Wastewater treatment plant effluent as a source of microplastics: review of the fate, chemical interactions and potential risks to aquatic organisms

Shima Ziajahromi, Peta A. Neale and Frederic D. L. Leusch

ABSTRACT

Wastewater treatment plant (WWTP) effluent has been identified as a potential source of microplastics in the aquatic environment. Microplastics have recently been detected in wastewater effluent in Western Europe, Russia and the USA. As there are only a handful of studies on microplastics in wastewater, it is difficult to accurately determine the contribution of wastewater effluent as a source of microplastics. However, even the small amounts of microplastics detected in wastewater effluent may be a remarkable source given the large volumes of wastewater treatment effluent discharged to the aquatic environment annually. Further, there is strong evidence that microplastics can interact with wastewater-associated contaminants, which has the potential to transport chemicals to aquatic organisms after exposure to contaminated microplastics. In this review we apply lessons learned from the literature on microplastics in the aquatic environment and knowledge on current wastewater treatment technologies, with the aim of identifying the research gaps in terms of (i) the fate of microplastics in WWTPs, (ii) the potential interaction of wastewater-based microplastics with trace organic contaminants and metals, and (iii) the risk for aquatic organisms.

Key words | fate, microplastics, removal, risk, sorption capacity, wastewater

ABBREVIATIONS

DDT = dichlorodiphenyltrichloroethane; FT-IR = Fourier transform infrared spectroscopy; LOAEC = lowest observable adverse effect concentration; PAH = polycyclic aromatic hydrocarbon; PBDE = polybrominated diphenyl ether; PCB = polychlorinated biphenyl; PVC = polyvinyl chloride; TrOC = trace organic contaminants; UPVC = unplasticized polyvinyl chloride; WWTP = wastewater treatment plant; K_{PE-W} = polyethylene-water partition coefficient; K_{OW} = octanol-water partition coefficient.

INTRODUCTION

The occurrence of plastic debris as a widespread pollutant has been reported in various aquatic environments, including rivers, lakes, estuaries, coastlines and marine ecosystems, and represents an increasing environmental concern (Eerkes-Medrano et al. 2015).

Currently, the polymers that contribute to 90% of plastic products are (low-density and high-density) polyethylene, polypropylene, polyvinyl chloride, polystyrene, and polyethylene terephthalate (or polyester) (Zarfl & Matthies 2010; Ivar do Sul & Costa 2014). Consequently, the majority of released plastic pollutants in the environment are composed of these polymers (Ivar do Sul et al. 2014).

Microplastics, which are plastic fragments in the micrometer size range (<1 mm) (Browne et al. 2011), are available to a wide range of aquatic organisms as they occupy the same size range as their natural food sources (Zhao et al. 2014). Microplastics have the capacity to interact with a variety of contaminants in the surrounding environment (Holmes et al. 2012; Bakir et al. 2014; Wang et al. 2015), which may also pose a risk for aquatic organisms. Further, additives that are included in the polymer structure during manufacturing may also lead to harmful consequences if they are leached to exposed organisms (Browne et al. 2015). Nevertheless, the degree to which various microplastics themselves and their associated contaminants may cause a risk to aquatic organisms still remains poorly understood.
Generally, the sources of microplastics in the aquatic environment have been classified as (i) aquatic-based sources and (ii) land-based sources and include both primary and secondary microplastics (Law et al. 2010; Hammer et al. 2012; Eriksen et al. 2013b). Primary microplastics are manufactured as microplastics, while secondary microplastics are produced from the breakdown of larger plastic particles (Zhao et al. 2014). Aquatic-based microplastics may originate from the degradation of larger plastic objects discarded through human activities such as littering, fishing and shipping, and are considered as secondary microplastics (Hammer et al. 2012). Land-based sources include run-off containing microplastics (solid wastes). After heavy rainfall, microplastics can be flushed into the aquatic environment, especially from poorly managed or illegal landfills (Teuten et al. 2009; Hammer et al. 2012; Wagner et al. 2014; Duis & Coors 2016), and these particles can also be classified as secondary microplastics given that they undergo breakdown before entering the aquatic environment (Eerkes-Medrano et al. 2015). Wastewater treatment plant (WWTP) effluent is another land-based source of microplastics (Cole et al. 2011; Wagner et al. 2014; Eerkes-Medrano et al. 2015) and can include both primary and secondary microplastics. Depending on the wastewater treatment facilities, effluent may be released directly into the ocean or into the fresh water environment, such as rivers or lakes, which may then be transported to the marine environment (McCormick et al. 2014).

While land-based sources are assumed as one of the most important sources of microplastics (da Costa et al. 2016), there is a lack of quantitative estimation on the relative contribution of different sources to the total concentration of microplastics in the aquatic environment (Koelmans et al. 2014; Duis & Coors 2016).

Amongst the different sources of microplastics to the aquatic environment, wastewater effluent has recently received considerable attention as a potential point source of microplastics (Browne et al. 2011; Carr et al. 2016). However, the contribution of microplastics released via wastewater effluent compared to other sources is largely unknown (Magnusson & Norén 2014; Eerkes-Medrano et al. 2015).

The importance of WWTP effluent as a source of microplastics may increase in the future, given current population growth as well as rapid urbanization and industrialization, which can lead to increasing volumes of wastewater (UNEP 2010; Browne et al. 2011). Subsequently, the contribution of wastewater-based microplastics into the receiving environment may continue to increase if its importance is not fully realized and no further actions are taken to prevent and reduce microplastics in wastewater.

The current review highlights the presence and potential importance of WWTP effluent as one of the sources of microplastics in the aquatic environment by reviewing the current state of knowledge. The review will also provide an in-depth analysis on the fate of microplastics in WWTPs, the potential mechanisms involved in the interaction of wastewater-based microplastics and trace organic contaminants (TrOCs), and the adverse effects of microplastics and their associated TrOCs from a risk-based perspective. The review will conclude by highlighting areas of future research.

