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Review

Microplastics in aquatic environments: Implications for Canadian ecosystems☆

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ABSTRACT

Microplastics have been increasingly detected and quantified in marine and freshwater environments, and there are growing concerns about potential effects in biota. A literature review was conducted to summarize the current state of knowledge of microplastics in Canadian aquatic environments; specifically, the sources, environmental fate, behaviour, abundance, and toxicological effects in aquatic organisms. While we found that research and publications on these topics have increased dramatically since 2010, relatively few studies have assessed the presence, fate, and effects of microplastics in Canadian water bodies. We suggest that efforts to determine aquatic receptors at greatest risk of detrimental effects due to microplastic exposure, and their associated contaminants, are particularly warranted. There is also a need to address the gaps identified, with a particular focus on the species and conditions found in Canadian aquatic systems. These gaps include characterization of the presence of microplastics in Canadian freshwater ecosystems, identifying key sources of microplastics to these systems, and evaluating the presence of microplastics in Arctic waters and biota.

1. Introduction

Plastic is a general term that refers to a family of organic polymers derived from petroleum sources, including polyvinylchloride (PVC), nylon, polyethylene (PE), polystyrene (PS), and polypropylene (PP) (Vert et al., 2012). Common plastic polymers include PP, PE, low-density polyethylene (LDPE), and polyacrylates (Imhof et al., 2013; Frias et al., 2014; Hidalgo-Ruz et al., 2012). Plastic production and use has increased steadily over the past 50 years, with global production reaching over 300 million tonnes in 2014 (Plastics Europe, 2015). These usage patterns suggest that plastic production and quantities of plastics (including microplastics) in aquatic environments will likely continue to increase over time (Andrady, 2011; Galgani et al., 2010). The defined size of a particle constituting a "microplastic" varies, but an upper limit of 5 mm is generally agreed upon in the literature, and many researchers use 0.5 or 1 mm as the cut-off between macro or mesoplastic and microplastic (Andrady, 2011; Cole et al., 2011). Prior to about 2010, studies typically investigated plastic particles ranging from 1 to 5 mm, and data relevant to smaller particle sizes are scarce prior to that time (Claessens et al., 2011).

Microplastics may pose a risk to aquatic environments due to their documented ubiquity in marine ecosystems, long residence times, and propensity to be ingested by biota (Arthur et al., 2008a; Galgani et al., 2010; Andrady, 2011). While studies and reviews on plastic pollution in the marine environment are increasingly common, to date, few studies have assessed the presence, fate, and effects of microplastics in freshwater environments. Even fewer studies have been completed in Canada, despite the fact that 7% of the world's renewable freshwater is contained within these water bodies (Environment Canada, 2012). While the presence, sources, fate, and effects of microplastics have not been well characterized in freshwater systems, evidence from the marine environment suggests that microplastics could be considered contaminants of emerging concern (Wagner et al., 2014; Eerkes-Medrano et al., 2015).

In response to growing concerns from the scientific community regarding microplastics, the Netherlands, Austria, Luxembourg, Belgium, and Sweden issued a joint statement to the European Union Environment Ministers calling for a ban on microplastics in

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personal care products (UNEP, 2015). Likewise, bans on microplastics in cosmetic products were enacted in Illinois (Hitchings, 2014), California (Doughty and Eriksen, 2014), and New York (Office of the New York State Attorney General, 2015), and finally were signed into US federal law (*Microbead-Free Waters Act of 2015*) in January 2016. In mid-2015, the Government of Canada proposed to include microbeads on the List of Toxic Substances under the *Canadian Environmental Protection Act, 1999* (Environment Canada, 2015).

There were several objectives for this review, all of which were approached with Canadian aquatic ecosystems as the primary focus. First, we sought to summarize: a) inputs and composition of microplastic materials in the Canadian aquatic environment; b) their environmental fate and behaviour, including previously reported environmental levels of various size classes in surface waters, vertically throughout the water column, and in sediments; and c) pathways of ingestion by biota (direct ingestion and trophic transfer) and persistence in tissues. Second, the toxic potential of microplastics in aquatic biota, including invertebrates, fish, and mammals was summarized. Finally, knowledge gaps were identified to inform future work on the aquatic toxicity of microplastics.

2. Methods

To conduct this review, combinations of keywords (i.e., microplastic(s), microbead(s), freshwater, aquatic, Canada, presence, effect(s), toxicity) were entered into Stantec's Research and Development Resource eLibrary holdings (including articles from Elsevier, Scopus, JSTOR, EBSCOhost, and Science Direct). These holdings contain over 5700 journals available in full-text format. In addition, supporting searches were conducted using Google Scholar and the University of Manitoba libraries guest services, with literature considered up to and including articles available as of January 2016. Articles were selected for inclusion in this review based upon the relevance of the information to the topic of microplastics in the Canadian aquatic environment, as judged by the authors, as well as inclusion in or agreement with other peer-reviewed articles. Articles were usually considered if self-identified as studying "microplastics", but a threshold of 5 mm was used for excluding macroplastics. Studies in which microplastics were quantified in Canadian waterbodies were considered of greatest interest. Where information specifically pertaining to Canadian ecosystems was not available, data from international studies were included to help establish the current level of understanding of microplastics in aquatic environments. While the present review may not be an exhaustive summary of the literature, the authors believe it represents an accurate portrayal of the current state of knowledge regarding microplastics in the Canadian aquatic environment.

3. Sources and fate of microplastics in the aquatic environment

3.1. Sources of microplastics

In general, microplastics fall into two categories: they are either produced intentionally (e.g., microbeads, plastic production pellets) and called "primary microplastics" or are degraded from larger plastic to smaller pieces (e.g., fibres) and are called "secondary microplastics" (Cole et al., 2011; Gilman, 2013; Andersson, 2014). In Canada and globally, primary microplastics (often PE microbeads) have been added to a variety of personal care products, including toothpastes, shampoos, facial cleansers and moisturizers, cosmetics, and shaving products for emulsion stabilization, viscosity regulation, and skin conditioning (Cole et al., 2011; Derraik, 2002; Driedger et al., 2015; Leslie, 2014). A study of six brands of facial scrubs reported that between 4594 and 94,500 microbeads (164–327 μm in diameter) could be released into the wastewater stream per use of the products (Napper et al., 2015). Microplastics are also added to industrial cleaning products (e.g., scrubbers for removal of rust or paint) (Derraik, 2002; Cole et al., 2011), and pellets are used in production of plastic consumer goods (Mato et al., 2001; Turner and Holmes, 2015). Ultimately, any of these forms of plastic has the potential to end up in municipal wastewater and freshwater systems (Cole et al., 2011; Doughty and Eriksen, 2014; Leslie, 2014).

It has been proposed that freshwater systems can become contaminated by microplastics in one of three ways: 1) effluent discharge from wastewater treatment plants, 2) overflow of wastewater sewers during high rain events, and 3) run-off from sludge applied to agricultural land (Eriksen et al., 2013a). The directional flow of freshwater systems typically drives microplastics to river bottoms, lake bottoms, and the oceans, which become sinks. It has been estimated that approximately 80% of microplastics in oceans originate from land-based sources, and another 18% from aquaculture or fishing industries (Andrady, 2011; Cole et al., 2011). Storms and extreme weather events can also exacerbate the movement of microplastics from land into bodies of water (Cole et al., 2011). Most current wastewater treatment plants (WWTPs) are not designed to fully remove microplastics (Fendall and Sewell, 2009; Leslie et al., 2013). For example, in a Paris WWTP, raw influent contained 260–320 $\times$ 10^3 particles/m^3 that was reduced to 50–120 $\times$ 10^3 particles/m^3 after primary treatment, and 14–50 $\times$ 10^3 particles/m^3 in the final effluent. This represents removal of between 83 and 95% of microplastics (Dris et al., 2015a). Likewise, an assessment of a Helsinki WWTP reported ~97% removal of microplastic fibers and 98% removal of microplastic particles, but the effluent levels were still elevated relative to receiving waters, indicating that the WWTP represented a source of microplastics to the receiving environment (Talvitie et al., 2015). In Canada and the US, WWTPs are not currently required to monitor microplastics in effluents or influents (Driedger et al., 2015) and many WWTPs do not have any sort of advanced treatment (i.e., tertiary treatment), and therefore, optimized removal of microplastics would not be expected (CWWA, 2001; Office of the New York State Attorney General, 2015; Talvitie et al., 2015). Even when sewage sludge is applied to agricultural lands, plastic fibers can persist (i.e., 15 + years) and migrate off fields via runoff during storm events (Zubris and Richards, 2005).

3.2. Environmental fate and behaviour

Plastics consist of different polymers that can be buoyant, neutral, or sink, depending upon composition, density, and shape of the plastic (Fig. 1). PP and PE are typically low-density plastics that are expected to be relatively buoyant, while PVC, PS, polyester, and polyamide are considered high-density plastics that are more likely to sink (Browne et al., 2007; Cole et al., 2011). However, PE and PP can become higher density polymers as a result of addition of mineral fillers during production (Corcoran, 2015; Corcoran et al., 2015). Approximately half of manufactured plastics have a density higher than seawater (Ballent et al., 2013), and denser varieties, such as polyester, tend to submerge or even reach the sediment (Andrady, 2011). Turbulence and storm activity can cause (re-) suspension of high-density microplastics and redistribution throughout the water column (Moore et al., 2002; Lattin et al., 2004; Cole et al., 2011). Biofouling can occur, as can adsorption of clay minerals, increasing the density and weight of the microplastic particle and resulting in sinking to pelagic or benthic zones (Andrady, 2011; Cole et al., 2011; Corcoran, 2015; Corcoran et al.,

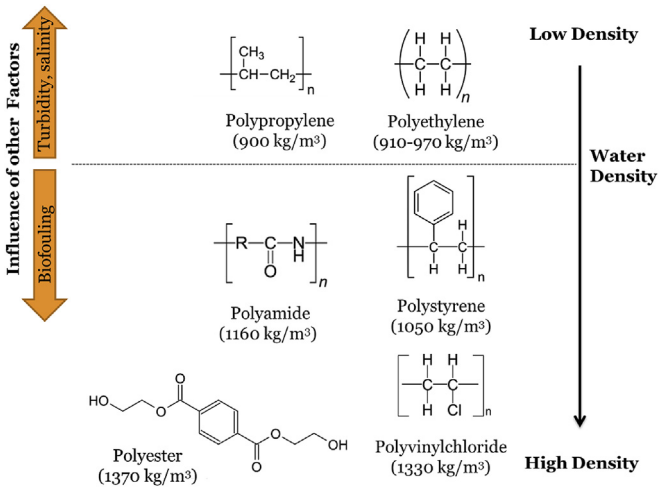

Fig. 1. Densities, structures, and expected distributions of different plastic polymers in the water column. Factors affecting buoyancy, and the direction of the change, are indicated with the arrows on the left.

