NEW JERSEY DEPARTMENT OF ENVIRONMENTAL PROTECTION
SCIENCE ADVISORY BOARD

FINAL REPORT
Microplastics in the aquatic environment: Sources, occurrences, and currently known risks

Prepared for:
Commissioner Shawn LaTourette

Prepared by:
Ad Hoc Microplastics Work Group

Approved by:

<u>NJDEP SCIENCE ADVISORY BOARD</u>
Kropp, Richard H., M.S., P.E. (Chairperson)
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Lippencott, Robert J., Ph.D.
Najarian, Tavit, Sc.D.
Robinson, David A., Ph.D.
Rothman, Nancy, Ph.D.
Velinsky, David, Ph.D.
Weis, Judith Ph.D. (Abstained)

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REPORT PREPARATION

This report was prepared by the following members of the members of the Ad Hoc Microplastics Work Group:

- Judith Weis, Ph.D. (Chair)
- Keith Cooper, Ph.D.
- Beth Ravit, Ph.D.
- John T. Gannon, Ph.D.
- Brian Buckley, Ph.D
- Dibyendu "Dibs" Sarkar, PhD, PG

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Executive Summary

The committee addressed the five questions posed by DEP.

1. **What does the current science on microplastics (and nanoplastics) indicate in terms of adverse effects on aquatic life and aquatic ecosystems?**

   Before addressing effects, it is important to know that microplastics (MPs) are a diverse suite of contaminants with differing chemistry, additives, density, sizes, shapes, which are all critical in determining their potential adverse effects and fate. It is also important to know that the types most frequently used in experimental studies of toxicity are not the types that are most abundant in the environment. The most abundant shape in the environment are microfibers, while microspheres are most frequently used in studies. In addition, many toxicity studies use concentrations far above levels in the environment. Thus, there is a lack of realism in many studies.

   With that said, the literature reports numerous effects on gene expression, biochemistry, physiology, behavior, growth, and development in aquatic species of all phyla. There are studies that use realistic shapes, concentrations, and exposure times and find deleterious effects. There are also studies that use unrealistically high concentrations and find no effects. So, despite the abundant literature, it is difficult to generalize on effects. Environmental levels do appear to cause some harm in some species.

2. **What are the primary sources of these contaminants to rivers and streams specific to New Jersey; which forms of plastic cause the most harm; and which forms are most feasible for prevention, source reduction and/or exposure reduction?**

   The major sources are textiles, tire wear particles, city dust, marine paint, fragmented macroplastics. We know that microfibers (MFs) from textiles are the most abundant only because of collection methods that do not use nets, since long thin MFs tend go through the holes of nets and are undercounted. MFs are released from laundry into water in washing machines; the water goes to sewage treatment plants (in areas with sewers), which are the major route to get into aquatic environments. In terms of harm, studies comparing MFs with other shapes including microspheres (rare in the environment), MFs tended to cause more harm, possibly because they remain in the digestive tract longer. Tire wear particles are shed onto roads by moving vehicles and will go down storm sewers during rainfall and may be released directly into the water or sent to the sewage treatment plants. MPs that have been in the water for a time ("weathered") tend to be more toxic, having adsorbed environmental chemicals. MFs are most feasible for source reduction, prevention, and exposure reduction since people can modify how they do the laundry to reduce shedding. There are also filters available to be attached to washing machines that will capture most MFs. Public education will be necessary to implement fairly simple things that people can do. France has a law mandating that new washing machines must be equipped with a filter by 2025. Other areas of potential reduction are at sewage treatment plants – although they capture the vast majority, the numbers going out in the effluent are high. The textile industry is looking into different weaves and coatings that could reduce shedding. We know of no actions by the tire industry to reduce release of MPs.

3. **What technical steps should the DEP take to understand and manage microplastics and nanoplastics?**

   The state of California has issued guidelines with two tracks that we suggest NJ could follow.

- *Pollution Prevention*: Eliminate plastic waste at the source (products or materials from which MPs originate).

- *Pathway Interventions*: Intervene within specific pathways (stormwater runoff, wastewater, aerial deposition) that transport MPs into California waters.
- *Outreach and Education*: Engage and inform the public and industries of MP sources, impacts, and solutions.