**SOURCES OF WASTEWATER-BASED MICROPLASTICS INTO THE AQUATIC ENVIRONMENT**

Over the last few years, various types of microplastics with a suspected WWTP origin have been reported in aquatic ecosystems (Table 1). One of the common wastewater-based microplastics is microbeads, such as polyethylene and propylene, which contribute to over 90% of microbeads associated with cosmetic and personal care products (Zitko & Hanlon 1991; Fendall & Sewell 2009; Leslie 2014; Eriksen et al. 2014; Dris et al. 2015c; Gouin et al. 2015). While a number of cosmetic companies have voluntarily started to remove plastic microbeads from their products and replace them with natural compounds (Gouin et al. 2015), it was recently reported that 80 body and facial scrub products in the UK market still contain polyethylene microbeads that can be consumed by millions of consumers worldwide (Beat the Microbeads 2015; Napper et al. 2015). Moreover, a very recent study of eight municipal WWTPs, effluent across Southern California revealed that the majority (over 90%) of detected microplastics had a profile resembling the blue polyethylene microplastics extracted from toothpaste (Carr et al. 2016). Thus, microbeads in personal care products are currently an important potential source of microplastics in WWTP effluent.

Recently, the US Government established the ‘Microbead-Free Waters Act of 2015’, which prohibits the production or sale of rinse-off personal care products (e.g. toothpaste or facial scrubs) containing microbeads from 2017 (Congress.gov 2015).

Other plastic polymers also used in cosmetic products include polyactic acid, polyethylene terephthalate (Rochman et al. 2015), polyethylene isophthalate, nylon-12, nylon-6, poly(methyl methacrylate), polytetrafluoroethylene, and
Table 1 | Some studies detecting microplastics in the aquatic environment pointing to a suspected wastewater origin

<table>
<thead>
<tr>
<th>Study</th>
<th>Study location</th>
<th>Sampled environment</th>
<th>Average abundance of sampled microplastics</th>
<th>Mesh size of sampling method</th>
<th>Origin of sampled microplastics</th>
</tr>
</thead>
<tbody>
<tr>
<td>Zhao et al. (2014)</td>
<td>Yangtze Estuary and the coastal waters of the East China Sea</td>
<td>Surface water</td>
<td>4,137 particles/m³ in Yangtze Estuary, and 0.167 particles/m³ coastal waters of East China Sea</td>
<td>333 μm mesh neuston net</td>
<td>Approximately 80% were fibres, which were suspected to derive from sewage and/or from degradation of rope material and fishery activities</td>
</tr>
<tr>
<td>Castañeda et al. (2014)</td>
<td>St Lawrence River, Canada</td>
<td>Sediment</td>
<td>13,832 particles/m²</td>
<td>500 μm sieve</td>
<td>99.9% of polyethylene microbeads had a similar size, shape and chemical composition to microbeads in cosmetic products</td>
</tr>
<tr>
<td>Claessens et al. (2011)</td>
<td>Belgian coast</td>
<td>Sediment</td>
<td>37,000–149,000 particles/m²</td>
<td>Bulk sample of sediment was taken and passed through 38 μm mesh</td>
<td>Nylon and polypropylene fibres followed by polyethylene and polypropylene fragments were the dominant type of microplastics that were suggested to be associated with carpet and clothing and cosmetic products, respectively</td>
</tr>
<tr>
<td>Browne et al. (2011)*</td>
<td>Australia, Oman, United Arab Emirate, Chile, Philippines, Portugal, USA, South Africa, Mozambique, UK</td>
<td>Sediment</td>
<td>2–31 particles /250 ml of sediment</td>
<td>NA</td>
<td>The dominant types of collected microplastics were polyester and acrylic, followed by polypropylene, polyethylene, and polyamide fibres, with detected fibres similar to fibres found in wastewater effluent, most likely from washing synthetic clothes</td>
</tr>
<tr>
<td>Dris et al. (2015a)*</td>
<td>River Seine, France</td>
<td>Surface water</td>
<td>30 and 0.35 particles/m³ for plankton net and manta trawl, respectively</td>
<td>80 and 330 μm mesh, plankton net and manta trawl</td>
<td>The dominant type of sampled microplastics was fibres, and urban water system discharges, including wastewater, were considered as one of the sources of the detected microplastics</td>
</tr>
<tr>
<td>Eriksen et al. (2015a)</td>
<td>Lakes Superior, Huron and Erie, USA</td>
<td>Surface water</td>
<td>43,000 particles/km²</td>
<td>333 μm mesh manta trawl</td>
<td>Many microplastics were multi-coloured with a spherical shape and were suspected to be microbeads from cosmetic products associated with wastewater</td>
</tr>
<tr>
<td>Gallagher et al. (2015)</td>
<td>Solent Estuarine Complex, UK</td>
<td>Water column</td>
<td>Total of 2,759 particles at four different sampling site</td>
<td>300 μm plankton net trawl</td>
<td>The majority of detected microplastics were fibres (expected to be polyester) and rounded pellets (expected to be polyethylene and polypropylene), which were likely to be associated with wastewater effluent discharge</td>
</tr>
</tbody>
</table>

(continued)
Studies that confirmed wastewater as a source of detected microplastics using Fourier transform infrared spectroscopy.

Polyurethane (Leslie 2014), and these polymers may enter wastewater influent following production and use of those products. Additionally, microplastics with physical properties that resemble microbeads, such as polyethylene, polystyrene and thermoset polyester polymer applied in air and sand blasting technologies, are also a potential source of microplastics in wastewater (Eriksen et al. 2013a).

Microfibres, such as polyester, acrylic, and nylon associated with clothes, carpets and other textile products (Browne et al. 2007, 2011; Claessens et al. 2011), may unintentionally find their way into the aquatic environment via wastewater during washing and/or manufacturing of synthetic fibres and represent another type of wastewater-based microplastics (Teuten et al. 2007; Browne et al. 2011; Law & Thompson 2014; Driedger et al. 2015; Rocha-Santos & Duarte 2015). More than 100 fibres per litre of laundry wastewater can be released into WWTPs from washing polyester clothes (Browne et al. 2011; Dris et al. 2015c). Thus, textile and carpet industries as well as commercial laundries are likely to release significant amounts of microplastic fibres to WWTPs and, depending on the level of treatment, this wastewater effluent may represent a relatively large source of microplastic fibres. However, this has yet to be estimated.