2015). As a result, density of the parent material alone does not dictate the location of the particle within the aquatic environment.

Plastic can undergo a number of different types of degradation: biological, photo, thermal, mechanical, thermo-oxidative, and hydrolysis (Browne et al., 2007; Andrady, 2011). For LDPE, HDPE (high density polyethylene), PP, and nylon, degradation is primarily via UV-B photo-oxidation followed by thermo-oxidation. Degraded products can then be micro-sized or potentially even nano-sized (dimensions on the order of nanometers) (Galgani et al., 2010; Cole et al., 2011). These fragments can undergo further degradation (e.g., biological) where carbon in the matrix is converted to carbon dioxide and incorporated into biomass (Andrady, 2011). However, UV degradation of plastic floating in water is impeded by lower temperatures and oxygen levels relative to on-land, making conversion of macroplastic to microplastic much more rapid on beaches than in the water (Arthur et al., 2008b; Andrady, 2011; Corcoran et al., 2009). As a result, microplastics are considered relatively recalcitrant in aquatic environments, and degradation of plastic and leaching of additives is more likely along the shoreline than in open waters (Andrady, 2011; Cole et al., 2011). The time required to completely mineralize plastics is estimated to be on the order of hundreds to thousands of years (Barnes et al., 2009).

3.2.1. Abundance in aquatic systems

Microplastics are ubiquitous in marine environments (Eriksen et al., 2014; Norwegian Institute for Water Research, 2014) and widespread contamination of freshwater systems is likely inevitable (Ross and Morales-Caselles, 2015). Microplastics have been found in sediments, throughout the water column, and in digestive systems, respiratory structures, and tissues of marine organisms (Andrady, 2011; Depledge et al., 2013). Quantitative reporting of global abundances of microplastics has been limited by time- and labour-intensive sampling, remoteness of sites, and fine-scale analytical processes required (Arthur et al., 2008b; Ballent et al., 2013; Driedger et al., 2015). Microplastics will accumulate in coastal sediments, on the ocean floor, and at the sea surface. Due to the relative ease of accessibility and sampling, beaches have been most heavily surveyed and form the basis for much of the currently available information regarding the distribution of microplastics (Gilman, 2013).

As a result of the lack of standardized size definitions for microplastics, mesh and filter sizes used for collecting and analyzing samples vary between studies (Hidalgo-Ruz et al., 2012). Differences in collection techniques affect the abundances of microplastics reported, especially for the smallest particle sizes, and limits comparisons among studies. There is a need for standardized methods and size fractions to improve comparisons between studies as this area of research grows (Dris et al., 2015b). This is especially true for nanoplastics, which have not been a focus of research studies previously but could represent a considerable future challenge for monitoring and risk assessment (Galgani et al., 2010; Hidalgo-Ruz et al., 2012; Norwegian Institute for Water Research, 2014).

Microplastics tend to accumulate in proximity to cities or regions of relatively high human use (Kusui and Noda, 2003; Browne et al., 2011; Eriksen et al., 2013a; Depledge et al., 2013; Desforges et al., 2014; Yonkos et al., 2014; de Sá et al., 2015). However, there are also detectable levels present in very remote areas (Derraik, 2002; Gilman, 2013), such as a mountain lake in Mongolia (Free et al., 2014), Arctic Sea ice (Obbard et al., 2014) and deep-sea sediments (Van Cauwenberghe et al., 2013). Because physical factors (e.g., wind, salinity, temperature, precipitation, presence of impervious surfaces on-shore) strongly influence the distribution of microplastics and their final sinks, the density of human populations alone cannot be used to predict abundances of microplastics within a region (Browne et al., 2010; Gilman, 2013; Lusher et al., 2014; Yonkos et al., 2014).

3.2.1.1. Abundance in sediment. Microplastics in nearshore or offshore marine sediments have been characterized along many coastlines around the world (Table S1). Dekiff et al. (2014) observed relatively homogenous distribution of microplastics across a 500-m stretch of North Sea shoreline and found that abundances of microplastics did not correlate significantly with visible plastic debris along the beach sediments. The authors cautioned that the presence or absence of visible plastic should not determine the sampling sites selected, and that microplastic and macroplastic should be monitored separately (Dekiff et al., 2014). Likewise, Laglbauer et al. (2014) observed no differences in microplastics on Slovenian beaches with high or low human use, or between infralittoral or shoreline zones. Also, no primary microplastics were observed, meaning that all fragments and films observed on the sampled beaches likely resulted from the breakdown of larger pieces of plastic (Laglbauer et al., 2014).

In a study conducted in southern New England, researchers noted the presence of PS spherules in coastal waters ranging from 0.1 to 2 mm in diameter (Carpenter et al., 1972). Eight species of fish were observed consuming the spherules, leading to concerns of intestinal blockage and physical injury in smaller fish (Carpenter et al., 1972). Many years later, sediment samples were collected in the same region from three sites surrounding Halifax Harbour, Nova Scotia, including two protected beaches and one exposed to tidal action (Mathalon and Hill, 2014). There were interactive effects of tide level and site type (i.e., protected or exposed), but the three sites were similar, with abundances of microplastics between 20 and 80 pieces/10 g of sediment (Mathalon and Hill, 2014). In the Tamar Estuary of the North Atlantic Ocean, microplastics collected from shoreline sediments were dominated by denser polymers, including PVC, polyester, and polyamide. No relationship was observed between the abundance of microplastics and the proportion of clay in the sediments, but greater production of secondary microplastics was evident in areas with strong wave action and abrasion by sediment particles (Browne et al., 2010).

Deep-sea sediments have only been sampled a small number of times, so our understanding is fairly preliminary at this point. Sediments and coral were collected from deep-sea locations at depths up to 3500 m in the Mediterranean Sea, southwest Indian

Ocean, and northeast Atlantic Ocean in 2001 and 2012 (Woodall et al., 2014). Microplastics were detected in all samples, at an average abundance of 13.4 pieces/50 mL of sediment. Given the extent of deep-sea regions, these sediments could represent a considerable sink for microplastics, and might account for plastics "missing" previously from mass balance calculations (Woodall et al., 2014).

Relative to marine sediments, freshwater sediments have been only rarely sampled for microplastics (Table S1). In one of the few studies conducted in Canadian freshwater systems, ten sites were sampled in the St. Lawrence River between Lake St. Francis and Quebec City in 2013 (Castañeda et al., 2014). Microplastics between 0.4 and 2.16 mm were found along the entire length of river sampled, with mean and median quantities in sediment of 52 microbeads/m^2 and 13,832 ($\pm$13,677) microbeads/m^2, and a maximum density of 1.4×10^5 microbeads/m^2. The microbeads collected were consistent in shape and size with those found in the digestive tracts of *Neogobius melanostomas* (goby) reported in a 2008 study in the St. Lawrence (Castañeda et al., 2014). Sediment samples were collected from three estuary sites surrounding Halifax Harbour, Nova Scotia, including two protected beaches and one exposed to tidal action (Mathalon and Hill, 2014). There were interactive effects of tide level and site type (i.e., protected or exposed), but the three sites were similar, with abundances of microplastics between 20 and 80 pieces/10 g of sediment (Mathalon and Hill, 2014). The results of freshwater sediment studies suggest that freshwater bodies could accumulate microplastics, in addition to serving as a source of plastics to marine environments (Imhof et al., 2013).

Recently, several studies have been conducted to characterize the presence of microplastics in the Great Lakes. Plastics with diameters <5 mm were collected from sediments and beaches in Lake Ontario, in the Humber Bay region which receives inputs from Toronto via the Humber River (Corcoran et al., 2015). Over 4000 plastic pellets were collected during three sampling events in 2013, and these were generally similar in colour to those retrieved from the Humber River bank. There were between 5.98 and 9.45 g/m^2 of microplastics recovered from the shoreline and lake bottom sediments. The authors suggested that these plastics have been accumulating for approximately 38 years, based on sediment depths and accumulation rates, and the Humber River is likely a pathway for entry of microplastics into Lake Ontario (Corcoran et al., 2015). In another study, microplastics were detected in beaches around Lake Huron, but were not quantified independent of macroplastics. However, PE was identified as the predominant plastic, with the greatest abundances occurring near industrial areas (Zbyszewski and Corcoran, 2011). Ballent et al. (2016) collected sediment samples from Lake Ontario, and found microplastics in all samples, to depths of $\geq$15 cm in nearshore sediments and $\geq$30 cm in beach sediments. In Lake Erie, lake bottom sediments were found to contain up to 39 microplastic pieces/100 g, beach sediments contained up to 71 pieces/100 g, and tributary sediments contained up to 51 pieces/100 g (Dean et al., 2016). Though studies in Canadian waterbodies have been focused on several key regions, including the Great Lakes (Fig. 2), the studies to date suggest that microplastics can be found ubiquitously in Canadian systems.

3.2.1.2. Abundance in water. Reported abundances of microplastics in global surface waters vary considerably, from less than 1 piece/m^3 to thousands of pieces/m^3 (Table S-2). However, these particles are widespread and have been detected in marine, estuarine, and freshwater systems. In cruises of the northeast Atlantic Ocean, microplastics were found in 94% of samples collected, with black or blue fibers dominating the composition (Lusher et al., 2014). The estimated density of microplastics in that region of the Atlantic could be as high as 2.46 pieces/m^3 (Lusher et al., 2014), but it is currently unknown whether similar abundances would be expected in the northwest Atlantic Ocean near the coast of Canada.

Only a few studies have characterized the quantities of microplastics in surface waters of freshwater lakes and rivers. Surface waters were sampled in 2012 to characterize microplastics in the heavily urbanized region of the Great Lakes as these represent a potential source of microplastics to the St. Lawrence River and, ultimately, the North Atlantic (Eriksen et al., 2013a). The average abundance among Lake Superior, Lake Huron, and Lake Erie was 43,000 pieces/km^2, and the primary source of microplastics to these lakes seemed to be consumer products. This was based on proximity to urban areas and because the shape, size, colour, and elemental composition of the observed microbeads were consistent with polyethylene beads used in two national brands of facial cleanser (Eriksen et al., 2013a). In 2014, surface waters were sampled in Lake Ontario, Lake Erie, and urban streams entering Lake Ontario. These waters were found to contain between 90,000 and 6,700,000 particles/km^2, with fragments representing the dominant form of microplastics at most sampling locations (Helm et al., 2016). In Lake Winnipeg, all surface water samples collected in 2014 contained microplastics, with the greatest concentrations detected at the lake outflow, and elevated levels at the northern inflow compared to the southern inflow (Rennie and Anderson, 2016).

