White Paper –
Microplastics in Aquatic Systems: An Assessment of Risk

SUMMARY OF CRITICAL ISSUES AND RECOMMENDED PATH FORWARD

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2017
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WE&RF Project Number: CEC7R17

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## Contents

Executive Summary........................................................................................................................................5

1. **Background Rationale and Approach** .................................................................................................7
   1.1 Rationale........................................................................................................................................7
   1.2 Critical Questions ..........................................................................................................................7
   1.3 Literature Review Process..............................................................................................................8

2. **Risk = Exposure x Effects** ................................................................................................................9

3. **Microplastic Exposures: Laboratory vs. Nature and Effluents** .........................................................10

4. **Can Microplastics Introduce Toxic Chemicals to Aquatic Organisms?** ..........................................12

5. **Which Microplastics Dominate in the Aquatic Systems and How Are They Quantified?** .............13

6. **What About Microplastics in Treated Municipal Wastewater?** ......................................................15

7. **What About Microplastics in Treated Drinking Water?** ................................................................17

8. **Conclusions** ....................................................................................................................................17

9. **Critical Knowledge Needs and a Strategic Path Forward** ...............................................................18

10. **Literature Cited** .............................................................................................................................19

Appendix A: Useful Microplastics References .......................................................................................23
Executive Summary

Microplastics (MPs) in the aquatic environment have become a focus of concern in recent years for government, business, academia, non-profit organizations, and the general public. Reports of millions to trillions of these small particles (less than 500 microns) occurring in freshwaters and marine systems have alarmed many and led to widespread banning and phasing out of microbeads (one component of MPs).

The publication of scientific papers in the peer-reviewed literature is increasing at an incredible rate, now ~150 per year, addressing the occurrence, fate, effects, sources, and wastewater treatment of MPs. As in any new science, many early publications are overly simplistic, qualitative in nature, and suffer from methodological limitations and over interpretation. The over interpretation of the significance of MPs created public alarm with resulting regulatory and industry actions. The science is rapidly improving in terms of analytical methods to identify MP types and numbers and the realization that real-world exposures are not being tested in biological effect studies. The author states that this means the majority of research identifying adverse effects of MPs has used excessively high exposures, elevated by up to several orders of magnitude.

Despite these shortcomings, a number of conclusions can be drawn concerning MPs in the environment:

1) Macro-plastics, not MPs, damage fish-eating birds, aquatic mammals and reptiles, and fish due to physical harm.
2) MPs measured in recent field studies have not yet been shown to cause adverse effects to aquatic wildlife (fish, shellfish, and macroinvertebrates) at *environmentally relevant* concentrations (in the laboratory or field) but estimation techniques remain inadequate.
3) MPs adsorb some toxic chemicals, but are not an exposure route of significance in aquatic birds or aquatic organisms, as compared to prey consumption.
4) MPs are more likely to be elevated near urban centers and in depositional sediments near municipal wastewater treatment plant (WWTP) outfalls.
5) Benthic macroinvertebrates in sediments near WWTP outfalls are the most likely receptors to be exposed to potentially adverse levels of MPs.
6) MP concentrations in waters containing the highest number of reported particles are below 10 particles per 1,000 liters, resulting in very low potential for exposure and uptake by biota.
7) MPs in aquatic systems are typically dominated by fibers and/or fragments while microbeads are a minor component.
8) While microbeads are being phased out of consumer products in the United States, MPs will not likely decrease due to fibers and breakdown fragments from macro-plastics.
9) The predominant source of fibers is likely from washing clothes and antifouling boat paints.
10) WWTPs remove the majority of MPs, with most being captured in sludge.
11) Filtration is an optimal treatment for removing MPs from wastewater effluents and intake waters.
12) Several governmental agencies have identified similar knowledge gaps and research needs to better manage and understand the environmental implications of MPs.
13) It is questionable whether currently existing aquatic toxicity tests assess the potential physical impacts of MPs.
14) Improved MP exposure models for effluent discharges into receiving waters are needed to predict whether they may be a stressor of concern.
15) No standard methods exist for sampling and quantifying MPs, therefore making it impossible to compare studies or reliably predict exposure, effects, hazard, or risk.
A strategic research plan is needed to address a few critical knowledge gaps within the next five years. This plan should be conducted in concert with interested federal/national agencies, e.g., United States Environmental Protection Agency (U.S. EPA), Environment and Climate Change Canada (ECCC), National Oceanic and Atmospheric Administration (NOAA), United States Geological Survey (USGS), European Chemicals Agency (ECHA), Commonwealth Scientific and Industrial Research Organization (CSIRO) and with standards-setting organizations such as U.S. EPA, American Society for Testing and Materials (ASTM), International Organization for Standardization (ISO), and/or Organization for Economic Co-operation and Development (OECD). Some of the knowledge gaps are currently being addressed by these agencies and individual researchers, so the strategic plan should describe an organizational process for engaging key parties and stakeholders to optimize research expenditures, topics, and leads.

The priority research needs are:

1) Determine if and where adverse effects to freshwater and marine populations and communities may occur under reasonable worst-case scenarios.
2) Determine the most cost-effective treatment option for removing MPs from wastewaters and which WWTPs are most likely to require MP treatment.
3) Standardize MP sampling methods for 300 to 500 micron sizes and for MPs smaller than 300 microns;
4) Standardize analytical methods for determining the type, shape and size of MPs with associated QA/QC guidelines.
5) Determine if standardized aquatic toxicity tests provide relevant information on the adverse effects of MPs (physical and chemical) at environmentally relevant concentrations.
6) Ascertain the threshold concentration for key receptors, spatial extent, MP type, source and ecological risk of small MPs below 300 microns to the nanoparticle size (e.g., anti-fouling paint chips).
7) Develop exposure models for WWTP effluents and receiving waters.
8) Develop a translational research program to inform WE&RF subscribers, stakeholders, regulators, and the public.