McCormick et al. (2014) reported high concentrations of microplastics, mostly fibre associated with synthetic fabrics, in the North Shore Channel of the Chicago River, which receives secondary effluent from a domestic treatment plant that currently operates with secondary clarifiers (Storm Water Solution 2013).

Microplastics with a similar shape and composition to microbeads used in cosmetic products have been detected in surface waters from three of the Great Lakes in North America, with their source suspected to be wastewater effluent discharges (Eriksen et al. 2013a). This finding was supported by the detection of microbeads in wastewater effluent discharged into the Great Lakes (Schneiderman 2013). Further, Claessens et al. (2011) found high concentrations of microplastics with the dominant shape of fibres and granules in marine sediment along the Belgian coast. Although this study was not able to determine the source of the detected microplastics, it was speculated that the granules, which included polyethylene and polypropylene, were likely to have originated from cosmetic products, air blasting media and the plastics industry. Additionally, a proportion of fibres, such as nylon, were suspected to be derived from fibrous material including clothes and carpets.

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**Table 1**

<table>
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<th>Study</th>
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<th>Sampled environment</th>
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<th>Mesh size of sampling method</th>
<th>Origin of sampled microplastics</th>
</tr>
</thead>
<tbody>
<tr>
<td>Talvitie et al. (2013)*</td>
<td>Helsinki archipelago coastal Gulf of Finland</td>
<td>Water column &amp; sediment</td>
<td>0.01 and 0.65 fibres/L, 0.5 and 9.4 synthetic particles/L</td>
<td>20–200 μm using a filtration device</td>
<td>Fibres and synthetic particles with properties similar to microplastics detected in wastewater effluent were found in the receiving water body of a WWTP</td>
</tr>
<tr>
<td>McCormick et al. (2014)</td>
<td>North Shore Channel, Illinois, USA</td>
<td>Surface water</td>
<td>17.93 particles/m³ downstream of WWTP</td>
<td>333 μm mesh neuston nets</td>
<td>1.21 and 10.57 particles/m³ were fibres and 0.45 and 0.01 particle/m³ were pellets, in downstream and upstream samples, respectively. Microplastics were suspected to be associated with synthetic textiles and cleansing products in wastewater</td>
</tr>
<tr>
<td>Woodall et al. (2014)</td>
<td>Mediterranean Sea, SW Indian Ocean and NE Atlantic Ocean</td>
<td>Sediment</td>
<td>14,000 particles/m² of Indian Ocean Sediment</td>
<td>Collected sediments passed through 32 μm mesh sieve</td>
<td>The dominant type of detected fibres was polyester (33%) followed by polyamide (34%) and acrylic (12%). Textile was suggested as one of the potential sources</td>
</tr>
</tbody>
</table>

Note: 70 kg sediment, 0.1 m² of sampling site (Claessens et al. 2011).
NA: Not available.
*Studies that confirmed wastewater as a source of detected microplastics using Fourier transform infrared spectroscopy.
(Claessens et al. 2011), and thus are likely to be associated with domestic and/or industrial wastewater discharge.

Deep-sea sediments sampled at different locations worldwide, including the Mediterranean Sea, South West Indian Ocean and North East Atlantic Ocean, showed that polyester and acrylic polymers contributed to the majority (86%) of analysed microplastics and were suspected to derive from sources such as packaging material, electronics and textiles (Woodall et al. 2014). Wastewater was suggested as one of the potential sources of microplastics in all above studies, but the actual contribution of wastewater to the detected microplastics has yet to be established. In contrast, Klein et al. (2015) detected microplastics in sediment along the shore of the Rhine and Main Rivers in Germany and found lower concentrations of microplastics near WWTPs compared to other monitored areas, with no relationship between the number of microplastics detected and the proximity to WWTPs found. Moreover, the detected microplastics primarily included polyethylene, polypropylene and polystyrene, which suggests that they originated from discarded plastic products, as these polymers comprise 55.7% of European plastic demand (Klein et al. 2015).

A number of factors may affect the contribution of wastewater as a source of microplastics and also the dominant type of wastewater-based microplastics in a specific area including (i) population density, (ii) the number and type of industries involved in producing plastics, (iii) the level of wastewater treatment, (iv) the public awareness with respect to microplastics and their subsequent effects and (v) waste management processes (Free et al. 2014; Jambeck et al. 2015). The correlation between some of the above-mentioned factors and input into the aquatic environment has been reported for other pollutants (Abril et al. 2002). Thus, it is likely that highly populated and industrialized communities with low levels of wastewater treatment, as well as inefficient waste disposal management, may be major contributors to the release of wastewater-based microplastics to the environment. It is reported that 90% of all wastewater in developing countries is discharging untreated to the aquatic environment such as lakes, rivers and the oceans (UNEP 2010).

**FATE OF WASTEWATER-BASED MICROPLASTICS IN WWTP**

While the presence of wastewater-based microplastics in the aquatic environment has recently attracted the attention of scientists (Fendall & Sewell 2009; Cole et al. 2011; Eriksen et al. 2013a; McCormick et al. 2014), less attention has been drawn to the fate of microplastics during wastewater treatment (Duis & Coors 2016).

Before returning to the aquatic system, wastewater may be treated in WWTPs. Depending on the type and scale of the facility, a WWTP can have primary, secondary and tertiary treatment trains to purify the influent using physical, chemical and biological treatment processes. During physical and chemical processes many suspended and floating solids including plastic particles are removed through screening, sedimentation, flotation, coagulation-flocculation and filtration within treatment ponds or settling tanks (Stephenson 2009). These processes generally capture a large amount of solid particles, which may include plastic debris depending on the size and density of particles as well as the size of the screen bars and filters (Stephenson 2009). The degree to which these processes can specifically remove microplastics has recently been investigated by Talvitie et al. (2015), who found that the number of microfibres in wastewater influent reduced from 180 fibres per litre to 14.2 and 13.8 fibres per litre after primary and secondary sedimentation processes, respectively.