Arctic regions have generally been understudied and information relevant to these environments could be important for Canadian researchers and policy makers. One study of ice cores collected from the Arctic Ocean reported abundances of microplastics in ice ranging from 38 to 234 pieces/m^3. Polyamide, PP, PS, and PE were present in the cores, suggesting that sea ice could be a potential sink for microplastics and could account for some of the plastic that has been considered "missing" from mass balance calculations (Obbard et al., 2014).

3.2.1.3. Abundance in biota. Microplastics, and their associated contaminants, can be assimilated by aquatic biota in a number of ways: filter feeding, suspension feeding, inhalation at air-water surface, consumption of prey exposed to microplastics, or via direct ingestion (Baulch and Perry, 2014; Depledge et al., 2013). The ability of microplastics to enter food webs at different trophic levels is a particular concern, given the difficulty in removing microplastics from the environment once they are present (Barnes et al., 2009; Baulch and Perry, 2014; Doughty and Eriksen, 2014). Microplastics are in the same size range as plankton, creating even greater potential for mistaken identity and uptake by predators than in larger size ranges (i.e., particles >5 mm) (Browne et al., 2007). Bioavailability or likelihood of uptake of microplastics depends upon particle size, colour, and abundance in water, sediment, or biota. Currently, many of these factors are not well understood, and need to be better characterized in various environments.

The abundance of microplastics in wild biota has most often been reported in plankton or neuston samples (Table S-3). Plankton samples have been collected in the northeast Atlantic Ocean since the 1960s, and retrospective analysis of archived plankton and neuston samples indicates that microplastics were present in the earliest samples and have been increasing in the region over time (Lozano and Mouat, 2009).

Few studies have quantified microplastics in aquatic mammals and seabirds. One study collected scat from fur seals on Macquarie Island (South Pacific Ocean) and noted the presence of plastic particles ranging from 2 to 5 mm in diameter, although no pieces were observed that were between 0.5 mm (mesh size used) and 1 mm (Eriksson and Burton, 2003). It was determined that the plastics were secondary plastics, and the authors hypothesized that

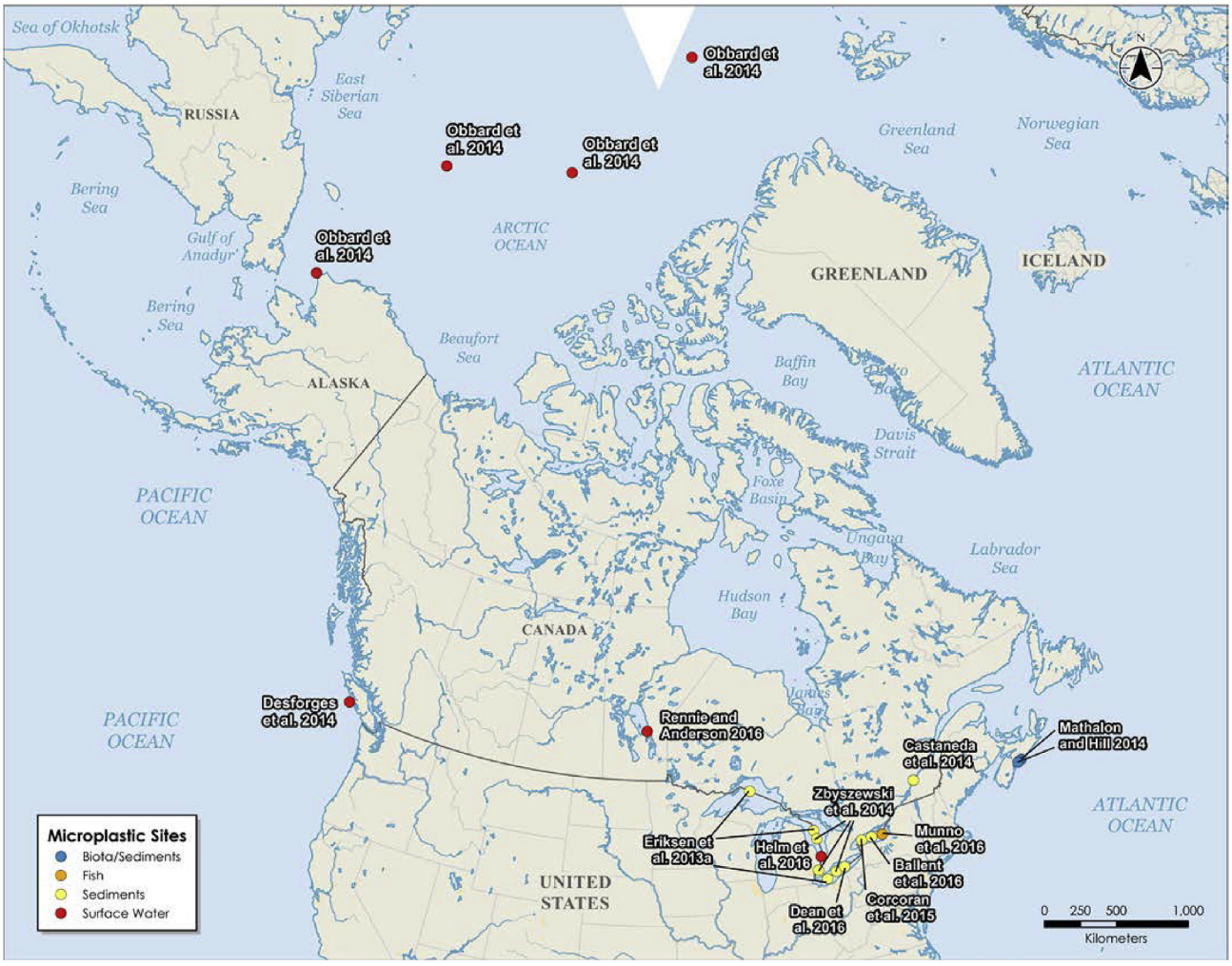

Fig. 2. Microplastic field studies conducted in Canada and adjacent water bodies.

the seals had acquired them through ingestion of *Electrona subaspera*, a common prey species that tends to feed on copepods within the size range of the observed plastic fragments (Eriksson and Burton, 2003). Though not studying microplastics specifically, Williams et al. (2011) demonstrated the presence of plastics in areas along the coast of British Columbia, which are occupied by marine mammals. Desforges et al. (2014) also documented widespread microplastic pollution in coastal BC, noting the need to further elucidate potential impacts on biota. Based on this, it would be expected that microplastics could also be present and interacting with these aquatic species.

In another Canadian study, wild mussels were collected from the same locations near Halifax Harbour that were sampled for sediments (see *Abundance in sediment*), and farmed mussels reared off the coast of Newfoundland and Labrador were purchased for comparison purposes (Mathalon and Hill, 2014). Greater amounts of microplastics were measured in the farmed mussels than in wild mussels, which may be a result of farming practices that use polypropylene lines to anchor the mussels. In the same study, microplastics in polychaete casts from the harbour sampling sites were consistent with the abundances in sediments (20–80 pieces/10 g), suggesting a relatively steady state of ingestion and egestion by these organisms (Mathalon and Hill, 2014). Preliminary results from several species of fish collected at three sites in Lake Ontario indicate that microplastic fibers were most prevalent in these fish (Munno et al., 2016). Eighty-three percent of Norway lobsters (*Nephrops norvegicus*) collected from the Clyde Sea by Murray and Cowie (2011) had plastics, mostly filaments, in their stomachs. In a follow-up laboratory feeding study, 100% of individuals had 5 mm PP filaments in their stomachs, and therefore, this species certainly has the potential to uptake microplastics (Murray and Cowie, 2011).

At this time, there is limited information about the presence of microplastics in wild marine or freshwater organisms in Canada. This represents an area for future opportunistic sampling and analysis.

4. Effects of exposure to microplastics

To date, there have been some attempts to characterize the toxicity of microplastics to aquatic organisms, but the effects and mechanistic pathways involved are not yet well understood (Bakir et al., 2014; Desforges et al., 2014). The potential toxicity of microplastics likely stems from three pathways: 1) stress of ingestion – e.g., physical blockage, energy expenditure for egestion, 2) leakage of additives from plastic – e.g., plasticizers, and 3) exposure to contaminants associated with microplastics (e.g., persistent organic pollutants or 'POPs') (Andrady, 2011; Cole et al., 2011; Ross and Morales-Caselles, 2015). The effects of exposure to microplastics would be expected to vary depending on

accumulation and translocation within tissues, ability of the organism to egest the particles, and potential for trophic transfer (Wright et al., 2013a). Given the paucity of data from Canadian studies, data from international studies were also examined.

4.1. Evidence of microplastic ingestion

Microplastics can be ingested by aquatic organisms, including coral, barnacles, sea cucumbers, polychaete worms, zooplankton, rotifers, ciliates, crustaceans, amphipods, molluscs and fish (Thompson et al., 2004; Browne et al., 2008; Cole et al., 2011; Graham and Thompson, 2009; Imhof et al., 2013; Wright et al., 2013b; Setälä et al., 2014; Hall et al., 2015) (Table S-4). Once ingested, these particles can be transferred to higher trophic levels (Murray and Cowie, 2011; Farrell and Nelson, 2013; Setälä et al., 2014; de Sá et al., 2015). Some species are capable of rapid excretion or egestion, while others retain, accumulate, and/or mobilize microplastics into their circulation. For example, *Gammarus pulex* and *Potamopyrgus antipodarum* (mudsnail) allowed to graze on fluorescent microplastics for one week deposited particles into 96% and 83%, respectively, of feces produced, demonstrating ingestion and egestion (Imhof et al., 2013). *Eurytemora affinis* copepods also ingested microplastics within a 12 h exposure period and egested the majority of the particles within an additional 12 h (Setälä et al., 2014). In contrast, Wegner et al. (2012) observed PS particles in hemolymph of blue mussels 48 d post-exposure.

Particles can be ingested by filter feeders directly from the water column or by benthic organisms after the particles have settled to the sediment (Arthur et al., 2008b). Filter feeders might have greater exposure to microplastics than organisms employing other feeding strategies (Leslie et al., 2013). For example, farmed mussels and oysters purchased in Germany were found to contain 0.36 microplastics/g wet weight (ww) and 0.47 particles/g ww, respectively. Based on estimates of the average consumption of molluscs by European consumers, the average person could ingest between 1800 and 11,000 microplastic particles per year (Van Cauwenberghe and Janssen, 2014). Another filter feeder, a baleen whale (*Balaenoptera physalus* - Mediterranean fin whale) was reported by Fossi et al. (2012) to contain microplastics in blubber samples at levels up to 9.67 pieces/m^3. Filter feeding and long life spans in baleen whales in general present opportunities for uptake of microplastics and potential accumulation of POPs (Fossi et al., 2012).