**Keywords:** Microplastics and environment, microbeads and environment, microfibers and environment, plastic debris and environment, wastewater treatment and microplastics, microplastics and personal care products, microplastics and cosmetic products, aquatic systems, risk.
1. Background Rationale and Approach

1.1 Rationale

A number of challenging issues exist regarding the understanding the significance of the presence of microplastics (MPs) in aquatic ecosystems. The scientific community, public, government, and environmental advocacy sectors largely believe MPs are causing significant water quality problems and injuring aquatic and terrestrial wildlife. The science is young and there are many knowledge gaps that should be filled regarding MPs in the environment and an assessment of the true exposures to the aquatic biota.

Macro-plastics are indeed a threat to the environment. Pictures showing fish, sea turtles, fish-eating birds, surface-feeding birds, and sea mammals with their gastrointestinal systems clogged and their bodies entangled and strangled by plastics are common. This results from a wide range of macro-plastic debris, their fragments, fishing nets and fishing line. This threat is primarily associated with marine ecosystems as they are the final dumping ground for flowing waters and most of humanity lives near coastal areas. Currently, the macro-plastic threat seems very difficult to control given our dependence on plastic items for everyday life and industrial applications.

The concern over plastics and distressing pictures of suffering wildlife has likely contributed to the negative focus on MPs. Certainly, as long as there are macro-plastics, MPs cannot be regulated as they are largely comprised of breakdown fragments of macro-plastics and fibers from clothing, anti-fouling marine paints, and other products. However, microbeads which are but one component of MPs, are a different matter as they are added to consumer products and, therefore, can be regulated. And so, there has been widespread legislative action both in the U.S. and abroad, phasing out and banning the use of microbeads from use in consumer products (e.g., facial cleansing scrubs) (Burton, 2016). The plethora of publications in the recent years documenting the hazards of MPs, including their occurrence in marine and freshwaters, their ability to adsorb pollutants, and their adverse effects on aquatic biota, has hastened the legislative action against microbeads and the alarm over MPs being discharged in municipal wastewater.

This White Paper consists of a critical literature review with an identification of key knowledge gaps based. The recommendations of recent U.S. EPA, NOAA, and critical reviews on research needs were considered. The review focused on addressing the following risk-associated issues to identify the extent of knowledge gaps.

1.2 Critical Questions

1. What is the true exposure of aquatic organisms to MPs?

2. What are the levels of MPs in WWTP effluents (i.e., secondary and tertiary) (range and percentile distributions)? What is the distribution of MP types (e.g., microbeads, fragments, fibers) and sizes associated with urban WWTP effluents? Fibers appear to be more common in freshwater fish than other MPs – why? What are their sources? Are they an ecological concern?

3. How does the removal efficiency vary between wastewater treatment processes?

4. What are the concentrations likely found in Combined Sewer Overflows?

5. What are the worst-case scenarios for MP concentrations in drinking water intakes?
6. Are MPs likely removed in the process of drinking water treatment?

7. What are the reported levels of MPs (as MPs/L – surface volume units) in human-dominated waterways (reasonable worst case) including freshwater streams and coastal area mixing zones?

8. Are the size fractions of MPs usually being sampled (300-500 micron) appropriate from an ecological exposure and effects view? What is the occurrence and potential effects of MPs smaller than 300 microns?

9. Are adverse effects on aquatic biota possible at concentrations found in worst-case scenarios?

10. Can metals and trace organic compounds adsorbed to MPs be a risk concern, given their concentrations in nature and chemical uptake rates?

11. Could MPs concentrations in sediments near WWTP outfalls be a concern to benthic biota?

12. Are MPs a carrier for pathogens, due to their large surface area? If so, this this increased load of health significance?

13. What is the state-of-the-science for quantifying MPs and identifying their type in surface waters, sediments, and WWTP effluents?

14. Are standard aquatic and sediment toxicity tests adequate for assessing MPs, particularly given that effects may be due to physical stress and may be chronic?

15. Are new standardized methods needed with determinations of intra- and inter-laboratory variance (method associated QA/QC guidelines with possible round-robin testing)? Which standardization process should be used (state, province, national, international, non-profit) given this may be a policy discussion?

Following the identification of knowledge gaps, a strategic plan is proposed for prioritizing research needs. These needs are summarized into tentative research project designs that combine various knowledge gaps to increase funding effectiveness and efficiency.

1.3 Literature Review Process

An extensive (but not comprehensive) review of the peer-reviewed literature was conducted on MPs in the environment, including wastewater effluents, to address the above research questions. The recent publication of peer-reviewed papers in international scientific journals has sky-rocketed in recent years, increasing more than an order of magnitude per year in the last decade (Connors et al., 2017; Lusher et al., 2017). This year (2017) there will be likely be in excess of 150 papers published. There have been several useful government reports, critical reviews and perspectives (e.g., Connors et al., 2017; Duis and Coors 2016; Hidalgo-Ruz et al., 2012; Lenz et al., 2016; Lusher et al., 2017; NOAA 2013, 2015; Rochman et al., 2016; Shim et al., 2017; U.S. EPA, 2015, 2016; Wagner et al., 2014).