To investigate the fate of microplastics during treatment processes and the role of wastewater as a source of microplastics, it is necessary to estimate the mass of microplastics entering WWTPs and the amount released with wastewater effluent. Recently, there have been a number of pilot studies assessing the load of microplastic in WWTPs in Europe, Russia and the USA (HELCOM 2014; Magnusson & Norén 2014; Schneiderman 2015), with the results summarized in Table 2. One such study demonstrated that mechanical, chemical and biological treatment can remove about 99% of microplastics, from 15,000 particles/L in the influent to 8 particles/L in the effluent (Magnusson & Norén 2014). Likewise, in a Russian WWTP, the number of microplastics (textile fibres) reduced from 467 particles/L in the influent to 16 particles/L in the effluent (~96% reduction in microplastics after secondary treatment) (HELCOM 2014). While this is a promising finding, given a volume of treated wastewater of 2.18 million m$^3$/d from the above Russian WWTP (Vodokanal 2015), over $3.4 \times 10^9$ particles/L are still discharged to the receiving environment every day. A recent study on a WWTP in Finland also reported an average concentration of 4.9 and 8.6 microplastic fibres and particles, respectively, per litre in tertiary treated effluent (~98% removal after tertiary treatment) (Talvitie et al. 2015). Therefore, a daily discharge of $1.3 \times 10^9$ fibres and $2.3 \times 10^8$ particles to the receiving environment can be expected from that one WWTP, given a flow rate of around 270 million litres of wastewater per day (Talvitie 2015).
<table>
<thead>
<tr>
<th>Study</th>
<th>Study country</th>
<th>Sampling method</th>
<th>Size range of sampled particles (μm)</th>
<th>Identification method</th>
<th>Dominant shape &amp; type of detected microplastics</th>
<th>Number of microplastics influent</th>
<th>Number of microplastics in final effluent</th>
<th>Treatment type</th>
<th>Removal rate</th>
</tr>
</thead>
<tbody>
<tr>
<td>Talvitie et al. (2015)</td>
<td>Finland</td>
<td>Filtration device</td>
<td>20–200</td>
<td>Visual observation &amp; FT-IR spectroscopy</td>
<td>Synthetic particle and textile fibres</td>
<td>430 synthetic particle/L, 180 textile fibres/L</td>
<td>8.6 synthetic particles/L, 4.9 textile fibres/L</td>
<td>Tertiary</td>
<td>98%</td>
</tr>
<tr>
<td>Dris et al. (2015b)</td>
<td>France</td>
<td>Automatic sampler</td>
<td>ND</td>
<td>Visual observation</td>
<td>Fibres</td>
<td>260–320 particles/L</td>
<td>14–50 particles/L</td>
<td>Primary &amp; biological treatment</td>
<td>84–94%</td>
</tr>
<tr>
<td>Murphy et al. (2016)</td>
<td>Scotland</td>
<td>Steel bucket and steel sieve</td>
<td>&gt;65</td>
<td>Visual observation &amp; FT-IR spectroscopy</td>
<td>Fibres (polyester, polyamide, acrylic)</td>
<td>15.7 particles/L</td>
<td>0.3 particles/L</td>
<td>Secondary</td>
<td>98.4%</td>
</tr>
<tr>
<td>Carr et al. (2016)</td>
<td>USA</td>
<td>Sieving</td>
<td>20–400</td>
<td>Visual observation &amp; FT-IR spectroscopy</td>
<td>Irregularly shaped polyethylene</td>
<td>1 particle/L</td>
<td>~0.0007 particles/L</td>
<td>Tertiary</td>
<td>99.9%</td>
</tr>
<tr>
<td>Magnusson &amp; Norén (2014)</td>
<td>Sweden</td>
<td>Filtration</td>
<td>&gt;300</td>
<td>Visual observation &amp; FT-IR spectroscopy</td>
<td>Fibres and fragment</td>
<td>15 particles/L</td>
<td>8.3 particles/L</td>
<td>Mechanical, chemical &amp; biological</td>
<td>99.9%</td>
</tr>
<tr>
<td>HELCOM (2014)</td>
<td>Russia</td>
<td>A filtration device</td>
<td>20–300</td>
<td>Visual observation</td>
<td>Fibres &amp; synthetic particle</td>
<td>467 fibres/L, 160 synthetic particles/L</td>
<td>16 fibres/L, 7 synthetic particles/L</td>
<td>Mechanical treatment and purification</td>
<td>96%</td>
</tr>
<tr>
<td>Browne et al. (2011)</td>
<td>Australia</td>
<td>Glass bottle (750 ml), filtration</td>
<td>ND</td>
<td>Visual observation &amp; FT-IR spectroscopy</td>
<td>Fibres</td>
<td>ND</td>
<td>1 particle/L</td>
<td>Tertiary</td>
<td>ND</td>
</tr>
</tbody>
</table>

ND: Not detected; FT-IR: Fourier transform infrared spectroscopy.
et al. 2015). Carr et al. (2016) reported a removal efficiency of 99.9% of microplastics in tertiary WWTPs in the USA, with a total daily discharge of $9.3 \times 10^3$ microplastics. It should be noted that the lack of a uniform and standardized method for sampling, sample processing and analysis of microplastics may lead to non-comparable results. The methods used by some studies that detect microplastics in WWTPs are shown in Table 2.

Thus, while 96 to 99.9% is a significant reduction, the small percentage remaining in the effluent is likely to nevertheless represent an important source of microplastics to the aquatic environment given the large volume of effluent produced annually.