The second track outlines a comprehensive research strategy to enhance the scientific foundation for microplastic monitoring, source identification, risk assessment, and development of management solutions, and includes:

- *Monitoring*: Standardize a statewide monitoring approach. Understand and identify trends of MP pollution statewide.
- *Risk Thresholds and Assessment*: Improve understanding of impacts to aquatic life and human health.
- *Sources and Pathways Prioritization*: Identify and prioritize future management solutions based on local data.
- *Evaluating New Solutions*: Develop and implement future pollution prevention and pathway intervention solutions.

4. To what extent does this issue currently affect New Jersey's water quality/designated use support, tourism, economy, and Environmental Justice communities?

The Department is interested in any research findings and papers that could support the Department in developing policy related to the above.

To affect tourism or the economy, the public would have to believe that MP exposure in coastal areas is a hazard to their health, and therefore avoid going to NJ coastal areas. Since there is no evidence that MP exposure in coastal areas is indeed a health hazard such a public response is unlikely. EJ communities may be exposed to higher levels by eating fish from polluted water, but there is no evidence that levels from eating MPs from contaminated fish will cause harm; more contaminants are acquired from the fish tissue itself than from MPs in the fish tissue.

The Clean Water Act Section 101(a)(2) goals for the protection and propagation of fish, shellfish, and recreation in and on the water should be considered for impacts, as well as the use as a potable water supply after conventional treatment, per the New Jersey Surface Water Quality Standards at N.J.A.C. 7:9B-1.12. Any studies related to the impact of microplastics on the fishing or shellfishing industry would be valuable in assessing impacts to New Jersey's economy.

To affect fish and shellfish resources, there would need to be evidence that MPs in fish and shellfish are reducing their growth, health, and/or reproduction. The fishing or shellfishing industry could be negatively impacted if the presence of MPs in the fish or shellfish posed a food safety issue causing people to avoid eating them. There is no such evidence available. Some preliminary calculations have been made on how MPs in bivalves might contribute to human exposure to contaminants. The general conclusions are that the risks are very low.

5. Are there any research initiatives that the SAB would recommend pertaining to the contaminants of emerging concern in addition to the ongoing initiatives (e.g., municipal wastewater treatability, analytical methods for municipal wastewater)?

Some of the research recommendations are in #3, others on sources, fate, and effects are in the text.

Background:

The Department is looking for current research that may help it evaluate and quantify the risk microplastics pose to aquatic life and ecosystems, as well as potentially develop aquatic life criteria for microplastics. The Department recognizes that the research on the impact of microplastics and nanoplastics on aquatic life and aquatic ecosystems is still in its early phase, but requests that the SAB consider the impacts of both to the extent possible. This report addresses the following questions:

- Question 1. What does the current science on microplastics (and nanoplastics) indicate in terms of adverse effects on aquatic life and aquatic ecosystems?
- Question 2. What are the primary sources of these contaminants to rivers and streams specific to New Jersey; which forms of plastic cause the most harm; and which forms are most feasible for prevention, source reduction and/or exposure reduction?
- Question 3: What technical steps should the DEP take to understand and manage microplastics and nanoplastics?
- Question 4. To what extent does this issue currently affect New Jersey's water quality/designated use support, tourism, economy, and Environmental Justice communities? The Department is interested in any research findings and papers that could support the Department in developing policy related to the above.
- Question 5. Are there any research initiatives that the SAB would recommend pertaining to the contaminants of emerging concern in addition to the ongoing initiatives (e.g., municipal wastewater treatability, analytical methods for municipal wastewater)?

The mechanisms of microplastic production are directly related to their point of origin. Figure 1.1 depicts the theoretical fate of plastic debris as it physically breaks down into smaller and smaller components. One of the largest sinks for these plastic particles, whether initially airborne or waterborne, are surface waters, where particles can be ingested by the biota within the aquatic system.

Polymers denser than water tend to sink to the bottom sediments. What is not depicted are the numerous non-covalently bound additives and associated manufactured compounds that may leach from the plastic fibers or particles. This is important when trying to assess the toxicity associated with plastic polymers since the additives have their own innate toxicity. There are a wide range of these additives (phthalates, dyes, metals) used depending on the manufactured material and its end use (fire retardants, PFAS).