A number of planktonic and (micro-) crustacean species have been studied in an effort to understand microplastic ingestion (Table S-4). In one study, individuals from 15 different taxa were exposed to PS beads for 24 h, and 13 of the 15 taxa were observed to ingest the beads via different feeding strategies (i.e., filter-feeding, entrapping into the body cavity, or directly engulfing) (Cole et al., 2013). The two species that did not ingest the PS were raptorial predators, suggesting that they were likely not enticed by the microbeads. Size selection was employed so that not all taxa ingested all sizes of PS, and generally, the beads were expelled in feces within a few hours of ingestion. In addition to ingestion, it was also observed that the microbeads adhered to external surfaces of the test animals, including swimming legs, antennae, and carapaces (Cole et al., 2013). Desforges et al. (2015) also documented microplastics in wild −caught copepods and euphausiids sampled at multiple sites in the NE Pacific Ocean. Encounter rates were 1 particle per 34 copepods and 1 particle per 17 euphausiids, with the mean size of ingested particles reflecting food size selection by each species. Based on the results in zooplankton species, the authors estimated that juvenile salmon in coastal British Columbia could ingest 2–7 microplastic particles per day and returning adults could ingest up to 91 particles per day (Desforges et al., 2015).

Ten species of pelagic and demersal fish were collected from the English Channel by Lusher et al. (2013) to investigate rates of ingestion of microplastics. Based on gut content analysis, 36.5% of individuals had ingested between 1 and 15 pieces of microplastic, with the majority being fibers and many pieces that were black in colour. Rates of ingestion did not differ between the pelagic species, which typically feed on pelagic fish and invertebrates, and the demersal species, which feed on benthic fish, crustaceans, and molluscs (Lusher et al., 2013). Slightly lower rates of ingestion were reported in freshwater and marine fish collected from watersheds of the Gulf of Mexico (Phillips, 2014). A total of 51 fish species from 17 families were examined and microplastics were found in 8% of freshwater individuals and 10% of marine individuals. The plastics were PS, PP, acrylate, and polyamide, with greater ingestion by freshwater fish in urbanized streams than non-urbanized. Ingestion of microplastics was widespread across species (individuals from 65% of species had ingested microplastics), and occurred in both invertivore and herbivore-omnivore feeding guilds (Phillips, 2014).

Lab-cultured fish exposed via diet to different microplastic shapes for 24 h demonstrated greater rates of ingestion and retention for PS spheres and microbeads than other shapes in preliminary results (Munno et al., 2016). In another study, eleven streams in France with varying degrees of agricultural or industrial pressures were sampled for wild *Gobio gobio* to characterize the degree of microplastic ingestion in these freshwater systems (Sanchez et al., 2014). Ingestion was evident in 11–26% of individuals in the different streams, and microplastics were most abundant in fish at sites with the greatest anthropogenic pressures. There was no relationship between the percentage of individuals at a site consuming microplastic and measured fish metrics (e.g., gender, length, weight), and effects of ingestion on individual fish in these streams are unknown (Sanchez et al., 2014).

4.2. Effects of microplastic ingestion

While many species are capable of ingesting microplastics, the effects of microplastics have only been investigated to a limited extent in aquatic biota (Table S-4). Whether microplastics can have effects on smaller aquatic organisms, consistent with effects caused by macroplastic exposure in larger organisms (e.g., internal damage due to ingestion, choking hazard, entanglement), is not known (Ross and Morales-Caselles, 2015).

Given their importance as primary producers in aquatic ecosystems, preliminary studies have been conducted to evaluate the impacts of microplastics in algae. Exposure to nano-PS for 72 h caused reduced growth and cellular chlorophyll-a content of *Scenedesmus obliquus* (Besseling et al., 2014). PS adsorbed to *Chlorella* and *Scenedesmus* spp., causing reduced photosynthesis and increased production of reactive oxygen species (Bhattacharya et al., 2010), possibly due to either light shading or obstruction of CO_2 and nutrient pathways. Feeding algae (*Scenedesmus obliquus*) aged in the presence of microplastics reduced chlorophyll concentrations and growth (Besseling et al., 2014), indicating the potential for chronic effects. While these effects on algae might be positive in habitats where algae are overabundant, there is potential for food chain disruption if primary producers are negatively affected (Bhattacharya et al., 2010).

Chronic exposure to microplastics may result in reproductive and developmental effects in crustaceans. *Tigriopus japonicus* nauplii and adults were exposed to PS microbeads for 96 h (and an acute 24 h period), during which time they were observed to feed upon microplastics in the presence or absence of food (Lee et al., 2013). While there were no effects on survival in the acute tests, chronic exposure resulted in reduced survival and fecundity and developmental delays in offspring compared to control organisms.

There were also no reported effects on sex ratio or evidence of maternal transfer of PS to eggs or offspring (Lee et al., 2013). Chronic exposure to nano-PS caused reduced clutch and neonate size, as well as malformations in the offspring of *Daphnia magna* compared to controls (Besseling et al., 2014). Reproductive effects were observed in *Calanus helgolandicus* exposed to PS for 9 days (Cole et al., 2015). The presence of PS reduced ingestion of algae, and after 7 d, eggs were significantly smaller and there was impaired hatching success (Cole et al., 2015). While there were no effects of PE ingestion on survival, intermoult duration, or growth of *Idotea emarginata* over the course of a 7-week exposure period, there was a reduction in food uptake (Hämer et al., 2014).

Hyalella azteca were exposed to fluorescent PE and PP microplastics in acute and chronic tests to determine lethal exposure levels and long-term effects (Au et al., 2015). The 10-d LC50s were 4.64×10^4 pieces of PE/mL and 71.43 pieces of PP/mL. For the chronic exposure, amphipods were exposed to up to 20,000 pieces of PE/mL for 42 days, and levels of 5000 and 10,000 pieces/mL resulted in a reduced number of neonates per female. Negative effects on growth were also noted, which could be a result of slower egestion and longer residence time of microplastics in the gut than food materials (Au et al., 2015).

Intake and egestion of microplastics has been investigated in blue mussels (*Mytilus edulis* (L.)) (Table S-4). Ingestion and accumulation of PS microbeads can occur into the gut cavity, where they can remain in the digestive tract, be absorbed into the gut epithelium via phagocytosis, or be egested via feces (Browne et al., 2007, 2008). *M. edulis* displayed less filtering activity and production of pseudofeces in the presence of PS microbeads, which could indicate a purging response to low nutritional value of the filtered compounds (Wegner et al., 2012). Another study reported that PS ingestion caused no significant effects on oxidative status of hemolymph, viability or phagocytic activity of hemocytes, or filter-feeding activity after a 3- or 12-h exposure duration and 48-day depuration period (Browne et al., 2008). However, microplastics were still present in the hemolymph after 48 days (Browne et al., 2008), and due to the open circulatory system of *M. edulis*, could be transported to vital organs (Browne, 2008). When exposed to HDPE fragments, this species accumulated HDPE in the gills, stomach, lysosomal system, and digestive gland (von Moos et al., 2012). An inflammatory response occurred within 6 h of ingestion, and the lysosomal membrane was destabilized over the course of the 96-h exposure duration. While *M. edulis* appears to have the ability to egest microplastics via feces, there is potential for accumulation and trophic transfer (Farrell and Nelson, 2013) and/or effects of long-term exposure.

Other freshwater and marine species that have been fed microplastics have experienced effects related to reduced nutritional status or general stress response, but generally did not experience acute toxicity. For example, larvae of the sea urchin *Tripneustes gratilla* ingested PE microspheres over the course of a 5-day exposure, causing reduced body width at the highest exposure tested abundance (300 spheres/mL), but no effects on survival (Kaposi et al., 2014). Another sea urchin, *Paracentrotus lividus*, ingested either PS-NH_2 or PS-COOH for 48 h post-fertilization. Exposure to PS-NH_2 and PS-COOH caused cell-specific apoptosis and a general stress response, respectively, and the differences in response were likely a result of the differences in charge of the particles (Della Torre et al., 2014). Lugworms (*Arenicola marina)* fed PVC particles for four weeks fed less, had reduced lipid reserves, and increased inflammatory response. In addition, gut retention times were 1.5 times that of the controls, indicating the use of extended energy-intensive digestive processes (Wright et al., 2013b). In another study, PVC was found to increase susceptibility of *A. marina* to oxidative stress (Browne et al., 2013). In contrast, a 24-h exposure to PE followed by a 7-day depuration period did not cause significant effects (i.e., mortality) in adult sandhoppers (*Talitrus saltator*), but the authors acknowledge that there could be long-term effects in this species as a result of microplastic ingestion. The common littoral crab (*Carcinus maenas* (L.)) accumulated PS microspheres into hemolymph following feeding on blue mussels, but there were no observable changes to the physical or behavioural conditions of the crabs over a 21-day period (Farrell and Nelson, 2013).

One laboratory study examined ingestion of microplastics by fish. Common goby (*Pomatoschistus microps*) were collected in the field and then exposed to PE for 96 h (de Sá et al., 2015). Juvenile fish consumed PE both in the presence and absence of food, and tended to mistake white particles for food more commonly than black or red. Interactive effects were observed whereby individuals from habitats with higher background pollution suffered reduced predatory performance (calculated as the percentage of *Artemia franciscana* nauplii ingested of those offered) in the presence of microplastics. Therefore, habitat characteristics might influence the susceptibility of fish to mistake microplastics for prey (de Sá et al., 2015).

In addition to direct effects in individuals consuming microplastics, predators at higher trophic levels might consume prey contaminated by plastic particles. In a study examining effects of trophic transfer, algae were mixed with nanoparticles of PS, which was fed to *Daphnia magna*, and the daphnids were then fed to *Carassius carassius* (*crucian carp*) (Mattsson et al., 2015). After repeating this 3-day feeding cycle over the course of 61 days, reduced feeding activity was observed in the carp, as well as changes to behaviour, metabolites in liver and muscle, and brain histology. Specifically, ethanol was increased in the liver and inosine/adenosine and lysine were increased in muscle tissues, while nanoparticle-fed fish had heavier, swollen brains, with greater water content. Given the lipophilic nature of microplastics, PS particles may aggregate in lipid rich organs, including the brain, disrupting biological membranes and inducing the effects observed in fish (Mattsson et al., 2015).

4.3. Microplastic-associated contaminants

Plastics can contain a wide variety of additives. Phthalates are among the most common plastic additives and are often associated with PVC (Norwegian Institute for Water Research, 2014). The exchange of additives or contaminants between microplastic particles and the surrounding water depends on the concentration gradient, the matrix in which the microplastics are present, physical and chemical properties of the plastic polymer and chemical, and degradation processes acting upon the microplastic particles (Norwegian Institute for Water Research, 2014; Teuten et al., 2007, 2009). Chemicals generally adsorb to the non-crystalline regions of plastic polymers, and smaller additives tend to move out of plastics fastest (Norwegian Institute for Water Research, 2014). A study by Bakir et al. (2014) showed that phenanthrene and 4,4′-DDT reached sorption equilibrium on plastics relatively quickly (within 24 h), while Mato et al. (2001) reported that after 6 days, virgin pellets deployed in the field had not yet reached concentrations of contaminants (PCBs, DDE, and nonylphenols) measured in aged pellets collected from the same location.