Currently, WWTPs are not specifically designed to remove microplastics from the wastewater effluent (Schneiderman 2015). However, the results of recent studies showed that the majority of microplastics are captured through skimming of floating solids during the primary treatment stage (Carr et al. 2016; Murphy et al. 2016). In this process microplastics removal is more dependent on density rather than size, as lower density microplastics may float and be removed by surface skimming (Carr et al. 2016). Moreover, there is evidence that tertiary treatment showed greater removal of microplastics compare to secondary and primary (Talvitie et al. 2015). Analyzing the final effluent of a German WWTP revealed that advanced filtration could only remove 93% of microplastics in the size range of $<500 \mu$m (Mintenig et al. 2014) as cited in Duis & Coors (2016)). More research is needed to assess the capacity of WWTPs to capture microplastics.

The physiochemical properties of microplastics can be altered during treatment processes. A recent study provides evidence regarding the presence of a layer of biofilm on the observed wastewater-based microplastic. These biofilms may influence the surface properties and relative density of microplastics (Carr et al. 2016), and consequently the fate of wastewater-based microplastics in the aquatic environment. Further, Lobelle & Cunliffe (2011) showed that biofilm can significantly modify the physiochemical properties of plastic debris, by making them less hydrophobic and more buoyant. Further research is required to study physical and chemical modifications of microplastics during wastewater treatment processes.

When considering the fate of microplastics in WWTPs, another important issue is what happens to the retained microplastics. It has been reported that a large proportion of the captured microplastics will be retained in the sludge phase (Magnusson & Norén 2014), with a number of studies finding high concentrations of microplastics in WWTP sludge samples (Brandsma et al. 2014; Magnusson & Norén 2014; Carr et al. 2016). Carr et al. (2016) reported 5 microplastic particles in 5 g of biosolids samples from a WWTP, which based on the daily production of biosolids, $1.9 \times 10^9$ particles were estimated to be removed daily with the biosolids. Thus, the use of treated sewage sludge as a biosolid may be an important source of microplastics to the environment given their wide application in agriculture and forestry for soil amendment (Rillig 2012; Dris et al. 2015c; Eerkes-Medrano et al. 2015; Rocha-Santos & Duarte 2015). It is also suggested that a proportion of microplastics in soil may eventually enter the aquatic environment via runoff after heavy rain, irrigation or through transportation by the wind (Hammer et al. 2012; Rochman et al. 2015).

INTERACTION OF MICROPLASTICS WITH TROCS AND METAL IONS

Given that wastewater contains a large number of organic and inorganic pollutants, it is likely that microplastics can interact with the variety of contaminants present in wastewater. This can potentially make microplastics carriers of wastewater-based contaminants, including industrial chemicals, pharmaceuticals, hormones, pesticides and metal ions, and thus modify their environmental fate. Using the textile industry as an example, substantial amounts of chemicals are used in bleaching, dyeing and finishing processes, thus auxiliary chemicals and heavy metals such as Cr, As, Cu and Zn can be released into the wastewater (Carmen & Daniela 2012; Ghaly et al. 2015) and can potentially interact with the microfibres associated with textiles.

The interaction of microplastics and metal ions such as Al, Fe, Mn, Cu, Zn, Pb, Ag, Cd, Co, Sr, Mo, Sb, Sn, and Ni and the capacity of plastics to act as a ‘transport vector’ of toxic metals has been extensively reported in the literature (Ashton et al. 2010; Holmes et al. 2012; Nakashima et al. 2012; Rochman et al. 2014). Metallic sorption can occur in both aged/weathered and virgin microplastics. It was assumed that the sorption mechanism in weathered microplastics would occur through the interaction of bivalent cations and oxyanions with the charged regions of modified plastic surfaces (Holmes et al. 2012, 2014). However, it was demonstrated that the interaction of virgin microplastics and trace metals (Cd, Co, Ni, Pb) occurs through the rapid adsorption of existing organic matter in water to the surface of the microplastics, leading to the acquisition of a negative charge from the organic matter function groups, due to the
fact that virgin plastics do not exhibit inherent charge or acid-base behaviour (Holmes et al. 2014). Therefore, the potential interaction of microplastics with metal ions in wastewater, as well as the common treatment chemicals, such as aluminium and iron used for flocculation-coagulation processes (Ukwe et al. 2014), may be an issue that needs further investigations.

In both the laboratory and the field, microplastics have been shown to sorb a wide range of TrOCs at many orders of magnitude higher than the surrounding water, with the reported concentrations ranging from 1 to 10,000 ng/g (Rios et al. 2010; Koelmans et al. 2013). Further, microplastics are able to accumulate more contaminants in highly polluted environments, such as industrial areas (Frias et al. 2010; Hirai et al. 2011; Karapanagioti et al. 2011; Antunes et al. 2013; Mizukawa et al. 2015). Thus, it is expected that microplastics in wastewater will behave similarly and will be able to accumulate high concentrations of TrOCs and metals. The sorption of some organic pollutants including dichlorodiphenyltrichloroethane (DDT) and phenanthrene to polyethylene microbeads extracted from shower gel has recently been showed to be comparable with the sorption capacity of polyethylene originating from other sources (Napper et al. 2015). Moreover, there is evidence of the interaction of microplastics and wastewater derived TrOCs, such as perfluorooctanesulfonate (PFOS) and perfluorooctanesulfonamide (PFOSA), with perfluorochemicals found frequently in municipal and industrial wastewater and sewage sludge (Ma & Shih 2010; Wang et al. 2015). To date, research on the interaction of microplastics and TrOCs has mainly focused on the marine environment, and more recently fresh water and estuarine environments, but the sorption behaviour and chemical fate of microplastics in wastewater still remains unexplored.

**Sorption capacity of microplastic for TrOCs**

The sorption of TrOCs to microplastics polymers is likely to occur through (i) adsorption and (ii) adsorption mechanisms, which are strictly dependent on the physico-chemical characteristics of both TrOCs and microplastic (Mato et al. 2001; Matsuzaawa 2010; Rochman et al. 2013a). Thus, different microplastics are expected to show different sorption affinities for different types of TrOCs.