As in many new and emerging issues in the environmental sciences, the initial research identified the issue while using traditional methods without fully understanding some critical methodological and environmental exposure issues. Unfortunately, given the high visibility of initial studies and focus by environmental advocates, state, and federal regulations have been put into place for microbeads before fully understanding their ecological risk. However, the science is expanding quickly and improving, as evidenced by the findings below. Recent studies are increasingly providing an improved characterization of ecological risk and, thereby, allow for more effective decision-making.

The peer-reviewed literature concerned with MPs in the environment can be grouped into three categories: 1) surveys on concentrations in surface waters (and a few on wastewater effluents, sediments, and beach sands); 2) uptake and effects of MPs in aquatic organisms, and 3) analytical methods to quantify the type, number, and shape of MPs.
Papers selected for critical review are listed in the Literature Cited section and Appendix A. This critical review allows for some initial generalities in regards to these three MP subject categories:

1. **Surveys** largely conducted with surface net trawls by ships, with MP numbers converted to numbers per unit surface area; The MP size fraction sampled as largely been between 300 and 500 microns;

2. Uptake and effect studies largely conducted on single species dosed in the laboratory at relatively high MP densities; and

3. Analytical methods proposed as optimal for identifying plastic types, while others best for identifying numbers and shapes. Each method varies in its respective strengths and limitations; and, no methodological approach has been standardized.

### 2. Risk = Exposure x Effects

To determine the risk of MPs, it is first important to understand the basic principles of risk determination and what comprises “hazard” and “risk”. These terms are often used interchangeably and each have unique regulatory consequences, particularly in the European Union (EU). A “hazard” is any source of potential damage, harm or adverse health effects on something or someone (https://www.ccohs.ca/oshanswers/hsprograms/hazard_risk.html). It also can be defined as a situation posing a level of threat to life, health, property, or the environment. Hazards can be dormant or have a potential to occur with only a theoretical risk of harm (https://en.wikipedia.org/wiki/hazard). In the EU, varying degrees of “hazard” are used to classify chemicals through the REACH process. Environmental risk is primarily considered in the Water Framework Directive (Burton et al., 2012).

Sometimes risk is used in the “hazard” context. The U.S. EPA ecological risk assessment process is often conducted as a retrospective hazard assessment, not defining future probability. The U.S. EPA defines risk as the chance of harmful effects to ecological systems resulting from exposure to environmental stressors, such as chemicals, land change, disease, invasive species, and climate change (https://www.epa.gov/risk). There is a continuum between hazard and risk which varies depending on the user and regulatory body.

It is important to recognize risk cannot have adverse consequences (i.e., effects) without exposure to a hazard. Exposure assessments examine what is known about the frequency, timing, and levels of contact of an organism with a stressor. Frequency, duration, and magnitude of stressor exposures vary dramatically across ecosystems and depending on the organism, their life stage, and time of year. These three critical components of exposure have been identified by the U.S. EPA Science Advisory Board as a common weakness in that they are rarely defined adequately in ecological risk characterizations (Daley et al., 2005). This is not surprising, as it is challenging to quantify all three and then relate them to adverse effects.

Not only do stressors (e.g., MPs) each vary in magnitude through space and time; but, organisms rarely are sedentary and are moving across space and through time – moving in and out of stressor exposures while they primarily focus on feeding, finding desirable habitat and reproduction (Burton, 2017).

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1 A reviewer of the MP White paper notes that many studies have relied on visual observation leading to false positive and false negative MP characterization. Effects of chemical extraction on particples is unknown potentially destroying particles, interfering with positive ID, and even fragmenting particles – leading to over counting.
Assessing exposure is half of the determination of risk, while the other half is the linkage of exposure to effects. These two components are combined in the risk characterization to examine the degree of relationship between an exposure and adverse effects (http://www.epa.gov/risk/exposure.htm; http://www.epa.gov/risk/risk-characterization.htm). In summary, this means there can be no risk without a realistic exposure to a concentration of the stressor that will illicit an adverse effect.

Another important consideration is exposure (and sublethal biological responses) does not always equate to an adverse effect. For example, liver enzymes are induced after consuming a glass of wine, but the effect is not “adverse” until there are excessive exposures. Also, all of us likely have trace levels of some “toxic” chemicals (e.g., PCBs, PBDEs, PFASs) in our tissues, yet most of us have no apparent adverse effects. Unfortunately, there are only a small number of chemical-specific tissue residue thresholds developed that can be related to adverse responses in wildlife (including fish), such as selenium, PCBs, and dioxin.

These exposure issues must be addressed to determine the ecological risk of MPs. While the knowledge gaps and research needs will be addressed below, it is obvious to many that the science of MPs in the environment should quickly address the following broad questions to allow for their effective and efficient management:

1. What is a “reasonable worst-case” (RWC) and “typical” exposure of aquatic biota in nature to MPs?
2. Where does this RWC exposure occur?
3. Are RWC exposures causing adverse biological effects?
4. Do MPs pose a hazard or risk to aquatic organisms, and if so, which ones?
5. How is the risk quantified and what is its associated uncertainty?


The laboratory studies of MP effects published in the peer-reviewed literature, including the world’s top journals, have used MP concentrations that were two to seven orders of magnitude higher than those observed in the most polluted MP waters in studies to date (Burton, 2015; Lusher et al., 2017). The perhaps unrealistic exposures used were likely a result of one or more of the following reasons:

1) Higher MP concentrations were required to elicit effects.

2) Higher MP concentrations are easier to measure.

3) Literature values reported MP numbers per square kilometer of surface area which are not easily translated to a volume measure (number or density per liter, i.e., mg/L) as used in the lab to expose organisms.

4) There is a large amount of error in counting and identifying MP particles, with high levels of false positives and negatives.

5) MP concentrations below 300 μm and outside of surface trawls capture zones have rarely been counted.