In addition to the compounds used in plastic manufacturing, there is also potential for exposure of aquatic biota to other contaminants that adsorb to microplastic particles. Microplastics are typically hydrophobic and have large surface areas, allowing them to accumulate organic pollutants such as polycyclic aromatic hydrocarbons (PAHs), polybrominated diphenylethers (PBDEs), polychlorinated biphenyls (PCBs), and dichlorodiphenyltrichloroethane

(DDT) (Andrady, 2011; Cole et al., 2011; Napper et al., 2015; Norwegian Institute for Water Research, 2014). Not surprisingly, these POPs have been shown to adsorb onto microplastics at concentrations that are several orders of magnitude higher than in the surrounding water, increasing the exposure of aquatic organisms (Betts, 2008; Andrady, 2011; Engler, 2012). Organic pollutants will also preferentially bind to microplastics, as opposed to larger plastic fragments, as a result of their increased surface area to volume ratio (Teuten et al., 2007). In fact, recognition of the binding capacity of plastics has led to their increasing use as passive sampling devices to estimate POP concentrations in water, sediment and biota. Low and high density PE, PVC, and PP devices have all been employed as passive samplers because of their ability to concentrate hydrophobic contaminants from the surrounding water or sediment (USEPA, 2012).

Likewise, plastic particles can associate with metals, which can potentially accumulate at concentrations equivalent to, or greater than, those in the surrounding sediments or water (Ashton et al., 2010). Holmes et al. (2014) demonstrated sorption of Cd, Co, Cr, Cu, Ni, Pb, and Zn to microplastics under varying salinity and pH conditions. Adsorption to aged (beached) pellets was generally greater than to fresh pellets, and for all metals except Cu and Cr, adsorption was greater in freshwater than seawater (Holmes et al., 2014). In a later study, adsorption of Ag, Al, Cd, Co, Cr, Fe, Hg, Mn, Ni, Pb, and Zn to plastic pellets was greater for aged pellets than fresh pellets, and increasing pH increased adsorption of Ag, Cd, Co, Ni, Pb, and Zn (Turner and Holmes, 2015). Median concentrations of metals on the pellets ranged from <3 ng/g Ag and Hg to 34,400 ng/g Fe when exposed to solutions containing 5 μg/L of a test metal for 168 h (Turner and Holmes, 2015). These studies demonstrate the ability of microplastics to accumulate metals to greater concentrations than in the surrounding water, potentially increasing exposure concentrations for aquatic organisms ingesting these particles.

Sorption rates vary by plastic type and specific contaminant (Teuten et al., 2007). For example, in one study, DDT was found to out-compete phenanthrene for sorption onto PVC and PE (Bakir et al., 2012). Sorption capacities of PS, PE, and PP for PAHs, hexachlorohexanes, and chlorinated benzenes were experimentally determined to be within one order of magnitude for most of the compounds tested (Lee et al., 2014). For all compounds except the most hydrophobic PAHs, PS had the greatest affinity of the three plastic polymers. For PAHs and benzenes, PE had greater affinity than PP, but the opposite was true for hexachlorohexanes (Lee et al., 2014). Another study reported strong sorption of nano-PS to planar PCBs and generally greater sorption of 17 PCBs to nano-PS than micro-PE or organic matter in sediments (Velzeboer et al., 2014).

While salinity does not seem to affect sorption capacity of plastics for most POPs, it can affect the rate and amount of sorption for some of these contaminants (Bakir et al., 2014). For example, one study reported increasing sorption of PCBs to nano-PS and micro-PE as salinity increased, with a simultaneous decrease in sorption to organic matter in sediment (Velzeboer et al., 2014). In addition, it has been proposed that concentrations of contaminants might be higher in estuarine than riverine or marine waters, and estuaries represent important potential sources and sinks of contaminated microplastics (Bakir et al., 2014).

Plastics themselves are considered biologically inert since most aquatic organisms lack enzymes to break down synthetic polymers. However, plastic-associated contaminants or additives can leach, depending upon temperature, pH, particle size, and other properties of the plastic and surrounding medium (Andersson, 2014; Andrady, 2011; Baulch and Perry, 2014). The potential for these chemical contaminants to leach into aquatic organisms following ingestion of microplastics is a major concern, but is only beginning to be understood.

While not extensively characterized, microplastic-associated contaminants have been detected in a number of different global water bodies (Table S-5), indicating that adsorption does occur under environmental conditions. For most aquatic species, the predominant route of contaminant transfer from plastic is via ingestion (Teuten et al., 2009). While the contaminants themselves are generally well studied, their transfer and effects as a result of interaction with microplastics still requires additional research to be fully understood. In much the same way that partitioning coefficients define partitioning of contaminants from water to plastic, desorption of contaminants from microplastic particles and absorption by biota that have ingested those particles is also described by partition coefficients. However, these constanst have not been clearly defined and can be affected by the presence of surfactants, organic matter, and/or gastric juices (Teuten et al., 2009).

Additives associated with microplastics, such as bisphenol A (BPA) and phthalates, can potentially affect the endocrine systems of aquatic organisms, impacting mobility, reproduction, and development. Phthalates and BPA are known endocrine disruptors in fish, crustaceans, and invertebrates, and have been shown to cause whole-body and molecular effects at concentrations in the ng/L to μg/L range (Cole et al., 2011).

Several laboratory exposure studies have been conducted with lugworms (*A. marina*) and artificially contaminated microplastics (Table S-5). In one study, lugworms were exposed for 10 days to PVC sorbed with nonylphenol, triclosan, phenanthrene, and PBDE-47 (Browne et al., 2013). The contaminants desorbed from the plastic and accumulated into the guts at concentrations 326 to 3700% larger than the exposure concentrations. The uptake of nonylphenol from PVC caused impaired inflammatory responses, and triclosan from PVC caused reduced survival and sediment engineering activities by the worms (Browne et al., 2013). In another study, lugworms were exposed to PCBs associated with PS particles for 28 days (Besseling et al., 2013). There were no effects on survival, but PS caused reduced activity and weight loss, and PCBs were accumulated into tissues by a factor of 1.1–1.5 (Besseling et al., 2013).

Biological models have also been created in an attempt to model the potential toxicity of POPs associated with microplastics. One model suggested that leaching of BPA and nonylphenol from ingested microplastics into *Arenicola marina* (polychaete worm) and *Gadus morhua* (cod) contributed negligibly to total exposure to these compounds (Koelmans et al., 2014). Another model similarly predicted that dilution and clearing processes in *A. marina* would dominate and prevent accumulation of POPs from microplastics. Greater risks were associated with chemicals for which plastic is the main source (e.g., plasticizers and other additives) (Koelmans et al., 2013). However, these results were determined under modeled conditions and should be interpreted with caution (Koelmans et al., 2014).

Fish species were studied under laboratory conditions using both artificially contaminated microplastics and those that had been deployed under environmental conditions for a set period of time (Table S-5). In a study by Oliveira et al. (2013), the common goby (*P. microps*) was exposed to PE-associated pyrene for 96 h. Pyrene caused abnormal swimming and lethargy, but both activity of glutathione-S-transferase and lipid peroxidation were unaffected. The presence of PE slightly delayed mortality caused by pyrene, and also caused reduced activity of acetylcholine esterase and isocitrate dehydrogenase, and increased accumulation of pyrene metabolites (Oliveira et al., 2013). Two studies were conducted with Japanese medaka exposed to microplastics for two months, the first with PE sorbed to PAHs, PCBs, and PBDEs, and the second

with LDPE pellets that had been aged in San Diego Bay for three months. The contaminated PE pellets caused down-regulation of vitellogenin, choriogenin, and estrogen receptor gene expression in females and down-regulation of choriogenin in males, indicating potential for endocrine disruption (Rochman et al., 2014). Medaka exposed to environmentally-contaminated LDPE showed liver stress, including glycogen depletion, fatty vacuolation, and single cell necrosis. These fish also accumulated detectable amounts of PBDEs from the aged pellets (Rochman et al., 2013).

The overall results of fish and invertebrate studies indicate that microplastic-associated contaminants have the potential to accumulate in tissues and cause effects that are specific to the contaminant and plastic polymer, including endocrine disruption, behavioural modifications, and changes in metabolic processes.

4.4. Other ecological effects of microplastics

In additional to the potential for physical or toxicological effects, microplastics introduce hard substrate into aquatic ecosystems, which can subsequently alter pelagic and bacterial communities (Carpenter et al., 1972; Goldstein et al., 2012). For example, species of *Vibrio* and heterotrophic bacteria have been found to colonize plastic debris (Quilliam et al., 2014). Two other studies noted that bacterial assemblages on microplastics were diverse and differed significantly from those on organic material in the channel and in the water column (Zettler et al., 2013; McCormick et al., 2014). It was unknown whether this was a result of differences in chemical composition or the presence of a hard surface to colonize (McCormick et al., 2014), but these surfaces are longer lived than natural substrate, and the hydrophobic surface might promote biofilm formation (Zettler et al., 2013).

Likewise, in the North Pacific Subtropical Gyre, abundances of microplastics have increased by two orders of magnitude in the past 40 years, releasing *Halobates sericeus* (water strider) from substrate limitations on oviposition. There has been an overall increase in egg densities of this insect, and abundances of microplastics have been positively correlated with populations of *H. sericeus* in the region (Goldstein et al., 2012). As microplastics increase in Canadian marine and freshwater environments, similar shifts in community structures and colonization could occur. Some researchers suggest that microplastics could act as vectors for pathogens (Zettler et al., 2013; Quilliam et al., 2014) and/or exotic species (Wagner et al., 2014) as a result of these opportunistic colonizers, but these interactions are poorly understood, especially for freshwater ecosystems (Wagner et al., 2014).