Adsorption is assumed to be the general sorption mechanism that would occur not only through the interaction of hydrophobic surfaces of the polymer and TrOCs (Matsuzaawa 2010) (non-polar surface interaction/hydrophobic interaction), but also by specific interactions, such as electrostatic interactions and hydrogen bonding (Pan & Xing 2008). The polymer adsorption capacity to uptake TrOCs is likely to be positively correlated with the polymer surface area, which provides more adsorption sites at the surface of the polymer (Teuten et al. 2007; Fotopoulou & Karapanagioti 2012; Rochman et al. 2013a; Rochman 2015). Thus, the smaller the microplastic size, the more surface area and consequently the greater the sorption capacity (Rochman 2015). The surface interaction is likely to reach a threshold once the adsorption sites are saturated, and the time required to reach saturation may also vary depending on the polymers and TrOCs (Rochman et al. 2013a).

Absorption, on the other hand, refers to the diffusion of contaminants into the matrix of the sorbent, which is strongly associated with the polymer molecular structure (Rochman et al. 2013a). Polymers have been classified as rubbery and glassy based on their molecular structure (Crawford 1998). The structure of glassy polymers is more condensed, with little void spaces, whilst rubbery polymers exhibit greater flexibility, with large amounts of free volumes between molecules (Teuten et al. 2009; Bakir et al. 2012). It has been shown that rubbery polymers, such as polyethylene and polypropylene, have a greater sorption capacity for TrOCs such as phenanthrene, DDT, polychlorinated biphenyl and polycyclic aromatic hydrocarbons compared to glassy polymers such as polyvinyl chloride (PVC) (Bakir et al. 2012; Rochman et al. 2013a). Consequently, it is obvious that the sorption capacity of microplastics for TrOCs varies appreciably among different types of polymer depending on their surface area, crystallinity and diffusivity (Rochman et al. 2013a; Rochman 2015).

While specific types of polymer may show a high tendency to sorb a specific class of TrOCs, this may not extend to all classes of TrOCs (Rochman et al. 2013a; Rochman 2015). The chemical properties of TrOCs such as hydrophobicity (octanol-water partition coefficient, log K<sub>ow</sub>) and molecular weight (Smedes et al. 2009) can also affect the sorption rate and the equilibrium time (Rochman et al. 2013a).

Other factors that are likely to affect the partition behaviour and sorption rate of a TrOC to microplastics are (i) the concentration of organic matter and (ii) the presence of other contaminants in the system. The relative importance of organic matter compared to microplastics for sorption of hydrophobic TrOCs has been recently suggested using a multimedia model (Gouin et al. 2011). Based on an environmentally realistic volume ratio of polyethylene to organic matter (1.26 × 10<sup>7</sup>) for the marine coastal system, it has been predicted that the majority of TrOCs, even those with...
A high polyethylene-water partition coefficient (log K_{PE-W} > 7), will tend to partition more (>99%) to organic matter than polyethylene (Gouin et al. 2011). Thus, the organic matter-microplastic volume ratio may also affect the sorption rate of TrOCs to microplastics. This is particularly relevant for wastewater-based microplastics, as wastewater can contain high concentrations of organic matter, microplastics and TrOCs.

The competition behaviours between various TrOCs to sorb to microplastics can also influence the sorption rate of a TrOC to microplastics (Bakir et al. 2012). Given that wastewater will contain a complex mixture of TrOCs, it is likely that the sorption capacity of TrOCs by microplastics in wastewater would be lower than expected based on single compound experimental studies (Bakir et al. 2012). More research is required to investigate sorption behaviour in real but more complex matrices, such as WWTP effluent.

**A RISK-BASED PERSPECTIVE OF BIOLOGICAL EFFECTS OF MICROPLASTICS ON AQUATIC ORGANISMS**

To evaluate the risk that microplastics may pose to aquatic organisms, it is important to understand to what extent microplastics and their associated contaminants contribute to adverse effects and subsequent health outcomes in aquatic organisms. Toxicity may be related to acute or chronic effects through short or long-term exposure to microplastics and/or associated contaminants (Hammer 2008).

**Biological effects of microplastics themselves**

The adverse health effects from exposure to microplastics can result from physical and/or chemical effects (Wright et al. 2013b). According to a Marine Strategy Framework Directive Report, the possible direct detrimental effects of ingested microplastics include interference with enzyme production, nutrient dilution, reduced growth rate, decreased steroid hormone levels, delayed ovulation, reproductive failure and death (Galgani et al. 2010). Microplastics can also cause physical damage through accumulation in the digestive tract following ingestion (Wright et al. 2013b), although the amount of microplastics necessary to produce sub-lethal or lethal effects varies as the process is influenced by the type and sensitivity of the organism (Wright et al. 2013a; Wright et al. 2013b), exposure duration (Eisler 2000), the daily uptake dose of microplastics through normal food ingestion, the egestion of microplastics through excrement and the retention time of microplastics in the organism’s body, as well as the type, size and shape of ingested microplastics (Huang et al. 2010; Wright et al. 2013b). To date, there is limited work assessing the risks after exposure to a range of microplastic concentrations (Syberg et al. 2015). Lee et al. (2015) reported reduced survival of copepods (Tigriopus japonicas) exposed to 0.05 and 0.5 μm microplastic beads at concentrations of 1.25 and 25 mg/L, respectively, and no effect of 6 μm beads at concentrations as high as 25 mg/L.

The negative effects of microplastics on exposed organisms have been reported even after short-term exposure duration. For instance, aggregated polyethylene microplastics were observed in digestive gland lysosomes in blue mussels (Mytilus edulis), which caused a considerable decrease in lysosomal membrane stability at the cellular level by increasing granulocytoma formation during 96 h exposure as a cellular self-protection response to contaminants (von Moos et al. 2012). The physical impact of unpolymerized polyvinyl chloride (UPVC) microplastics within the size range of 20–100 μm was also investigated on lugworms, where 30% and 50% depletion of lipid reserves was observed after exposure to 1% and 5% PVC, respectively, and was directly associated with a reduction in feeding activity due to the long retention time of microplastics in the organism’s gut (Wright et al. 2015a). Similarly, a recent study with copepods exposed to 20 μm polystyrene microbeads over 24 h showed reduced energy supplies and growth rate due to interference with feeding capacity at a concentration of 75 particles mL$^{-1}$, which was 10% of their available food (Cole et al. 2015).