MP particles vary in density, size, and shape whether produced as microbeads, fibers, or as the fragments of larger, disintegrating plastics. Many natural particles, such as silica and organic material particles are incorrectly identified as MPs, as discussed below. Conversion of surface trawl area numbers to mg/L is tenuous and uncertain, and that uncertainty increases as the size of the MP decreases below 300 microns to the size of nanoparticles. It is well known nanoparticle toxicity depends on size and shape, in addition to numbers (Klaine et al., 2008). It appears this is also the case with MPs (Austin and Weinstein, 2017). Water samples are a mix of particle types (plastic and natural), sizes, shapes and
densities, with varying degrees of biofouling and the MPs varying spatially and temporally – so, it is difficult to imagine how laboratory exposures can effectively translate to in situ conditions.

As Paracelcus said almost 500 years ago "All things are poison and nothing is without poison; only the dose makes a thing not a poison." Virtually any substance consumed can be toxic. If organisms are exposed to a concentration that exceeds their ability to compensate, then there will be adverse effects. While extrapolating from high doses to low doses for prediction threshold effects is an accepted approach in human toxicology when testing carcinogens, it is not acceptable in environmental toxicology if the true exposures are below acute or chronic toxicity (i.e., adverse) effect levels. In aquatic toxicology, acute to chronic ratios rarely exceed 100; therefore, acute toxicity that results from exposures several orders of magnitude higher than occur in nature, suggests that Predicted No Effect Concentrations (PNECs) will never be exceeded.

Lower MP concentrations perhaps nearer to true environmental exposures have elicited sublethal biomarker responses and have suggested alterations of endocrine system function may occur (Rochman et al., 2013, 2014). Unfortunately, biomarker responses, such as using enzymatic, ecogenicomic, and other protein or metabolic “-omics” makers, have not been reliably linked to adverse population or community effects. A sublethal response such as these in an organism is regarded as an indication of exposure, but not adverse effects.

Recently, the failure to use environmentally relevant MP concentrations in laboratory testing has been realized by many authors (Burton, 2015; Connor et al., 2017; Lenz et al., 2016; Phuong et al., 2015; Sussarellu et al., 2016; Ziccardi et al., 2016). In the oceans, reported values are approximately 0.1 MP/100 liters (maximum of 9,200/1000 liters). The highest concentrations in freshwaters have been reported in Lake Erie at approximately 32 particles per 1000 liters, with a median of 1.9. These are impressive MP surface area numbers. However, when the worst areas of MP contamination are converted to volume measurements (e.g., MP particle numbers per liter) the exposure numbers are unimpressive and perhaps insignificant. These values were obtained from 29 Great Lake tributaries feeding into Lakes Michigan, Huron, and Erie. Lake Erie is the most MP contaminated water ever recorded with numbers exceeding 446,000 particles/km² with beads comprising only 16% of MPs (and fibers/lines only 2%). This converts to 1 to 3 particles per 300 to 700 liters (~80-185 gallons). However, in that same lake water there are ~10,000 to 10 million algae per liter. This means there are a billion more algae than MPs. The primary organism at risk from eating these small MPs are zooplankton and their primary food source is algae. So, the zooplankton finding that a MP particle in a liter of water will be next to impossible.

The University of Michigan recently studied the presence of MPs in Lake Erie, which has been identified as one of the most contaminated (freshwater or marine) ever recorded (over one million MPs per square kilometer). They surveyed 153 individual fish, comprised of six species at risk from MPs due to the feeding method. Only 11% to 36% (depending on species) had MPs that were only fibers (1 to 5 in number) with no MP fragments or microbeads (Duhaime et al., 2016).

Survey in the northeast Pacific and most of coastal China found up to 75% of the MPs were fibers – not microbeads (Deforges et al., 2014; Li et al., 2016). A recent study found paint particles from ship hulls much more prevalent than MPs (95% of total MPs) and most particles were less than 200 um in size (Song et al., 2013, 2015b). These high numbers may be explained by the fact that many toxic antifouling paints are designed to slough off boat hulls and contain ~200 fibers/mm² (equating to over a billion on a large boat). There has been little attention by the scientific community or advocates concerning fibers, which appear to be much more prevalent than microbead exposures.

Rochman et al. (2015) cited three reports (two Swedish reports and one student report from Niagara College) showing 0-7 microbeads per liter in final effluents of WWTPs. She converted this to a U.S. discharge of 80 trillion L of water/day containing 0.1 microbeads/L, resulting in eight trillion
microbeads/d being discharged. *She later retracted this and stated the number was billions rather than trillions.* Again, these are impressive numbers, but are highly uncertain due to the questionable citations (non-peer reviewed) and the apparent lack of high numbers of microbeads even in the most contaminated sites.

Lenz et al. (2016) reviewed several papers and converted MP numbers to field numbers using size and mass assumptions and only found one exposure close to being realistic which used very small (2 to 6 micron) particles in hot-spot sediments (Sussarellu et al., 2016). Since many MPs have a density greater than 1, or become so due to biofouling, they sink and end up in depositional sediments. Besseling et al. (2017) found the fate of nano- and microplastics in freshwater systems is largely governed by both particle size and the hydrodynamics of the system. This appears to be the only study investigating this important fate pathway for MPs.

A recent study by Baldwin et al. (2016) of 29 Great Lakes tributaries receiving municipal wastewater effluent (the dominant source of U.S. MPs) found a maximum concentration of 32 particles/m³ (or 1,000 liters) and a median of 1.9/m³. These concentrations found in RWC conditions are similar to those calculated by Burton (2015) and Lenz et al. (2016) and further confirms concentrations used in laboratory testing have been orders of magnitude too high.