5. Knowledge gaps

Current knowledge of microplastics in the environment is based predominantly on studies conducted in the past decade, with many publications appearing after 2010. Gaps in understanding of the sources, fate, behaviour, and toxicity of microplastics and their associated contaminants in the aquatic environment exist. Several recent workshops, focused meetings, and review articles have sought to identify knowledge gaps and priorities for future research (e.g., Arthur et al., 2008a; Zarfl et al., 2011; Ivar do Sul and Costa, 2014; Lowe, 2014; Norwegian Institute for Water Research, 2014; Wagner et al., 2014; Eerkes-Medrano et al., 2015; Van Cauwenberghe et al., 2015). This review contributes to the conversation around identifying knowledge gaps that may guide future work on microplastics in aquatic environments in Canada as follows:

- **Sources to freshwater and marine environments** – More information is needed regarding the relative importance of primary and secondary sources of microplastics, both globally and in a Canadian context, as well as trends in abundances within different environmental compartments and geographic regions. A better understanding of abiotic and biotic factors that can affect transfer and uptake of these compounds will help in development of proactive approaches to preventing their release into the freshwater environment.
- **Uptake and effects of microplastics** – While ingestion of microplastics has been demonstrated in a variety of aquatic organisms, there has not been a definitive answer to the question of what the effects of ingestion are, or which organisms are most vulnerable to effects. Trophic transfer of microplastics (i.e., mechanisms, rates) and long-terms effects of these compounds in marine or freshwater systems are not well understood. Nanoplastics have only recently gained attention; therefore, much more work will need to be performed to understand the fate and effects of these as well. Dose-response relationships and mode(s) of action have not been established, and subsequently, thresholds for harm and regulatory guidelines cannot be developed at this time.
- **Fate, behaviour, and effects of microplastics in freshwater environments** – Microplastics in freshwater have been severely understudied in general, compared to marine systems, and therefore, the presence and effects of microplastics in freshwater ecosystems remains largely unknown. In Canada, only a handful of studies have been conducted to characterize the presence of microplastics in water and sediment in lakes and rivers. The Great Lakes have been, and will continue to be, important waterbodies for microplastics monitoring.
- **Microplastics in the Arctic** – Microplastics have been found in sea ice, fin whales, and seal scat, suggesting that Arctic organisms will not be spared exposure to microplastics and their associated contaminants. This region is of cultural and ecological importance to Canadians, and organisms such as polar bears and Cetacean species are long-lived and feed at high trophic levels, potentially increasing their susceptibility to accumulate microplastics and contaminants such as POPs.
- **Toxicology of associated contaminants** – Limited combinations of plastic polymers, contaminants, and target organisms have been studied to date to establish the potential for toxicity due to microplastic-associated contaminants. Interactions between different contaminants require additional study to aid in development of a risk assessment framework for microplastics, which can include mixture toxicity. As with the microplastics alone, trophic transfer of associated contaminants and long-terms effects in marine or freshwater systems is not well understood.

Globally, considerable progress has been made, particularly in the past five years, in characterizing the presence and potential effects of microplastics in the aquatic environment. However, our understanding of these compounds and their associated contaminants in Canadian waterbodies, especially freshwater, are relatively incomplete and represent opportunities for further research. In addition to the gaps outlined above, it is suggested that future work could seek to improve sampling tools and techniques, and consider chronic exposures, given the recalcitrant nature of these compounds.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at http://dx.doi.org/10.1016/j.envpol.2016.06.074.

References

Andersson, E., 2014. Micro Plastics in the Oceans and Their Effect on the Marine Fauna, 19 pp. Available online: http://stud.epsilon.slu.se/6634/7/andersson_e_140904.pdf.

Andrady, A.L., 2011. Microplastics in the marine environment. Mar. Pollut. Bull. 62, 1596–1605.

Arthur, C.A., Baker, J., Bamford, H., 2008a. Proceedings of the International Research Workshop on the Occurrence, Effects and Fate of Microplastic Marine Debris. University of Washington, Tacoma, Tacoma, WA.

Arthur, C., Bamford, H., Baker, J., 2008b. The Occurrence, Effects and Fate of Small Plastic Debris in the Oceans, 16 pp. Available online: http://savetheplasticbag.com/uploadedfiles/noaa%20white%20paper.pdf.

Ashton, K., Holmes, L., Turner, A., 2010. Association of metals with plastic production pellets in the marine environment. Mar. Pollut. Bull. 60, 2050–2055.

Au, S.Y., Bruce, T.F., Bridges, W.C., Klaine, S.J., 2015. Responses of *Hyalella azteca* to acute and chronic microplastic exposures. Environ. Toxicol. Chem. 34 (11), 2564–2572.

Bakir, A., Rowland, S.J., Thompson, R.C., 2012. Competitive sorption of persistent organic pollutants onto microplastics in the marine environment. Mar. Pollut. Bull. 64, 2782–2789.

Bakir, A., Rowland, S.J., Thompson, R.C., 2014. Transport of persistent organic pollutants by microplastics in estuarine conditions. Estuar. Coast. Shelf Sci. 140, 14–21.

Ballent, A., Pando, S., Purser, A., Juliano, M.F., Thomsen, L., 2013. Modelled transport of benthic marine microplastic pollution in the Nazaré Canyon. Biogeosciences 10, 7957–7970.

Ballent, A., Corcoran, P.L., Madden, O., Helm, P.A., Longstaffe, F.J., 2016. Sources and sinks of microplastics in Canadian Lake Ontario nearshore, tributary and beach sediments. Mar. Pollut. Bull. 110, 383–395. MPB-D-16–0037R1.

Barnes, D.K., Galgani, F., Thompson, R.C., Barlaz, M., 2009. Accumulation and fragmentation of plastic debris in global environments. Philos. Trans. R. Soc. Lond. B Biol. Sci. 364, 1985–1998.

Baulch, S., Perry, C., 2014. Evaluating the impacts of marine debris on cetaceans. Mar. Pollut. Bull. 80, 210–221.

Besseling, E., Wang, B., Lurling, M., Koelmans, A.A., 2014. Nanoplastic affects growth of *S. obliquus* and reproduction of D. magna. Environ. Sci. Technol. 48, 12336–12343.

Besseling, E., Wegner, A., Foekema, E.M., van den Heuvel-Greve, M.J., Koelmans, A.A., 2013. Effects of microplastic on fitness and PCB bioaccumulation by the lugworm *Arenicola marina* (L.). Environ. Sci. Technol. 47, 593–600.

Betts, K., 2008. Why small plastic particles may pose a big problem in the oceans. Environ. Sci. Technol. 42, 8995.

Bhattacharya, P., Lin, S., Turner, J.P., Ke, P.C., 2010. Physical adsorption of charged plastic nanoparticles affects algal photosynthesis. J. Phys. Chem. 114, 16556–16561.

Browne, M.A., Crump, P., Niven, S.J., Teuten, E., Tonkin, A., Galloway, T., Thompson, R., 2011. Accumulation of microplastic on shorelines worldwide: sources and sinks. Environ. Sci. Technol. 45, 9175–9179.

Browne, M.A., Dissanayake, A., Galloway, T.S., Lowe, D.M., Thompson, R.C., 2008. Ingested microscopic plastic translocates to the circulatory system of the mussel, *Mytlius edulis* (L. Environ. Sci. Technol. 42, 5026–5031.

Browne, M.A., Galloway, T., Thompson, R., 2007. Microplastic - an emerging contaminant of potential concern? Integr. Environ. Assess. Manage. 3, 559–566.

Browne, M.A., Galloway, T.S., Thompson, R.C., 2010. Spatial patterns of plastic debris along estuarine shorelines. Environ. Sci. Technol. 44, 3404–3409.

Browne, M.A., Niven, S.J., Galloway, T.S., Rowland, S.J., Thompson, R.C., 2013. Microplastic moves pollutants and additives to worms, reducing functions linked to health and biodiversity. Curr. Biol. 23, 2388–2392.

Browne, M.A., 2008. Ingested microscopic plastic translocates to the circulatory system of the mussel *M edulis*. Environ. Sci. Technol. 42 (13), 5026–5031.

Carpenter, E.J., Anderson, S.J., Harvey, G.R., Miklas, H.P., Peck, B.B., 1972. Polystyrene spherules in coastal waters. Science 178, 749–750.

Castañeda, R.A., Avlijas, S., Simard, M.A., Ricciardi, A., 2014. Microplastic pollution in St. Lawrence river sediments. Can. J. Fish. Aquat. Sci. 71, 1767–1771.

Canadian Water and Wastewater Association (CWWA), 2001. National Survey of Wastewater Treatment Plants, Final Report, 17 pp. Available online: http://cwwa.ca/pdf_files/freepub_wastewaterreport.PDF.

Claessens, M., De Meester, S., Van Landuyt, L., De Clerck, K., Janssen, C.R., 2011. Occurrence and distribution of microplastics in marine sediments along the Belgian coast. Mar. Pollut. Bull. 62, 2199–2204.

Cole, M., Lindeque, P., Fileman, E., Halsband, C., Galloway, T.S., 2015. The impact of polystyrene microplastics on feeding, function and fecundity in the marine copepod *Calanus helgolandicus*. Environ. Sci. Technol. 49 (2), 1130–1137.

Cole, M., Lindeque, P., Fileman, E., Halsband, C., Goodhead, R., Moger, J., Galloway, T.S., 2013. Microplastic ingestion by zooplankton. Environ. Sci. Technol. 47, 6646–6655.

Cole, M., Lindeque, P., Halsband, C., Galloway, T.S., 2011. Microplastics as contaminants in the marine environment: a review. Mar. Pollut. Bull. 62, 2588–2597.

Corcoran, P.L., 2015. Benthic plastic debris in marine and fresh water environments. Environ. Sci. Process. Impacts 8, 1363–1369.

Corcoran, P.L., Biesinger, M.C., Grifi, M., 2009. Plastics and beaches: a degrading relationship. Mar. Pollut. Bull. 58, 80–84.

Corcoran, P.L., Norris, T., Ceccanese, T., Walzak, M.J., Helm, P.A., Marvin, C.H., 2015. Hidden plastics of Lake Ontario, Canada and their potential preservation in the sediment record. Environ. Pollut. 204, 17–25.

Dean, B.Y., Corcoran, P.L., Helm, P., Ballent, A.M., 2016. Microplastics in beach and bottom sediments of Lake Erie and its tributaries. In: Abstract Submitted for the 59th Annual Conference on Great Lakes Research, International Association for Great Lakes Research (IAGLR), Guelph, Ontario, June 6-10, 2016.

de Sá, L.C., Luis, L.G., Guilhermino, L., 2015. Effects of microplastics on juveniles of the common goby *Pomatoschistus microps*: confusion with prey, reduction of the predatory performance and efficiency, and possible influence of developmental conditions. Environ. Pollut. 196C, 359–362.

Dekiff, J.H., Remy, D., Klasmeier, J., Fries, E., 2014. Occurrence and spatial distribution of microplastics in sediments from Norderney. Environ. Pollut. 186, 248–256.

Della Torre, C., Bergami, E., Salvati, A., Faleri, C., Cirino, P., Dawson, K.A., Corsi, I., 2014. Accumulation and embryotoxicity of polystyrene nanoparticles at early stage of development of sea urchin embryos *Paracentrotus lividus*. Environ. Sci. Technol. 48, 12302–12311.

Depledge, M.H., Galgani, F., Panti, C., Caliani, I., Casini, S., Fossi, M.C., 2013. Plastic litter in the sea. Mar. Environ. Res. 92, 279–281.

Derraik, J.G.B., 2002. The pollution of the marine environment by plastic debris: a review. Mar. Pollut. Bull. 44, 842–852.

