are transported by wind and surface runoff water. The degradation processes of large plastic debris mainly comprise physical degradation (such as mechanical abrasive forces), photolysis (usually by UV light), chemical degradation and biodegradation (by algae, bacteria, fungi) (Da Costa et al., 2018). Primary and secondary microplastics have been identified in both standing (lake) and flowing (river) freshwater habitats in the UK, in abiotic compartments including surface waters and bottom sediments. Hurley et al. (2017) estimated that the average microplastic concentration in bottom sediments from the start of the Manchester Ship Canal was 914 ± 844 microplastics kg^{-1}(1793 ± 1275 microplastics m^{-2}, with about 70% of microplastics being secondary (fragments 43%, fibers 24% and other 3%) and about 30% being primary microplastics (microbeads). Hurley et al. (2018) concluded that microplastic pollution was pervasive in all sampled river channel beds in northwest England, with a maximum concentration of around 517,000 microplastics m^{-2}. Fragments of thermoplastic road-surface marking paints were found in sediments of tributaries of the River Thames (Horton et al., 2017). Kay et al. (2018) measured microplastics up- and downstream of six wastewater treatment plants and found that effluent led to an increase in microplastics downstream and that fibres, fragments and flakes were the dominant morphologies in sampled rivers (surface water) in England. This study indicated that WWTPs are key sources of microplastics in river catchments. In addition to the above four studies conducted in flowing freshwaters in the UK, careful examination by Vaughan et al. (2017) indicated maximum concentrations of 25–30 microplastics per 100 g dried sediment, with fibres and films being the most common microplastic shapes in the sediments of Edgbaston Lake, a standing freshwater habitat in central Birmingham. Even though these reported studies only focused on some key regions, their results imply that microplastics are omnipresent in the UK's freshwaters.