A further complicating factor in determining true exposure concentrations is that MPs not only differ widely in size, composition, and also in density, all of which have a profound impact on exposure. Laboratory experiments that weigh MPs varying in size by one order of magnitude will have up to three orders of magnitude difference in the number of MP particles (Connor et al., 2017). MPs used in laboratory testing tend to be smaller than the size ranging collected in nature, and tend to be uniform spherical shapes comprised of polystyrene or polyethylene. The density, chemical adsorption capacity, and fate of MPs varies dramatically depending on their composition. In aquatic ecosystems, plastic fragments and fibers of many plastic types dominate. Trying to replicate real-world exposures for laboratory testing is, therefore, extremely difficult. Recently, Lambert et al., (2017) discuss the importance of considering particle properties when conducting ecotoxicity tests.

A recent study by Gray and Weinstein (2017) showed toxicity of MPs depended on their shape and size, with no acute effects from spheres and fragments less than 50 microns, but were acutely toxic at larger sizes, as were fibers at 93 microns. The MPs are quickly covered by biofilms which alter exposures to any adsorbed contaminants while potentially providing additional food to particle grazing zooplankton and fish larvae. Aquatic organisms feeding on small particles, the size of MPs, are simultaneously exposed to natural food particles in concentrations several orders of magnitude higher than MPs under RWC conditions (Burton, 2015). So, as mentioned above, their chances of encountering even a single MP are unlikely.

4. **Can Microplastics Introduce Toxic Chemicals and Pathogens to Aquatic Organisms?**

Many plastics are known to adsorb both inorganic and organic chemicals, and indeed are comprised of some compounds linked to endocrine disruption. Many field and laboratory studies have documented this phenomenon with MPs adsorbing PCBs, pesticides, metals, and other compounds (e.g., Teuten et al., 2007; Velzeboer et al., 2014). Some of these compounds and plastic additives have been found to leach off of the MPs when ingested by aquatic biota. However, some chemicals do not leach off as they are tightly bound to the plastic (Gouin et al., 2011; Koelmans et al., 2016; Zarfl and Matthies, 2010).
Koelmans et al. (2013, 2016) did a critical review of the literature dealing with MPs as a vector for chemicals and reevaluated data from multiple studies based on a biodynamic model for PCB accumulation. They found the major source of organic contamination of wildlife was from prey they consumed. A recent analysis of seabirds that feed along the surface of water, where MPs are more prevalent, provides further perspective. The study of northern fulmar seabirds off the coast of Norway found levels of hazardous organic pollutants in the birds had no correlation to the amount of plastic in their stomachs. Ingestion of hazardous chemicals is 21,000 times more likely from consumption of natural prey than from MPs.

Desorption of chemicals from MPs was a minor and negligible exposure pathway, even without discounting the low numbers of particles likely consumed (Koelmans et al., 2013, 2016; Velzeboer et al., 2014). Again, the issue of what is the true exposure arises for bioaccumulation. Laboratory studies have used elevated concentrations of MPs which could not be avoided by organism in test beakers, in a form of force feeding exposure. The RWC conditions in nature that result in less than one to a few particles per 1000 liters (264 gallons) does not allow for a measureable uptake by an organism which ingests it.

There have been no studies to investigate the possible role of MPs on increasing exposures to pathogens. Given that biofilms will form on most surfaces in shallow waters, it is likely that pathogens are a component of the biofilms in human-dominated waterways. The increased availability of nutrients on the particles would increase survival of pathogens, just as in sediments (Burton et al., 1987). Nevertheless, this should not be an ecological or human health issue given their low concentrations in comparisons to natural sediment particles.

5. **Which Microplastics Dominate in Aquatic Systems and How Are They Quantified?**

A few studies have attempted to quantify the various types of MPs occurring in marine and freshwaters; however, these surveys have been limited and do not allow for site-specific generalizations. As mentioned above, there appears to be a high degree of false positives and false negatives reported, depending on the methods used to sample, extract and analyze the MPs, and the experience of the laboratory, with no standard methods or QA/QC guidelines existing. The type of plastic materials and their shape can, in some cases, potentially identify sources of the plastics. The characterization of different types of plastic materials requires advanced instrumentation that is not readily available, such as Raman micro-spectroscopy and Fourier transform infrared spectroscopy (FTIR). However, the author notes that this approach can incorrectly identify particles as being plastic, resulting in false positives.

There are versions of this such as ATR-FTIR, Focal plan array-based reflection FT-IR (FPA-FT-IR). The lab of Dr. Mark Banaszak-Holl at the University of Michigan is identifying nanoplastics with a novel, but excellent approach combining atomic force microscopy-infrared spectroscopy (AFM-IR) instrument to characterize nanoplastic size, morphology, and chemical content. Other methods used to identify MPs include: field flow fractionation, Raman, TGA-GC, and optical microscopy (Table 1). Many of the best methods to identify MPs require advance instrumentation and each method has its own unique strengths and limitations.

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[2] A reviewer notes that Koelmans et al (2013, 2016) has several drawbacks as field studies evaluated relied on (unconfirmed) plastics of limited size ranges, and unknown contaminant loading.
Anothe present at concentration suggests urban waters ship hull paints contain tiny fibers and are designed to slough off as part of their biocide activity. This suggests urban waters where ships are painted with biocide-paints, they may be the dominant MP and present at concentrations higher than the larger MPs reported.

Another area of MP unknown is their presence in combined sewer overflows, which are a significant source of waste effluent loadings in some older cities and other parts of the world. There are no studies

### Table 1. Analytical Methods to Determine Microplastics Shape, Composition, and Number
Modified from Duis and Coors, 2016; Rocha-Santos 2015; Shim et al., 2017; Rocha-Santosand Durate, 2015.