It should be noted that some studies have conversely shown that microplastics can have negligible effects on some organisms. Hamer et al. (2014), recently demonstrated limited physical effects on isopods (Idotea emarginata) after 6 months of continuous exposure to three different microplastic types (polystyrene microbeads, polystyrene fragments and polyacrylic fibres), under low and high concentrations, with 100% of the ingested microplastics detected in the feces and no accumulation in the organism’s gut. Consequently, it is possible that some aquatic organisms may exhibit no negative effects to any concentrations of microplastics. Therefore, improved understanding of the adverse effect of microplastics requires a comprehensive risk assessment procedure that considers all factors mentioned above. Some studies on the effects of microplastics on various organisms, and the factors influencing biological effects, are summarized in Table 3.
Table 3 | Some reviews of the biological effects of microplastics on various organisms

<table>
<thead>
<tr>
<th>Study</th>
<th>Exposed Organism</th>
<th>Polymer type &amp; size</th>
<th>Concentration</th>
<th>Retention time</th>
<th>Exposure duration</th>
<th>Biological effects</th>
</tr>
</thead>
<tbody>
<tr>
<td>Browne et al. (2008)</td>
<td>Mussel (M. edulis)</td>
<td>Polystyrene (3–9.6 μm)</td>
<td>0.51 g/L</td>
<td>Over 48 days</td>
<td>3 hours</td>
<td>No toxicological effects were observed</td>
</tr>
<tr>
<td>Cole et al. (2015)</td>
<td>Copepod (C. helgolandicus)</td>
<td>Polystyrene (20 μm)</td>
<td>75,000 particles/L</td>
<td>NA</td>
<td>24 hours and 9 days</td>
<td>Prolonged exposure to microplastics led to: reduction of carbon biomass that may cause energetic deficiencies and lower growth; production of smaller eggs; and reduced hatching success</td>
</tr>
<tr>
<td>Hamer et al. (2014)</td>
<td>Isopods (I. emarginata)</td>
<td>Polystyrene (1–100 μm) microbeads and fragments, and polyacrylic fibres (20–2,500 μm)</td>
<td>No retention was observed</td>
<td>6 to 7 weeks</td>
<td>No adverse effects on survival or growth rate were observed</td>
<td></td>
</tr>
</tbody>
</table>
| Peda et al. (2016)    | European sea bass (D. labrax) | PVC pellets (>300 μm) | 0.1% (W/W) of their diet | NA           | 30, 60, and 90 days | • After 30 and 60 days of exposure: the majority of studied organisms showed moderate structural changes such as shortening and swelling of villi, circulatory changes and worsening of inflammatory changes  
• After 90 days: half of the individuals showed serious changes such as reduction of perivisceral fat and reduction of intestinal function  
• No mortality was observed after 90 days' exposure |

(continued)
<table>
<thead>
<tr>
<th>Study</th>
<th>Exposed Organism</th>
<th>Polymer type &amp; size</th>
<th>Concentration</th>
<th>Retention time</th>
<th>Exposure duration</th>
<th>Biological effects</th>
</tr>
</thead>
<tbody>
<tr>
<td>Lee et al. (2015)</td>
<td>Copepod (T. japonicas)</td>
<td>Polystyrene (three sizes: 0.05, 0.5 and 6 μm)</td>
<td>Acute toxicity test: 0, 6, 13, 31, 63, 187, 250 and 313 mg/L Chronic toxicity test: 0.125, 1.25, 12.5 and 25 mg/L</td>
<td>NA</td>
<td>96 hours (Acute effects) 14 days (chronic effects)</td>
<td>Acute effects: • No mortality was observed with any of three different sizes at the highest concentration (acute NOAEC &gt; 313 mg/L) Chronic effects: • Mortality observed at concentrations &gt; 1.25 mg/L for the 0.05 μm particles (LOAEC = 1.25 mg/L), with no effects on fecundity • Decrease in fecundity in 0.5 μm particles at the lowest concentrations (LOAEC = 0.125 mg/L), and in survival at the highest concentration tested • Decrease in fecundity at the lowest concentration tested (LOAEC = 0.125 mg/L) and no effect on survival with the 6 μm beads</td>
</tr>
<tr>
<td>Davarpanah &amp; Guilhermino (2015)</td>
<td>Microalgae (T. chuii)</td>
<td>Polyethylene microspheres (1–5 μm)</td>
<td>0.046 to 1.472 mg/l</td>
<td>NA</td>
<td>96 hours</td>
<td>• No significant effects of microplastics on the growth rate of T. chuii were found up to a concentration of 1.472 mg/l</td>
</tr>
<tr>
<td>von Moos et al. (2012)</td>
<td>Mussel (M. edulis)</td>
<td>Polyethylene (0–80 μm)</td>
<td>2.5 g/L of sea water</td>
<td>NA</td>
<td>Up to 96 hours</td>
<td>Strong inflammatory response, disruption of lysosomal membrane stability</td>
</tr>
<tr>
<td>Wright et al. (2015a)</td>
<td>Lugworm (A. marina)</td>
<td>UPVC (mean 230 μm)</td>
<td>0.5–5% by weight (mixed with sediment)</td>
<td>Prolonged residence time</td>
<td>28 days</td>
<td>Reduced feeding activity, reduced lipid and available energy reserve, inflammation</td>
</tr>
</tbody>
</table>
With an estimated 40.5–215 mg of polyethylene (PE) microbeads/person/d in the UK (Napper et al. 2015), and assuming an average wastewater production of 150 L/person/d, the expected concentration of PE in UK wastewater is 0.27–1.4 mg/L. This is in range of the lowest observable adverse effect concentration (LOAEC) reported in the UK, indicating a potential risk to the aquatic environment from wastewater-based microplastics.