4.2. Transport and fate in UK freshwater systems

Microplastics can either directly enter the recipient freshwater or indirectly by means of degradation of larger plastics. As soon as microplastics enter or are formed in freshwater ecosystems, they will be transported over a range of spatial scales with different residence times. In a study conducted in the River Taff, South Wales, UK, researchers noted almost half of all riverine litter were plastics, and tackling the plastics problem is very difficult, partly due to the mobility of litter allowing it to be rapidly transported away from its point of origin once deposited within a catchment (Williams and Simmons, 1999). Morritt et al. (2014) observed that a large unseen volume of submerged plastic was transported into the sea, indicating that rivers are an important medium for transportation of different types of plastic debris to the ocean. Microplastic movement (transportation with water flow or sinking to the bottom) in freshwater habitats, may be influenced by numerous factors, including characteristics of the particles (e.g. density, size, and shape), external forces (e.g. flow regimes: flow velocity, seasonal variability of water flows, water depth, storms, floods, wind-driven surface currents and tidal cycles in estuaries), physical site characteristics (e.g. substrate type, bottom topography, watercourse obstructions and vegetation overhang), degree of fouling, and anthropogenic activity (e.g. dam release). The distribution and retention of microplastics in the UK's freshwater bodies are still not fully understood, mainly due to the lack of data. The challenge of collecting and processing such samples for microplastics analysis, not to mention the time resources required, may be a reason for this. There are, however, several qualitative studies of plastics in UK aquatic environments, especially in freshwaters (Williams and Simmons, 1997; Williams and Simmons, 1999; Balas et al., 2001; Morritt et al., 2014; Nizzetto et al., 2016; Horton et al., 2017; Hurley et al., 2017; Vaughan et al., 2017; Hurley et al., 2018; Kay et al., 2018). Williams and Simmons (1997) observed movement patterns of riverine litter in South Wales, UK, and found that litter movements were influenced by a great number of factors, such as reach features (e.g. vegetation overhang and watercourse obstructions), and that fast-flowing water proved necessary for any significant litter transportation. Likewise, a riverine litter propagation simulation study, conducted on the River Taff, showed that high river discharge volumes were essential for any significant riverine litter propagation, and that litter movement was predominantly controlled by flow and reach characteristics (e.g. daily discharge, vegetation overhang and watercourse obstructions) (Balas et al., 2001). It is evident that particle density is usually a factor influencing transport and fate of microplastics in aquatic environments. A study conducted on the floating plastic debris sampled from surface waters of the Tamar Estuary, Southwest England, found that the most abundant types of plastic were polyethylene (40%), polystyrene (25%) and polypropylene (19%), and microplastics accounted for 82% of the debris (Sadri and Thompson, 2014). Both polypropylene and polyethylene (low/high density polyethylene) have densities less than 1 g mL^{-1}, and polystyrene has a neutrally buoyant density slightly higher than 1 g mL^{-1}. These results suggest that the transport of debris is not influenced by density alone, and that higher-density particles may be resuspend in the water column due to turbulence. In contrast to macroplastics, microplastics will be transported quicker in aquatic environments. A theoretical modelling study (the River Thames was used as a case study) showed that microplastics less than 0.2 mm are usually not retained, regardless of their density, and instead larger (i.e. 0.3–0.5 mm) particles with densities higher than water can be retained in the sediment during base flow periods (Nizzetto et al., 2016). These authors found that very small microplastics (0.001–0.005 mm) can be transported effectively, independent of their densities, and size seems to be a more sensitive parameter influencing microplastic movement dynamics. Microplastics with a density greater than water have the potential to deposit in sediment in freshwater bodies, and total retention efficiency, which is mainly dependent on the dimensions rather than density, is significantly higher for larger particles. However, this sink can be remobilized during intense flow (flooding) periods, indicating a strong hydrological control on the moving or sinking of microplastics within the river system. Hurley et al. (2018) documented that microplastic contamination was pervasive on all river channel beds, with a maximum microplastic concentration of about 517,000 microplastics m^{-2}, in the upper Mersey and Irwel catchments within the Greater Manchester region of northwest England; 70% of the microplastic load on these river beds was exported after a period of severe flooding. Interestingly, the authors also found the total microplastic concentrations at the River Tame pollution hotspot increased by 50% after this period of flooding, suggesting that severe microplastic pollution may develop rapidly when experiencing low stream power in urban rivers. Thus, microplastic impact assessments in river habitats should give priority to these environments. The location of river reach could also have an influence on the microplastic distribution in freshwater systems. Hurley et al. (2018) observed that one sampling site (a reach under the immediate influence of effluent from sewage treatment through a suburban area) in the lower Irwell is dominated by microbeads, while at a site immediately downstream (a highly urbanised reach with an abrupt increase in the density of combined sewer overflows) is dominated by microplastic fragments.