<table>
<thead>
<tr>
<th>Methods *</th>
<th>Advantages and Limitations</th>
</tr>
</thead>
<tbody>
<tr>
<td>• Microscopy</td>
<td>Simple, fast, inexpensive, okay to 1 mm. High False+/− no composition.</td>
</tr>
<tr>
<td>• SEM-EDS</td>
<td>No coating or prep, elemental composition and morphology. Expensive</td>
</tr>
<tr>
<td>• ESEM-EDS</td>
<td>Same as SEM-EDS.</td>
</tr>
<tr>
<td>• Microsc. + FTIR/Raman</td>
<td>Confirm subset MP &amp; type, no prep. Some False +/−, only subset so may miss some types and sizes, color interferes</td>
</tr>
<tr>
<td>• FTIR** spectroscopy</td>
<td>No false+, few false −, size to 10 μ, visualize, no prep. Expensive, slow</td>
</tr>
<tr>
<td>o u-ATR-FTIR</td>
<td>Best method for routine analyses (Shim et al., 2017)</td>
</tr>
<tr>
<td>• Raman spectroscopy</td>
<td>As FTIR + size to 1 μ. As FTIR + pigment interference</td>
</tr>
<tr>
<td>• AFM-IR</td>
<td>No false+, few false −, size to nano-levels, identifies number, size, shape and chemical content. Expensive and slow.</td>
</tr>
<tr>
<td>• Thermal (Pyro-GC/MS)</td>
<td>Analyze polymer + additives. Few polymers, complex, expensive</td>
</tr>
<tr>
<td>o + FPA-FTIR</td>
<td>This and other thermal may miss small particles and weathered polymers problems.</td>
</tr>
<tr>
<td>o TGA-GC</td>
<td>Alternative to FTIR and good for polyethylene and polypropylene</td>
</tr>
<tr>
<td>o TDS-GC-MS</td>
<td>Quantifies polymer particles in various matrices.</td>
</tr>
</tbody>
</table>


To complicate matters, most studies have used net trawls capturing only a part of the MP size range of 300 to 500 microns. What about the known plastics that are smaller, even in the nanoparticle range (≤ 100 nm)? MPs smaller than 300 microns appear to occur in higher concentrations, but are difficult to sample. It is reasonable to suspect, however, that this would be an important and dominant size range with potentially greater potential for adverse effects if the exposures are adequate. Song et al. (2014, 2015) studied the near coastal area of South Korea for micro-sized plastics with dramatic findings. Most particles were much smaller than 50 microns and the fewest numbers were recovered in the traditionally used Manta trawl net (330 micron mesh). Alkyds and poly (acrylate/styrene) accounted for 81% and 11%, respectively, of the total polymer content of the surface microlayer. The paint particles dominated over plastic polymers such as polypropylene, polyethylene, and polyester. A second study found fragmented MPs, including paint resin particles, accounted for 75% of the total MPs, with highest concentrations near urban areas (Song et al., 2015b). The paint particles in both studies originated from ship hull biocide-containing paints with a fiber-reinforced plastic matrix. The newer biocide-containing ship hull paints contain tiny fibers and are designed to slough off as part of their biocide activity. This suggests urban waters where ships are painted with biocide-paints, they may be the dominant MP and present at concentrations higher than the larger MPs reported.
that have looked at CSO loadings of MPs. Given the lack of any treatment removal, their loadings of MPs are potentially quite high which could result in detrimental exposures close to the outfall.

As discussed above, there are several MP characteristics that likely affect toxicity, including size, shape, surface charge and functionality, and density. However, these characteristics are rarely provided by suppliers and may vary lot to lot of MP product (Connor et al., 2017). This makes it difficult, if not impossible, to reproduce results and raises important quality assurance and quality control concerns for the previously published literature on MP effects.

Since many different methods have been used to quantify and identify MPs, it is difficult to compare published studies as each method has its own unique strengths and limitations (see Table 1) (Phuong et al., 2016; Hidalgo-Ruz et al., 2012). Even within one method of choice (FTIR) and stereomicroscopes, the same individual can misidentify MPs 18% to 22% (Song et al., 2015a). This intra-laboratory error is reduced by each study laboratory developing sample-based criteria for identification of major and typical MPs in sample groups are first determined. Microscopy has been the primary method used for published papers due to its simplicity and low cost (Hidalgo-Ruz et al., 2012). However, there is a high potential for false negatives (counts lower than actual). Its reliability decreases as the size of the particle decreases, as biofouling increases on particles, and as the sample matrix becomes more complex such as with beach sands and sediments. A few studies verified counts of MPs using FTIR or Raman spectroscopy and found alarming rates of false positives of 53% to 70% Dekiff et al. (27), Hidalgo-Ruiz et al. (29) Rocha-Santos paper, Eriksen et al. (41 and 11-Duis and Coors). All of these findings suggest that the wide differences found in MP densities (whether in surface waters, sediments, or organisms) may be largely due to methodological differences and inaccurate counts (Li et al., 2016; Song et al., 2015a).

Another complicating factor in the determination of what is the true occurrence of MPs in aquatic systems is sample collection and MP extraction or isolation from the sample matrix. As with the analytical identification methods, the sample methods have varied widely with differing net sizes for trawling, size separation, and extraction methods to concentrate the MPs from surface waters, surface films, sands, sediments, and organisms (Hidalgo-Ruz et al., 2012). While NOAA (2015) has recommended quantification methods, no standard methods exist. This makes it virtually impossible to compare data between studies, reproduce, or develop general conclusions about the state-of-the-science and the scope of the MP issue.