Another potential risk of microplastics for aquatic organisms may be from binding of microorganisms, such as pathogens and bacteria, to the microplastic surface, since wastewater-based microplastics have been shown to be covered with a layer of biofilm (Cart et al. 2016). Thus, more research is required to understand the possible interactions of microplastics with existing microorganisms in wastewater and during wastewater treatment (such as biological processes), as these microorganism communities may be transported to the receiving environment.

**Biological effects from contaminants associated with microplastics**

Considering the evidence of TrOC sorption to microplastics, it is conceivable that the toxicity of microplastics to organisms may be enhanced by the toxic effects of sorbed TrOCs. Indeed, studies have suggested that sorbed TrOCs can migrate from microplastics to exposed aquatic organisms (Browne et al. 2015; Rochman et al. 2015b; Chua et al. 2014).

A recent study by Luis et al. (2015) demonstrated that the presence of polyethylene microplastics caused a significant increase in the toxicity of Cr (VI) in marine fish (Pomatoschistus microps) compared to chromium alone. Likewise, Avio et al. (2015) reported significant transfer of pyrene to marine mussels (Mytilus galloprovincialis) through ingestion of contaminated polystyrene and polyethylene microplastics.

Some studies have suggested that the transfer of sorbed TrOCs from contaminated microplastics to the organism is likely to be a less significant source compared to the other exposure pathways (Gouin et al. 2011; Koelmans et al. 2013). Using a biodynamic model to simulate the uptake of TrOCs on plastics in a sediment-water system, Koelmans et al. (2013) found that the role of microplastics for TrOC bioaccumulation in marine lugworm was insignificant compared to other TrOC exposure pathways, such as dermal uptake and uptake from sediment. Moreover, a recent study has also demonstrated much lower uptake of polybrominated diphenyl ether (PBDE), and subsequently lower health risks, in amphipods’ tissue after 72 h exposure to PBDE sorbed to polyethylene microplastics compared to those exposed to PBDE alone (Chua et al. 2014). One proposed mechanism that can explain this result is the strong tendency of PBDEs to remain in/on the microplastics rather than being taken up into amphipod tissue. Consequently, such interactions may also reduce the bioavailability of sorbed TrOCs in organisms (Gouin et al. 2011; Hirai et al. 2011). While this study demonstrated the reduced contribution of hydrophobic TrOCs associated with microplastics to the organism’s health compared to the direct exposure pathway, it is thought that microplastics still act as a carrier of TrOCs to organism tissue (Chua et al. 2014).

Thus, it appears that the influence of microplastics on the bioavailability of contaminants and their potential to act as a vector of contaminants to organisms, as well as the degree to which the sorbed contaminants pose a risk, may vary between different types of microplastics, contaminants and organisms (Koelmans et al. 2013). Further, the potential of sorbed contaminants to migrate from microplastics to the organism can be influenced by factors including the kinetics of TrOC desorption between plastic and tissues and the residence time of contaminated microplastics in the organism’s body (Andrady 2011; Chua et al. 2014). There is evidence that the desorption and incorporation of TrOCs from microplastics to the organism’s tissue can be facilitated by the organism’s digestive fluid through desorbing sorbed TrOCs on contaminated microplastics, which may be enhanced by increasing the retention time of microplastics in the digestive tract (Bakir et al. 2014; Chua et al. 2014).

The risks associated with microplastics may also come from the chemicals embedded into the plastic polymer during manufacture, particularly if the components are not stable and can leach into the organism’s body after ingestion. The biological effects of some additives, including phthalates, bisphenol A, alkylphenols and PBDEs, on different wildlife species has been widely reported (Foster et al. 2011; Oehlmann et al. 2009; Jarmolowicz et al. 2013). The concerns regarding leaching of plastic additives in aquatic organisms is associated with microplastics derived from the breakdown of larger plastic objects in the environment, but there is limited experimental data regarding adverse biological effects from additives incorporated in wastewater-based microplastics, such as microbeads and fibres. For example, inclusion of PBDEs as a flame retardant in textiles and clothing is reported extensively (Rahman et al. 2001; Frederiksen et al. 2009). Brominated and
chlorinated phthalocyanines are other fire retardant additives that can be introduced to enhance the thermal stability of fabric (Gladunova et al. 2013), and can be toxic to marine organisms (Carmen & Daniela 2012). However, the leaching rate of the above contaminants from wastewater-based microfibres, as well as the potential risks for the variety of aquatic organisms, is still unexplored.

CONCLUSIONS AND FUTURE AREAS OF RESEARCH

The concerns regarding microplastics in WWTP effluent have rapidly increased in the last few years, and preliminary research has confirmed wastewater as a source of microplastics in the receiving environment, with the amount contributed depending on the level of wastewater treatment. Due to the fact that even advanced treatment technologies cannot always remove 100% of microplastics (Schneiderman 2015; Talvitie et al. 2015), preventing the release of microplastics to wastewater influent would be the best action to reduce the discharge of wastewater-based microplastics to the aquatic environment. While preventive measures have been taken to remove some microplastic beads from personal care products, this is more challenging for other sources of microplastics, particularly as synthetic clothing is also expected to be an important source of microfibres in wastewater. Consequently, further research on developing a sound mitigation strategy to prevent release to the environment would be beneficial.

This review has identified that there are still research gaps in this recently developing field of research, specifically: (i) to investigate the contribution of wastewater discharge from WWTPs to microplastics in the aquatic environments compared to other potential sources, (ii) to understand the behaviour of microplastics in WWTPs, and their physical and chemical modifications through wastewater treatment processes, and (iii) to understand the impact of sorption to microplastics on contaminant fate and toxicity in a more complex mixture that resembles wastewater. This research is necessary to better understand the relative contribution and significance of wastewater-based microplastics.

Since some wastewater-based microplastics from WWTPs can be transported from the discharged area and dispersed widely in the aquatic environment, collaborations at regional and global levels are required to establish management frameworks in terms of the maximum levels of microplastics in the wastewater effluent and to reduce the use of treated sewage sludge or biosolids with high concentrations of microplastics.

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