During transport, microplastics present in freshwater systems may also be prone to change in their properties due to degradation through physical, chemical and biological processes and fouling. These degradation processes can change the particle properties (e.g. size, shape and density), which will in turn further influence their fate. microplastic degradation will be influenced by many variables, such as exposure conditions (e.g. UV light), polymer properties (e.g. density and permeability), and type and quantity of chemical additives (e.g. antioxidants and antimicrobial agents). A diverse range of microplastic debris were detected in southern parts of Edgbaston lake, UK, indicating that exposure to the air and higher levels of light may trigger *in situ* fragmentation by photodegradation (Vaughan et al., 2017). Trophic status is another factor influencing the distribution of microplastics within the sediments of freshwater systems, especially standing waters (i.e. lake and reservoir). High microplastic biofouling rates, which were detected within the Edgbaston Pool, is mainly caused by both the greater microplastic presence and its status as a eutrophic lake (rich nutrient constitution). In contrast to standing waters, flowing waters may cause buoyant microplastics to transport downstream before they can become biofouled due to high flow and the trophic status in streams and rivers. Weekly surveys in the Thames and two of its major tributaries showed that baseflow phosphorus concentrations in the Thames reduced from 1584 $\mu g\ L^{-1}$ in 1998 to 376 $\mu g\ L^{-1}$ in 2006 (Neal et al., 2010). Soluble reactive phosphorus concentrations have significantly reduced from an annual maximum of 2100 $\mu g\ L^{-1}$ in 1997 to 344 $\mu g\ L^{-l}$ in 2010 in the River Thames, mainly owing to the introduction of phosphorus removal at sewage treatment works (Bowes et al., 2012). Hurley et al. (2018) did not observe microplastic biofouling in the sediments of the upper Mersey and Irwel catchments. This may be due to less nutrient availability, shorter residence times of the particles in the river beds and potentially abrasive effects of a significant sand load in the rivers. UK river flow patterns normally display significant seasonality, with a strong winter maximum and a summer or autumn minimum. For example, at Teddington, the upper limit of the tideway of the River Thames, the average flow is about 53 $m^3\ s^{-1}$ and can rise to about 130 $m^3\ s^{-1}$ after winter rain. Seasonal variations in water flow within riverine systems is also a factor affecting the presence and transport of microplastics (Horton et al., 2017). The rate of biofouling is contingent upon microplastic parameters such as dimension, surface energy and hardness (Wright et al., 2013a; Fazey and Ryan, 2016). Fibres are one of the most common microplastics detected in the sediments of Edgbaston lake. This may be explained by the fact that fibers have a relatively larger surface-to-volume ratio and thus are more prone to biofouling and sink (Vaughan et al., 2017). Likewise, Vaughan et al. (2017) observed that spherical debris like pellets and microbeads detected in the same sediment samples were lacking. This may be due to the lower surface-to-volume ratio of spherical debris. Therefore, these types of microplastics are not likely to biofoul and sink, and, on the contrary, could remain buoyant for longer and be transported out of Edgbaston Pool via surface currents.

During transport, in addition to experiencing the abovementioned degradation processes, microplastics can adsorb hydrophobic pollutants (e.g. Dichlorodiphenyltrichloroethane (DDT) and Polychlorinated Biphenyls (PCBs), belonging to persistent organic pollutants (POPs)), and affect the bioavailability and mobility of these pollutants. It is well known that microplastics can accumulate these environmental contaminants in marine environments (e.g. Endo et al., 2005; Faure et al., 2015; Bakir et al., 2016;

Guo et al., 2018). Microplastics can also adsorb heavy metals. A study under laboratory conditions confirmed the presence of metals on microplastics, and metals (e.g. Ag, Cd, Co, Ni, Pb and Zn) have a higher affinity for weathered plastic pellets than for new pellets (Turner and Holmes, 2015). It is thus becoming increasingly clear that microplastics, especially secondary microplastics, could represent a significant matrix for the transport of metals in aquatic systems. Ashton et al. (2010) observed that polyethylene pellets, which were suspended in a harbor (Southwest England) for 8 weeks, could accumulate metals from seawater through adsorption and precipitation. So far, very limited data concerning the interactions between microplastics and these water-borne pollutants in UK and global freshwater systems is available. There are few studies which have assessed POPs in sediments within freshwater habitats in the UK. For example, Lu et al. (2017) collected sediment samples from seven sites in the River Thames and its tributaries and found a range of concentrations of these pollutants (PCBs, hexachlorobenzene (HCB) and polybrominated diphenylethers (PBDEs)) in the sediments, with the highest values detected in the Cut at Bracknell (an urbanised tributary of the Thames). The partitioning between hydrophobic pollutants and microplastics will be influenced by variations in environmental conditions, and characteristics of microplastic particles (e.g. polymer type and weathering state) will influence the pollutant-absorption capacity.

5. Microplastic-biota interactions in UK freshwater environments

Across the globe, research on the ingestion of microplastics by biota has predominantly focused on a wide range of marine species with different feeding strategies (e.g. Goldstein et al., 2012; Wright et al., 2013a; Amelineau et al., 2016; Alomar and Deudero, 2017; Scopetani et al., 2018), however, there is an increasing number of studies focusing on this issue in freshwater species (e.g. Au et al., 2015; Rehse et al., 2016; Silva-Cavalcanti et al., 2017; Collard et al., 2018; Redondo-Hasselerharm et al., 2018). Ingesting microplastics can cause adverse impacts (e.g. on growth and development, feeding or reproductive behaviour) in a range of aquatic biota such as fish, zoobenthos, zooplankton and mollusks. These negative impacts can be classified as physical and/or particle, chemical and microbial effects.