6. What About Microplastics in Treated Municipal Wastewater?

Municipal WWTPs are the largest source of MPs into aquatic systems in the United States, and likely all developed countries (McCormick et al., 2015). Recently, Rochman et al. (2015) reported eight trillion microbeads are being discharged into U.S. waterways everyday by municipal WWTPs. This number came from the following questionable assumptions: all plants operating at half capacity; 0.1 microbeads per liter of discharged effluent (from two reports in the gray literature); and 99% of the microbeads settle into sludge (therefore, there are 800 trillion in the WWTP sludge). She closes by saying “the probability of risk from microbead pollution is high...”. Following publication, a retraction was made when it was found the numbers were in the billions instead of trillions. Mason et al. (2016) also reported widespread MP pollution from WWTP effluents, sampling 17 facilities in the U.S. The average discharge was 0.05 ± 0.024 MPs per liter effluent with a daily discharge of over four million per facility per day. Fibers (some non-plastic) and fragments dominated the MPs, yet the authors, surprisingly, attributed the fragments to microbeads from cosmetics and personal care products. They estimated three to 23 billion MPs are released each day by municipal WWTPs into U.S. waters. This estimate is less than cited in Rochman et al. (2015).

Other recent studies have measured MPs in municipal WWTP effluents and the numbers vary widely, even taking into account whether there is secondary or tertiary treatment (Table 2).
Table 2. Reported Levels of MPs in Municipal Wastewater Treatment Plant Effluents

<table>
<thead>
<tr>
<th>Reference</th>
<th>Level Reported</th>
<th>Modeled</th>
<th>Measured</th>
<th>Plant retention</th>
</tr>
</thead>
<tbody>
<tr>
<td>Wezel et al. 2016</td>
<td>0.2 – 66 ug/L 20-150 particles/L</td>
<td>X</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Talvitie et al. 2016</td>
<td>4.9 fibers and 8.6 MPs/L</td>
<td>X</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Ziajahromi et al. 2017</td>
<td>0.28 MP/L tertiary</td>
<td>X</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>0.48 MP/L secondary</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>1.54 MP/L primary</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Mintenig et al. 2017</td>
<td>.09 – 1 fibers/L (most polyester)</td>
<td>X</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>1 – 9 MPs/L</td>
<td>X</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>0.03 – 0.27 MPs/L (tertiary + filtration)</td>
<td>X</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Carr et al. 2016</td>
<td>8.8 x 10^4 MP/L secondary</td>
<td>X</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Leslie et al. 2017</td>
<td>51-81/L (10 to 5000 microns)</td>
<td>X</td>
<td>72%</td>
<td></td>
</tr>
<tr>
<td>Talvitie et al. 2017</td>
<td>0.7 – 3.5/L (secondary)</td>
<td>X</td>
<td>99%</td>
<td></td>
</tr>
<tr>
<td>Rochman et al. 2015</td>
<td>0.1/L</td>
<td>Cited</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Mason et al. 2016</td>
<td>0.05/L</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

The range of MP particles in treated municipal wastewater effluents ranges from 0.03 to 150/L. There is no apparent reason for the four order of magnitude differences based on levels of treatment, rather the wide range is likely due to a number of factors, such as treatment process and retention differences, influent loadings; but, also numerous limitations associated with the sampling and quantification of MPs as discussed above. The variations associated with methodologies for sampling and analyzing MPs do not allow for accurate conclusions regarding actual MP concentrations in environmental samples (Table 2). The implications of this limitation are profound, as they are impeding the development of the science and improved treatment processes, our ability to manage MP releases into the environment, and understand their ecological significance. When viewed in the context of total MP loadings from WWTP effluents, the numbers (like surface water contamination) are extremely high and alarming to the public. Nevertheless, as discussed above in the conversions of MP values to surface water volumes, the numbers (if they are close to being correct) being discharged by WWTPs appear inconsequential from an ecological perspective.

Total suspended solids (TSS) may be a useful surrogate for MPs in terms of physical damage that can result in aquatic biota. There is much more literature on their occurrence and biological effects in aquatic systems. If we compare MP particle numbers to the adverse effect of TSS it becomes apparent that MP particles are insignificant, meaning microbeads are even less so. Fish are adversely affected by TSS at levels ranging from 30-160 mg/liter. At the worst sites for MP pollution, such as Lake Erie, their concentrations are from 10 to 1 million-fold lower than adverse TSS levels.

7. What About Microplastics in Treated Municipal Drinking Water?

Virtually no research has been conducted on this topic. The reason for this is likely that no MPs in the traditional size class of 300 to 500 microns would be expected to make it through a modern day drinking water treatment plant that has filtration. A plant that can remove cryptosporidium oocytes which are 4 to 6 microns in size should remove most MPs. However, the potential that many MPs may be closer to the nano-size is a potential concern that has not been addressed.
8. Conclusions

Despite the paucity of high-quality data on MPs in the environment, there has been significant advances made through the hundreds of studies, allowing for a number of conclusions:

1) Macro-plastics, perhaps not MPs, damage fish-eating birds, aquatic mammals and reptiles, and fish due to physical harm.

2) MPs measured in recent field studies have not yet been shown to cause adverse effects to aquatic wildlife (fish, shellfish, and macroinvertebrates) at environmentally relevant concentrations (in the laboratory or field) but estimation techniques remain inadequate.

3) MPs adsorb some toxic chemicals, but are not an exposure route of significance in aquatic birds or aquatic organisms, as compared to prey consumption.

4) MPs may be allow for increased pathogen numbers and survival just as depositional sediments.