Desforges, J.P., Galbraith, M., Dangerfield, N., Ross, P.S., 2014. Widespread distribution of microplastics in subsurface seawater in the NE Pacific Ocean. Mar. Pollut. Bull. 79, 94–99.

Desforges, J.P., Galbraith, M., Ross, P.S., 2015. Ingestion of microplastics by zooplankton in the Northeast Pacific Ocean. Arch. Environ. Contam. Toxicol. 69 (3), 320–330.

Doughty, R., Eriksen, M., 2014. The case for a ban on microplastics in personal care products. Tulane Environ. Law J. 27, 277–298.

Driedger, A.G.J., Dürr, H.H., Mitchell, K., Van Cappellen, P., 2015. Plastic debris in the Laurentian Great Lakes: a review. J. Gt. Lakes Res. 41, 9–19.

Dris, R., Gasperi, J., Rocher, V., Saad, M., Renault, N., Tassin, B., 2015a. Microplastic contamination in an urban area: a case study in Greater Paris. Environ. Chem. 12, 592–599.

Dris, R., Imhof, H., Sanchez, W., Gasperi, J., Galgani, F., Tassin, B., Larforsch, C., 2015b. Beyond the ocean: contamination of freshwater ecosystems with (micro-) plastic particles. Environ. Chem. 12 (5), 32.

Eerkes-Medrano, D., Thompson, R.C., Aldridge, D.C., 2015. Microplastics in freshwater systems: a review of the emerging threats, identification of knowledge gaps and prioritization of research needs. Water Res. 75, 63–82.

Engler, R.E., 2012. The complex interaction between marine debris and toxic chemicals in the ocean. Environ. Sci. Technol. 46, 12302–12315.

Environment Canada, 2012. Frequently Asked Questions. Available online: http://www.ec.gc.ca/eau-water/default.asp?lang=En&n=1C100657-1.

Environment Canada, 2015. Microbeads – a Science Summary, 34 pp. Available online: http://www.ec.gc.ca/ese-ees/ADDA4C5F-F397-48D5-AD17-63F989EBD0E5/Microbeads_Science%20Summary_EN.pdf.

Eriksen, M., Mason, S., Wilson, S., Box, C., Zellers, A., Edwards, W., Farley, H., Amato, S., 2013a. Microplastic pollution in the surface waters of the Laurentian Great Lakes. Mar. Pollut. Bull. 77, 177–182.

Eriksson, C., Burton, H., 2003. Origins and biological accumulation of small plastic particles in fur seals from Macquarie Island. AMBIO A J. Hum. Environ. 32, 380–384.

Eriksen, M., Lebreton, L.C.M., Carson, H.S., Thiel, M., Moore, C.J., Borerro, J.C., Galgani, F., Ryan, P.G., Reisser, J., 2014. Plastic pollution in the world's oceans: more than 5 trillion plastic pieces weighing over 250,000 tons afloat at sea. PLoS One 9. http://dx.doi.org/10.1371/journal.pone.0111913.

Farrell, P., Nelson, K., 2013. Trophic level transfer of microplastic: *Mytilus edulis* (L.) to *Carcinus maenas* (L.). Environ. Pollut. 177, 1–3.

Fendall, L.S., Sewell, M.A., 2009. Contributing to marine pollution by washing your face: microplastics in facial cleansers. Mar. Pollut. Bull. 58, 1225–1228.

Fossi, M.C., Panti, C., Guerranti, C., Coppola, D., Giannetti, M., Marsili, L., Minutoli, R., 2012. Are baleen whales exposed to the threat of microplastics? a case study of the Mediterranean fin whale *Balaenoptera physalus*. Mar. Pollut. Bull. 64, 2374–2379.

Free, C.M., Jensen, O.P., Mason, S.A., Eriksen, M., Williamson, N.J., Boldgiv, B., 2014. High-levels of microplastic pollution in a large, remote, mountain lake. Mar. Pollut. Bull. 85, 156–163.

Frias, J.P., Otero, V., Sobral, P., 2014. Evidence of microplastics in samples of zooplankton from Portuguese coastal waters. Mar. Environ. Res. 95, 89–95.

Galgani, F., Fleet, D., Van Franeker, J., Katsanevakis, S., Maes, T., Mouat, J., Oosterbaan, L., Pitou, I., Hanke, G., Thompson, R., Amata, E., Birkun, A., Janssen, C., 2010. Marine Strategy Framework Directive: Task Group 10 Report, Marine Litter. European Commission Joint Research Centre Scientific and Technical Report. 57 pp..

Gilman, N.E., 2013. Examining Spatial Concentrations of Marine Micro-plastics on Shorelines in South Puget Sound. M.ES thesis. Evergreen State College, Washington, 101 pp.