Physical and particle effects of microplastics include inflammatory responses (e.g. abrasion and ulcer) and compromised energy reserves (reduced lipid stores) potentially caused by reduced assimilation from the natural diet (Wright et al., 2013a, b; K.W. Lee et al., 2013; Cole et al., 2015). Although there are few freshwater studies so far (Rochman et al., 2013; Rehse et al., 2016), these physical impacts may be applicable. Rehse et al. (2016) found that limnic zooplankton can ingest 1 μm polyethylene particles which led to immobilisation of daphnids at high concentrations. The microplastic observation under controlled laboratory conditions are different from the situation in the field (Triebskorn et al., 2019). There is a discrepancy between concentrations used in laboratory studies and those measured *in situ* (Botterell et al., 2019; Burns and Boxall, 2018). Measuring the effects of environmental concentrations is also important to understand what is happening *in situ*, although higher concentrations can indicate threshold levels which should not be reached in the future. Several factors can influence these physical effects on organisms, including accumulation, translocation, shape, and egestion.

Chemicals, including adsorbed chemical contaminants from the environment and the additives that are incorporated into the polymer at the production stage, may be transferred to organisms, the likelihood increasing with retention time, depending on the duration for equilibrium partitioning. Studies have observed that microplastics localise in the gut, gills and liver of aquatic organisms following exposure (e.g. Watts et al., 2014; Lu et al., 2016). The size and shape of plastic particles are the two most important parameters which determine the extent of microplastic retention. This is because smaller particles are more likely to be ingested and particles with angular shapes may be harder to egest. The available body of evidence indicates that trophic transfer of microplastics may occur (e.g. Farrell and Nelson, 2013; Nelms et al., 2018; Welden et al., 2018). Hence, pollutants may be transferred from microplastics by means of oral ingestion as well as other pathways, such as ventilation or simple microplastic attachment and re-suspension into the water column (e.g. Watts et al., 2016; Gray and Weinstein, 2017; Batel et al., 2018).

In freshwater habitats, the adsorption of different POPs (e.g. PCBs, HCB, PBDEs and metals) to hydrophobic plastic particles with a large surface area to volume ratio may be more significant than in marine habitats, due to the proximity to the sources and use of these chemicals (Dris et al., 2015). Organisms in freshwater habitats thus might experience higher exposures, especially close to industrial and populous areas, where there may be both higher concentrations of the hydrophobic contaminants and a greater presence of microplastics, and in proximity of some agricultural areas, in which both plastic products and POPs (i.e. pesticide) are applied. Rochman et al. (2013) concluded that polyethylene ingestion is a vector for PBTs (persistent, bioaccumulative and toxic substances) in fish, and that observed hepatic stress was caused by both the adsorbed pollutants and plastic material. Another laboratory study found that significant amounts of POPs adsorbed to microplastics could accumulate in adult zebrafish gills and zebrafish embryos (Batel et al., 2018). In practice, however, the vector effect hypothesis remains controversial. Besseling et al., 2013 observed that although a low polystyrene dose of 0.074% increased PCB bioaccumulation in a benthic marine organism, Lugworm *Arenicola marina (L.)*, PCB105 bioaccumulation notably decreased at higher polystyrene doses. Kwon et al. (2017) concluded that when the fugacity of the POPs in the plastic phase is lower than those in water and aquatic organisms, microplastics should be considered a sink for POPs. The current consensus is that, for most habitats, POP uptake via microplastics is likely negligible compared to bioaccumulation from natural pathways (Koelmans et al., 2016). An exception may be chemical additives, used in plastics to improve physical properties, which may be present at high concentrations and include: antioxidants, UV stabilisers, colour pigments, biostabilisers, antimicrobials/antibacterials, flame retardants, antistatics, biodegraders, foaming/blowing agents, lubricants, fillers, fragrances and impact modifiers.