5) MPs are more likely to be elevated near urban centers and in depositional sediments near municipal WWTP outfalls.

6) MP loadings from CSOs are unknown, though likely higher than from treated effluents.

7) Benthic macroinvertebrates in sediments near WWTP outfalls are the most likely receptors to be exposed to potentially adverse levels of MPs.

8) MP concentrations in waters containing the highest number of particles are below 10 particles per 1,000 liters, resulting in very low potential for exposure and uptake by biota.

9) MPs in aquatic systems are typically dominated by fibers and/or fragments while microbeads are a minor component.

10) While microbeads are being phased out of consumer products in the United States, MPs will not likely decrease due to fibers and breakdown fragments from macro-plastics.

11) The predominant source of fibers is likely from washing clothes (>1,900 fibers/garment/wash – Browne et al., 2011) and antifouling boat paints.

12) WWTPs remove the majority of MPs, with most being captured in sludge.

13) Filtration is an optimal treatment for removing MPs from wastewater effluents and intake waters.

14) Several governmental agencies and scientists have identified similar knowledge gaps and research needs to better manage and understand the environmental implications of MPs (Connors et al., 2017; Duis and Connor, 2016; NOAA, 2013, 2015; U.S. EPA, 2015, 2016; Wagner et al., 2016).

15) No standard methods exist for sampling, quantifying or testing the ecotoxicity, and associated physical stress of MPs (e.g., van Cauwnbeerghe et al., 2013; Ryan et al., 2009).
9. **Critical Knowledge Needs and a Strategic Path Forward**

A strategic research plan is needed to address a few critical knowledge gaps within the next five years. This plan should be conducted in concert with interested federal/national agencies (e.g., U.S. EPA, ECCC, NOAA, USGS, ECHA, CSIRO) and with standards-setting organizations such as U.S. EPA, ASTM, ISO and/or OECD.

Some of the knowledge gaps are currently being addressed by these agencies and individual researchers, so the strategic plan should describe an organizational process for engaging key parties and stakeholders to optimize research expenditures, topics, and leads. The priority research needs are:

1) Determine if and where adverse effects to freshwater and marine populations and communities may occur under reasonable worst-case scenarios. If studies find a potentially problematic exposure that is near a PNEC threshold, then additional species testing is recommended, and a determination of the likely receptors of concern.

2) Determine the most cost-effective treatment option for removing MPs from wastewaters and which WWTPs are most likely to require MP treatment.

3) Standardize MP sampling methods for all sizes. The author notes especially for the 300 to 500 micron sizes and for MPs smaller than 300 microns.

4) Standardize analytical methods for determining the type, shape and size of MPs with associated QA/QC guidelines that include verification of false positive and false negative rates for each method and laboratory.

5) Conduct laboratory chronic toxicity testing using standardized methods (e.g., U.S. EPA, ASTM, OECD) for sensitive species likely to be exposed to MPs in surface freshwaters, near coastal marine waters, estuarine waters, and sediments. These tests should be conducted with a range of MP types that correspond to common size/shape distributions at realistic environmental concentrations (RWC initially). Tests should evaluate toxicity due to physical stress and adsorbed chemicals and pathogens.

6) Conduct a survey of depositional sediments near municipal WWTP outfalls to determine the range of MP concentrations likely in surficial sediments and porewaters.

7) Develop exposure and fate models for MPs from wastewater effluents and CSOs.

8) Ascertained the concentration, spatial extent, type, source, and ecological risk of very small MPs below 300 microns to a few microns in size (e.g., anti-fouling paint chips) in surface waters near WWTP outfalls, drinking water treatment plant influents where the waters are effluent dominated, and depositional sediments near outfalls.

9) Develop a translational research program to educate WE&RF subscribers, regulators, and the public.

These knowledge gaps are many and potentially large in scope. They will require strategic collaborations, sharing of resources and findings, and effective research designs. The following study topics and elements are recommended for consideration:

- **Standardize MP sampling, quantification and characterization methods.** This may entail using the ASTM process and a group of recognized MP experts in these methods.

- **Conduct a survey of a range of predicted RWC ecosystems, encompassing freshwater (cold and warm water), marine and estuarine waters (east and west coast).** These should be areas with known, or likely high concentrations of MPs, such as Lake Erie, tributaries to the Great Lakes (sampled by Baldwin et al., 2016), Chicago River, Puget Sound, and Chesapeake Bay. The survey should use MP standardized methods if they are developed, if not, the methods should be approved by a MP Expert Panel. The survey should survey for 300-500 micron sizes and separately for <1 to 300 micron sizes.
• Conduct a survey of WWTP effluents in large urban areas where previous sampling has been conducted (e.g., Los Angeles, Chicago). Determine MP distribution in influent and effluent. Duplicate treatment types in the survey to attempt to draw a relationship between treatment type and MP removal efficiency. This aspect may be better studied in a lab/pilot scale setting if the appropriate range of MP types are used.

• Determine a PNEC for a range of marine and freshwater and sediment species using realistic concentrations and types of MPs. This tests should include MPs from 300 to 500 microns, and separate tests with 1 to 300 micron MPs. The concentration exposure range must encompass RWC scenarios.

• Develop an education program based on the results of the above studies that targets WE&RF subscribers, regulators, and the public.

10. Literature Cited


32. Phuong NN, A Zalouk-Vergnoux, L Poirier, A Kamari, A Charl, C Mouneyrac, FD Lagarde. 2016. Is there any consistency between the microplastics found in the field and those used in laboratory experiments? Environ Pollut 211:111-123.


APPENDIX A

Useful Microplastics References


Key Publications