Goldstein, M.C., Rosenberg, M., Cheng, L., 2012. Increased oceanic microplastic debris enhances oviposition in an endemic pelagic insect. Biol. Lett. 8, 817–820.
Graham, E.R., Thompson, J.T., 2009. Deposit- and suspension-feeding sea cucumbers (Echinodermata) ingest plastic fragments. J. Exp. Mar. Biol. Ecol. 368, 22–29.
Hall, N.M., Berry, K.L.E., Rintoul, L., Hoogenboom, M.O., 2015. Microplastic ingestion by scleractinian corals. Mar. Biol. 162, 725–732. http://dx.doi.org/10.1007/s00227-015-2619-7.
Hämer, J., Gutow, L., Kohler, A., Saborowski, R., 2014. Fate of microplastics in the marine isopod *Idotea emarginata*. Environ. Sci. Technol. 48, 13451–13458.
Helm, P., Zimmer, G., Stones, M., Thibeau, J., Sims, A., Thorburn, B., Page, W., 2016. Microplastics in and entering nearshore surface waters of the lower Great Lakes. In: Abstract Submitted for the 59th Annual Conference on Great Lakes Research, International Association for Great Lakes Research (IAGLR), Guelph, Ontario, June 6-10, 2016.
Hidalgo-Ruz, V., Gutow, L., Thompson, R.C., Thiel, M., 2012. Microplastics in the marine environment: a review of the methods used for identification and quantification. Environ. Sci. Technol. 46, 3060–3075.
Hitchings, L., 2014. Why Illinois Has Banned Exfoliating Face Washes. New Scientist. http://www.newscientist.com/article/dn25773-why-illinois-has-banned-exfoliating-face-washes.html#.VOtEoBY6l97.
Holmes, L.A., Turner, A., Thompson, R.C., 2014. Interactions between trace metals and plastic production pellets under estuarine conditions. Mar. Chem. 167, 25–32.
Imhof, H.K., Ivleva, N.P., Schmid, J., Niessner, R., Laforsch, C., 2013. Contamination of beach sediments of a subalpine lake with microplastic particles. Curr. Biol. 23, R867–R868.
Ivar do Sul, J.A., Costa, M.F., 2014. The present and future of microplastic pollution in the marine environment. Environ. Pollut. 185, 352–364.
Kaposi, K.L., Mos, B., Kelaher, B.P., Dworjanyn, S.A., 2014. Ingestion of microplastic has limited impact on a marine larva. Environ. Sci. Technol. 48, 1638–1645.
Koelmans, A.A., Besseling, E., Foekema, E.M., 2014. Leaching of plastic additives to marine organisms. Environ. Pollut. 187, 49–54.
Koelmans, A.A., Besseling, E., Wegner, A., Foekema, E.M., 2013. Plastic as a carrier of POPs to aquatic organisms: a model analysis. Environ. Sci. Technol. 47, 7812–7820.
Kusui, T., Noda, M., 2003. International survey on the distribution of stranded and buried litter on beaches along the Sea of Japan. Mar. Pollut. Bull. 47, 175–179.
Laglbauer, B.J., Franco-Santos, R.M., Andreu-Cazenave, M., Brunelli, L., Papadatou, M., Palatinus, A., Grego, M., Deprez, T., 2014. Macrodebris and microplastics from beaches in Slovenia. Mar. Pollut. Bull. 89, 356–366.
Lattin, G.L., Moore, C.J., Zellers, A.F., Moore, S.L., Weisberg, S.B., 2004. A comparison of neustonic plastic and zooplankton at different depths near the southern California shore. Mar. Pollut. Bull. 49, 291–294.
Lee, H., Shim, W.J., Kwon, J.H., 2014. Sorption capacity of plastic debris for hydrophobic organic chemicals. Sci. Total Environ. 470–471, 1545–1552.
Lee, K.W., Shim, W.J., Kwon, O.Y., Kang, J.H., 2013. Size-dependent effects of micro polystyrene particles in the marine copepod *Tigriopus japonicus*. Environ. Sci. Technol. 47, 11278–11283.
Leslie, H.A., 2014. Review of Microplastics in Cosmetics: Scientific Background on a Potential Source of Plastic Particulate Marine Litter to Support Decision-making. IVM Institute for Environmental Studies. Available online: http://www.ivm.vu.nl/en/Images/Plastic%20ingredients%20in%20Cosmetics%2007-2014%20FINAL_tcm53-409859.pdf.
Leslie, H.A., van Velzen, M.J.M., Vethaak, A.D., 2013. Microplastic Survey of the Dutch Environment: Novel Data Set of Microplastics in North Sea Sediments, Treated Wastewater Effluents and Marine Biota. IVM Institute for Environmental Studies, 30 pp. Available online. http://www.ivm.vu.nl/en/Images/IVM%20report%20Microplastic%20in%20sediment%20STP%20Biota%202013_tcm53-409860.pdf.
Lowe, S.E., 2014. The Great Lakes Land-based Marine Debris Action Plan. NOAA Technical Memorandum NOS-OR&R-49. 29 pp.
Lozano, R.L., Mouat, J., 2009. Marine Litter in the North-East Atlantic Region: Assessment and Priorities for Response, 136 pp. Available online: http://qsr2010.ospar.org/media/assessments/p00386_Marine_Litter_in_the_North-East_Atlantic_with_addendum.pdf.
Lusher, A.L., Burke, A., O'Connor, I., Officer, R., 2014. Microplastic pollution in the Northeast Atlantic Ocean: validated and opportunistic sampling. Mar. Pollut. Bull. 88, 325–333.
Lusher, A.L., McHugh, M., Thompson, R.C., 2013. Occurrence of microplastics in the gastrointestinal tract of pelagic and demersal fish from the English Channel. Mar. Pollut. Bull. 67, 94–99.
Mathalon, A., Hill, P., 2014. Microplastic fibers in the intertidal ecosystem surrounding Halifax Harbor, Nova Scotia. Mar. Pollut. Bull. 81, 69–79.
Mato, Y., Isobe, T., Takada, H., Kanehiro, H., Ohtake, C., Kaminuma, T., 2001. Plastic resin pellets as a transport medium for toxic chemicals in the marine environment. Environ. Sci. Technol. 35, 318–324.
Mattsson, K., Ekvall, M.T., Hansson, L.A., Linse, S., Malmendal, A., Cedervall, T., 2015. Altered behavior, physiology, and metabolism in fish exposed to polystyrene nanoparticles. Environ. Sci. Technol. 49, 553–561.
McCormick, A., Hoellein, T.J., Mason, S.A., Schluep, J., Kelly, J.J., 2014. Microplastic is an abundant and distinct microbial habitat in an urban river. Environ. Sci. Technol. 48, 11863–11871.
Moore, C.J., Moore, S.L., Weisberg, S.B., Lattin, G.L., Zellers, A.F., 2002. A comparison of neustonic plastic and zooplankton abundance in southern California's coastal waters. Mar. Pollut. Bull. 44, 1035–1038.
Munno, K.E., Helm, P., Jackson, D., Pourier, D., Bhavsar, S.P., Chong-Kit, R., Petro, S., 2016. Microplastic ingestion by shape in several species of fish from Lake Ontario. In: Abstract Submitted for the 59th Annual Conference on Great Lakes Research, International Association for Great Lakes Research (IAGLR), Guelph, Ontario, June 6-10, 2016.
Murray, F., Cowie, P.R., 2011. Plastic contamination in the decapod crustacean *Nephrops norvegicus* Linnaeus, 1758. Mar. Pollut. Bull. 62, 1207–1217.
Napper, I.E., Bakir, A., Rowland, S.J., Thompson, R.C., 2015. Characterisation, quantity and sorptive properties of microplastics extracted from cosmetics. Mar. Pollut. Bull. 99 (1), 178–185.
Norwegian Institute for Water Research, 2014. Microplastics in Marine Environments - Occurrence, Distribution, and Effects, 73 pp.
Obbard, R.W., Sadri, S., Wong, Y.Q., Khitun, A.A., Baker, I., Thompson, R.C., 2014. Global warming releases microplastic legacy frozen in Arctic Sea ice. Earth's Future 2, 315–320.
Office of the New York State Attorney General, 2015. Unseen Threat: How Microbeads Harm New York Waters, Wildlife, Health and Environment. Environmental Protection Bureau prepared by Jennifer Nalbone. 17 pp. Available online. https://ag.ny.gov/pdfs/Microbeads_Report_5_14_14.pdf.
Oliveira, M., Ribeiro, A., Hylland, K., Guilhermino, L., 2013. Single and combined effects of microplastics and pyrene on juveniles (0+ group) of the common goby *Pomatoschistus microps* (Teleostei, Gobiidae). Ecol. Indic. 34, 641–647.
Phillips, M.B., 2014. The Occurrence and Amount of Microplastics Ingested by Fishes in the Watersheds of the Gulf of Mexico. M.Sc. thesis. Texas State University, 34 pp.
Plastics Europe, 2015. Plastics – the Facts 2015: an Analysis of European Plastics Production, Demand and Waste Data, 30 pp. Available online: http://www.plasticseurope.org/Document/plastics—-the-facts-2015.aspx.
Quilliam, R.S., Jamieson, J., Oliver, D.M., 2014. Seaweeds and plastic debris can influence the survival of faecal indicator organisms in beach environments. Mar. Pollut. Bull. 84, 201–207.
Rennie, M.D., Anderson, P.J., 2016. First estimate of microplastic pollution in Lake Winnipeg. In: Abstract Submitted for the 59th Annual Conference on Great Lakes Research, International Association for Great Lakes Research (IAGLR), Guelph, Ontario, June 6-10, 2016.
Rochman, C.M., Hoh, E., Kurobe, T., Teh, S.J., 2013. Ingested plastic transfers hazardous chemicals to fish and induces hepatic stress. Sci. Rep. 3, 3263. http://dx.doi.org/10.1038/srep03263.
Rochman, C.M., Kurobe, T., Flores, I., Teh, S.J., 2014. Early warning signs of endocrine disruption in adult fish from the ingestion of polyethylene with and without sorbed chemical pollutants from the marine environment. Sci. Total Environ. 493, 656–661.
Ross, P.S., Morales-Caselles, C., 2015. Out of sight, but no longer out of mind: microplastics as a global pollutant. Integr. Environ. Assess. Manag. 11 (4), 719–728.
Sanchez, W., Bender, C., Porcher, J.M., 2014. Wild gudgeons *Gobio gobio* from French rivers are contaminated by microplastics: preliminary study and first evidence. Environ. Res. 128, 98–100.
Setälä, O., Fleming-Lehtinen, V., Lehtiniemi, M., 2014. Ingestion and transfer of microplastics in the planktonic food web. Environ. Pollut. 185, 77–83.
Talvitie, J., Heinonen, M., Pääkkönen, J.-P., Vahtera, E., Mikola, A., Setälä, O., Vahala, R., 2015. Do wastewater treatment plants act as a potential point source of microplastics? Preliminary study in the coastal Gulf of Finland, Baltic Sea. Wat. Sci. Tech. 72 (9), 1495–1504.
Teuten, E.L., Rowland, S.J., Galloway, T.S., Thompson, R.C., 2007. Potential for plastics to transport hydrophobic contaminants. Environ. Sci. Technol. 41, 7759–7764.
Teuten, E.L., Saquing, J.M., Knappe, D.R., Barlaz, M.A., Jonsson, S., Bjorn, A., Rowland, S.J., Thompson, R.C., Galloway, T.S., Yamashita, R., Ochi, D., Watanuki, Y., Moore, C., Viet, P.H., Tana, T.S., Prudente, M., Boonyatumanond, R., Zakaria, M.P., Akkhavong, K., Ogata, Y., Hirai, H., Iwasa, S., Mizukawa, K., Hagino, Y., Imamura, A., Saha, M., Takada, H., 2009. Transport and release of chemicals from plastics to the environment and to wildlife. Philos. Trans. R. Soc. Lond. B Biol. Sci. 364, 2027–2045.
Thompson, R.C., Olsen, Y., Mitchell, R.P., Davis, A., Rowland, S.J., John, A.W.G., McGonigle, D., Russell, A.E., 2004. Lost at sea - where is all the plastic? Science 304, 838.
Turner, A., Holmes, L., 2015. Adsorption of trace metals by microplastic pellets in fresh water. Environ. Chem. 12, 600–610.
United Nations Environment Programme (UNEP), 2015. Dutch Rally Support for a Europe Wide Microplastic Ban. Available online: http://www.unep.org/NewsCentre/default.aspx?DocumentID=2817&ArticleID=11120.
United States Environmental Protection Agency (USEPA), 2012. Guidelines for Using Passive Samplers to Monitor Organic Contaminants at Superfund Sediment Sites. OSWER Directive 9200.1–110 FS. 32 pp.
Van Cauwenberghe, L., Vanreusel, A., Mees, J., Janssen, C.R., 2013. Microplastic pollution in deep-sea sediments. Environ. Pollut. 182, 495–499.
Van Cauwenberghe, L., Janssen, C.R., 2014. Microplastics in bivalves cultured for human consumption. Environ. Pollut. 193, 65–70.
Van Cauwenberghe, L., Devriese, L., Galgani, F., Robbens, J., Janssen, C.R., 2015. Microplastics in sediments: a review of techniques, occurrence and effects. Mar. Environ. Res. 111, 5–17.
Velzeboer, I., Kwadijk, C.J., Koelmans, A.A., 2014. Strong sorption of PCBs to nanoplastics, microplastics, carbon nanotubes, and fullerenes. Environ. Sci. Technol. 48, 4869–4876.
Vert, M., Doi, Y., Hellwich, K.-H., Hess, M., Hodge, P., Kubisa, P., Rinaudo, M.,

Schue, F., 2012. Terminology for biorelated polymers and applications (IUPAC Recommendations 2012). Pure Appl. Chem. 84, 377–410.
von Moos, N., Burkhardt-Holm, P., Kohler, A., 2012. Uptake and effects of microplastics on cells and tissue of the blue mussel *Mytilus edulis* L. after an experimental exposure. Environ. Sci. Technol. 46, 11327–11335.
Wagner, M., Scherer, C., Alvarez-Muñoz, D., Brennholt, N., Bourrain, X., Buchinger, S., Fries, E., Grosbois, C., Klasmeier, J., Marti, T., Rodriguez-Mozaz, S., Urtbatzka, R., Vethaak, A.D., Winther-Nielsen, M., Reifferscheid, G., 2014. Microplastics in freshwater ecosystems: what we know and what we need to know. Environ. Sci. Eur. 26 (12). http://www.enveurope.com/content/26/1/12.
Wegner, A., Besseling, E., Foekema, E.M., Kamermans, P., Koelmans, A.A., 2012. Effects of nanopolystyrene on the feeding behavior of the blue mussel (*Mytilus edulis* L). Environ. Toxicol. Chem. 31, 2490–2497.
Williams, R., Ashe, E., O'Hara, P.D., 2011. Marine mammals and debris in coastal waters of British Columbia, Canada. Mar. Pollut. Bull. 62, 1303–1316.
Woodall, L.C., Sanchez-Vidal, A., Canals, M., Paterson, G.L.J., Coppock, R., Sleight, V., Calafat, A., Rogers, A.D., Narayanaswamy, B.E., Thompson, R.C., 2014. The deep sea is a major sink for microplastic debris. R. Soc. Open Sci. 1, 140317–140317.
Wright, S.L., Thompson, R.C., Galloway, T.S., 2013a. The physical impacts of microplastics on marine organisms: a review. Environ. Pollut. 178, 483–492.
Wright, S.L., Rowe, D., Thompson, R.C., Galloway, T.S., 2013b. Microplastic ingestion decreases energy reserves in marine worms. Curr. Biol. 23, R1031–R1033.
Yonkos, L.T., Friedel, E.A., Perez-Reyes, A.C., Ghosal, S., Arthur, C.D., 2014. Microplastics in four estuarine rivers in the Chesapeake Bay, U.S.A. Environ. Sci. Technol. 48, 14195–14202.
Zarfl, C., Fleet, D., Fries, E., Galgani, F., Gerdts, G., Hanke, G., Matthies, M., 2011. Microplastics in oceans. Mar. Pollut. Bull. 62, 1589–1591.
Zbyszewski, M., Corcoran, P.L., 2011. Distribution and degradation of fresh water plastic particles along the beaches of Lake Huron, Canada. Water, Air, & Soil Pollut. 220, 365–372.
Zettler, E.R., Mincer, T.J., Amaral-Zettler, L.A., 2013. Life in the "plastisphere": microbial communities on plastic marine debris. Environ. Sci. Technol. 47, 7137–7146.
Zubris, K.A.V., Richards, B.K., 2005. Synthetic fibers as an indicator of land application of sludge. Environ. Pollut. 138, 201–211.