Microplastic vector effects have mainly been assessed in marine conditions. There is limited information available concerning the sorption kinetics of microplastics in freshwater systems, although environmental exposure has been examined. For example, Faure et al. (2015) collected fish and water birds, which were examined to assess their potential exposure to hydrophobic pollutants as well as some potentially toxic additives, from freshwater lakes in Switzerland. These authors found that both birds and fish ingest microplastics, and both potentially adsorbed pollutants and additives were detected in these biota samples. However, whether microplastics were a mode of entry for these chemicals, and thus into freshwater food chains, remain inconclusive. A comprehensive and critical systematic review of microplastic vector effects, along with studies which truly test this and sorption kinetics in freshwater scenarios would progress the field.

Microplastics can also act as an artificial substrate for microorganisms. This has raised concern about the potential ecological effects on freshwater habitats, which provide essential benefits and services such as habitats for a wide variety of native plants and animals, drinking water, as well as recreational activities. However, limited information is available about the actual ecological impacts of microplastics within freshwaters. Because of their high surface area/volume ratio and hydrophobicity, microplastics provide novel surfaces for microorganisms to attach to, forming so-called biofilms. In ecological terms, this can influence the interaction between the microplastics and freshwater biota on a large scale, such as the likelihood of geographically rafting the colonized organisms over longer distances and making microplastics vectors for pathogens, toxic algae/bacteria and even invasive species. For example, McCormick et al. (2016) found that bacterial assemblages colonizing microplastics within a highly urbanised river in Chicago, Illinois, USA, were significantly different in taxonomic composition compared to those from the water column and suspended organic matter, and several taxa, including plastic decomposing organisms and pathogens, were more abundant on microplastics. Another study measured bacterial assemblage composition on microplastics and in river surface water and noted that bacterial assemblage composition was dissimilar among microplastics, seston, and the water column (McCormick et al., 2016). Eckert et al. (2018) mixed microorganisms from treated WWTP water and natural lake water to simulate a WWTP effluent and followed the bacteria survival on the plastisphere. The authors cautioned that the presence of microplastics favour the survival of WWTP-derived bacteria, which are involved in the transmission of antibiotic resistance genes in freshwater habitats (Eckert et al., 2018).

Consumption of microplastic particles by aquatic organisms has been reported in UK and global aquatic systems, especially in marine habitats (Browne et al., 2008; Cole et al., 2013; Lusher et al., 2013; Watts et al., 2014; Cole et al., 2015; Devriese et al., 2015; Watts et al., 2015; Watts et al., 2016; Welden and Cowie, 2016a; Welden and Cowie, 2016b; Murphy et al., 2017; Steer et al., 2017; Nelms et al., 2018; Hodgson et al., 2018). In UK's freshwater habitats, knowledge on uptake of microplastic particles by organisms is very limited, especially whether harmful effects arise in freshwater animals. Hurley et al. (2017) found that microplastic particles, of which 87% were microfibers and the remaining 13% were microplastic fragments, were ingested by Tubifex worms (one of the most abundant freshwater invertebrates). Microbeads were not present in Tubifex worms' tissue, indicating the observed microbeads were too large for ingestion (Hurley et al., 2017). Windsor et al. (2019) observed that the presence of microplastics within three invertebrate taxa in the South Wales valleys, representative of the highly urbanised river systems, revealed a possible risk from microplastics entering riverine food webs via at least two pathways, including detritivory and filter-feeding biota. Horton et al. (2018) observed that, within the non-tidal reach of the River Thames, microplastics were ingested by the roach *Rutilus*, and the maximum number of ingested microplastics per fish was significantly correlated to exposure and, additionally, larger (mainly female) fish are expected to ingest higher microplastic numbers than smaller fish. A study, conducted in the River Thames, revealed that up to 75% of sampled European flounder (*Platichthys flesus*) had plastic fibres in their gut compared to 20% of European smelt (*Osmerus eperlanus*) (McGoran et al., 2017). This difference may depend on their feeding strategies: European flounder are benthic feeders while European smelt are pelagic predators. In a laboratory feeding study, Murphy and Quinn (2018), found that the food intake of *Hydra attenuate* (freshwater cnidarian) was significantly reduced due to exposure to microplastics, that evident variations in *H. attenuata* morphology were detected but these were non-lethal, and that reproduction of *H. attenuate* was not influenced by the presence of microplastics.

Overall, laboratory studies have shown that microplastics have a range of impacts on aquatic organisms such as feeding, growth, reproduction, behaviour and survival. However, current understanding of the potential risks of microplastics in aquatic environments is still limited. Adverse effects are often reported for individual species rather than at a community level. Microplastics used in tests are often not representative of the polymer types and shapes detected in the environment; there is a focus on spherical particles rather than fragments and fibers, and single types of polymers are tested rather than a mixture. Moreover, whether observed negative outcomes are due to a plastic effect or a particle effect is unknown. Clay particles were found to cause effects in daphnids (Robinson et al., 2010) and the bioavailability of HOCs via microplastics was found to be lower than naturally-occurring particles (Beckingham and Ghosh, 2017). The long-term ecological implications for the UK's freshwater organisms are insufficiently studied so far but should be a research priority.

6. Conclusions and recommendations

While microplastics have been widely documented in marine ecosystems, relatively fewer studies have examined microplastic contamination in freshwaters both in the UK and globally. In this review, all available scientific publications on microplastic pollution in the UK's aquatic environments were examined. This review highlights the ubiquity of microplastics in UK freshwater ecosystems, but the existing information is still fragmented, incomplete and biased. Except for England, there is very little data available in other parts of the UK (Scotland, Wales and Northern Ireland). Research efforts seemed to be relatively biased towards rivers, while other freshwater types such as wetland, lakes and reservoirs have received much less research focus. Moreover, few studies have examined the interaction of UK freshwater species with microplastics. Microplastics have been found in surface waters, sediments and biota in the UK's rivers and lake habitats, indicating their pervasiveness. Both primary and secondary microplastics have been identified, and the latter are the major type of microplastic contamination. Although many sources have been detected, the pathways of microplastics reaching freshwater habitats are less well understood. Some types of microplastics, such as fibers (e.g. derived from synthetic textiles), fragments (e.g. released from road marking paints and the wear of vehicle tyres), films and flakes, have been highlighted as the most common contributors to microplastics in surface waters and sediments, and fibers and fragments have been identified as the dominant contributors to ingested microplastics. Visual (microscopy) and spectroscopic methods (FT-IR and RS) are currently the main technologies used for microplastic identification. These are laborious and for the field to progress, more high-throughput and accurate techniques are required.

The presence of microplastics in freshwater ecosystems is of increasing environmental concern. UK freshwater studies have mostly taken place over the last five years, and current knowledge of its sources, occurrence, transport, fate, potential impacts and possible solutions remain in its infancy. Some important gaps in our current knowledge needing addressing and are listed below, along with some recommendations.

- **Increase scientific efforts where field data on microplastic contamination is lacking.** It is necessary to expand the geographical range of research, especially into Scotland, Wales and Northern Ireland, since most scientific effort has focused on freshwater environments in England. Several further studies are needed to focus on key habitats (e.g. wetland, lake, reservoir, and pond) and freshwater species. However, the current widely accepted analytical methods are laborious and size restrictive. Urgent action to develop more time-efficient and reliable methods is required for monitoring microplastics in the environment in future studies.

- **Monitor microplastics in aquifers.** Aquifers, namely underground reservoirs, in some cases feed surface water supplies such as rivers and lakes in the UK. More research is needed, particularly in England where groundwater supplies almost a third of the drinking water, to determine whether microplastics pollute groundwater and, if so, what can be done to solve this. Groundwater is recharged by surface water, which has proven to be contaminated by microplastics (e.g. Kay et al., 2018), hence this is a likely scenario and urgent research area.
- **Examine the transport dynamics of microplastics from point and diffuse sources.** Point sources of microplastic pollution, such as wastewater treatment plants and industry, could effectively remove microplastics (more than 98%) using the latest available technologies, e.g. an advanced membrane bioreactor technology (Lares et al., 2018), to reduce the level discharged via effluents to a receiving water body. Diffuse (non-point) sources of microplastic pollution usually occur over a relatively wide area, such as agricultural lands where sewage sludge containing microplastics and agricultural plastics (e.g. polytunnels and plastic mulches) are applied. Diffuse microplastic pollution is generally more difficult to control than point source pollution. It should be mentioned that: (1) when it rains heavily, untreated raw sewage may overflow and discharge directly into the nearest waterbody, which can cause extensive damage to the receiving water habitats; (2) challenges remain in remediating microplastic contamination from diffuse sources and more attention is needed to address this.
- **Ensure sampling is as representative as possible.** Representative microplastic samples are essential for accurate analyses and interpretation. The results of the included studies show that microplastics in the UK's freshwater systems are heterogeneously distributed, with a considerable magnitude of variation. It is noteworthy that rivers can play a very critical role in transporting microplastics to oceans. The input of microplastics from rivers into the marine environment may be underestimated since small microplastics below the sampling cut-off size (e.g. 300 μm) and nanoplastics, defined as < 100 nm in size, are not considered or may be suspended beneath the sampling layer. To avoid random sampling and gain a complete picture of freshwater microplastic pollution in the UK, it is critical that current sampling techniques are improved and that the spatial and temporal variations of microplastics in freshwaters should be considered. Moreover, when collecting samples, sampling locations (site conditions: land use features, economic activities, topography, freshwater/wastewater inflows etc.) and sampling timing (e.g. weather and water flow) should be taken into consideration.
- **Evaluate and mitigate microplastic pollution risks in UK freshwater systems.** The available information on the potential adverse impacts of microplastics in the UK's freshwaters is extremely limited. It remains unknown what the extent and relevance of the harmful effects on aquatic life are, and if and how microplastics might affect human health via consumption of contaminated water or contaminated food harvested from freshwater habitats. Thus, effort and resources are required to close the knowledge gaps concerning the specific toxicities, threats and adverse effects of microplastics. The risk assessment of microplastics is in its infancy, and additional research is needed to produce qualitative and quantitative data to understand the extent of the problem in the UK.
- **Introduce more comprehensive and effective legislations/regulations to address the microplastics problem.** The UK has attempted to tackle environmental microplastics via changing from passive to proactive actions, namely, imposing a ban on the sale of products containing microbeads from June 2018 followed by January's ban on the production of products containing microbeads. The Bill targets personal care products (e.g. toothpaste, facial scrubs, sunscreen, soap and make-up). However, the predominant type of microplastics measured in UK freshwaters are secondary; microbeads are a small fraction of the UK's freshwater microplastics burden. Clearly, the ban is an initial step towards remediating microplastic pollution, and the UK should now develop and implement more effective and comprehensive regulatory frameworks to control microplastic contamination. In addition to the well-known effective regulations intended to reduce, reuse, recycle and even prohibit plastic products, remedial actions must be taken to cope with the microplastic pollution already occurring in UK freshwaters.

Acknowledgements

This research was part-funded by the China Scholarship Council (CSC) (No. 201806245015). The research was part-funded by the National Institute for Health Research Health Protection Research Unit (NIHR HPRU) in Health Impacts of Environmental Hazards at King's College London in partnership with Public Health England (PHE), and the Medical Research Council (MRC; MR/R026521/1). The views expressed are those of the author(s) and not necessarily those of the CSC, NHS, NIHR, PHE or MRC.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.envpol.2019.113445.

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