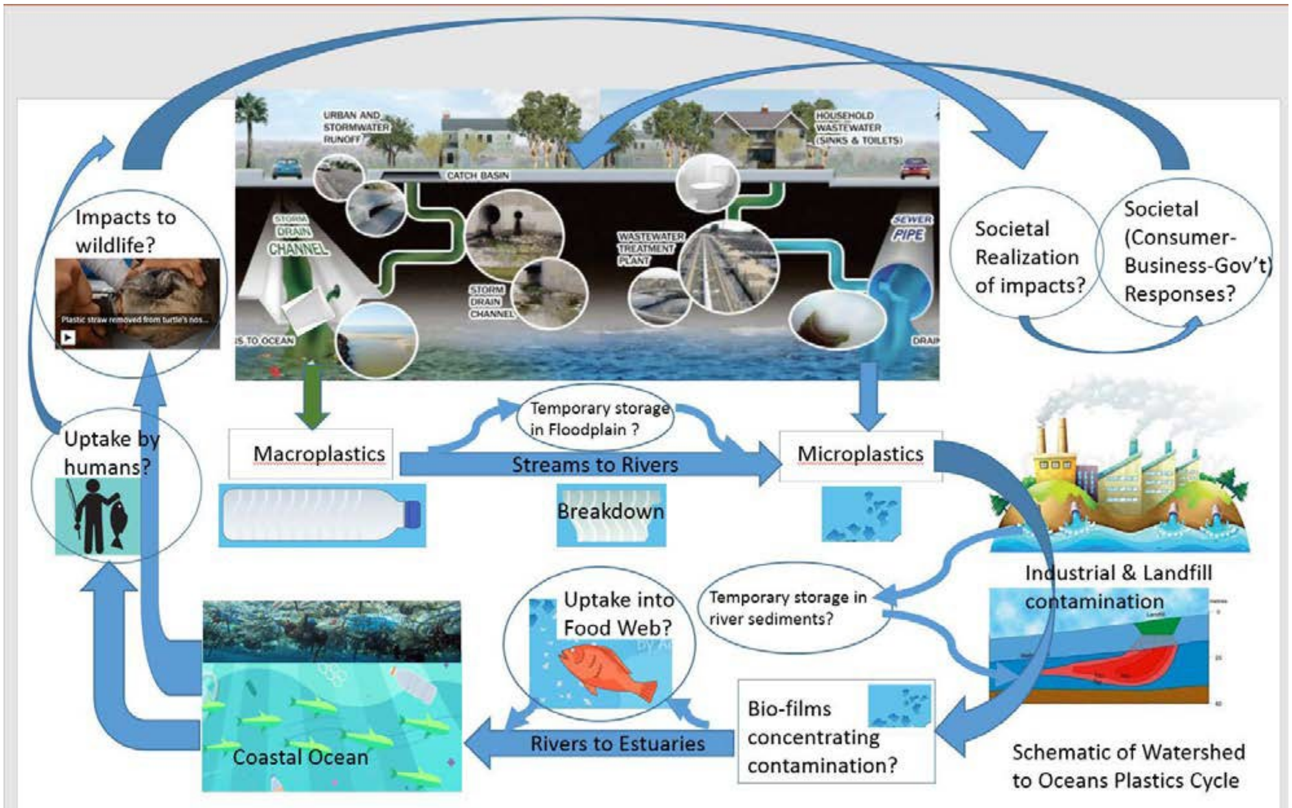

Figure 1.1 Schematic showing the theoretical fate of plastic debris through secondary processes over time and the breakdown to smaller and smaller fibers or particulates and their theoretical uptake in human and other organic food webs. Credit: Rick Lathrop Rutgers School of Environmental & Biological Sciences.

Question 1. What does the current science on microplastics (and nanoplastics) indicate in terms of adverse effects on aquatic life and aquatic ecosystems?

Before launching into a discussion of the various adverse microplastic effects found in experimental studies on aquatic life and ecosystems, there is background information and research limitations that are essential factors when evaluating research results. Plastics are not a contaminant, but a "suite of contaminants" as they have many different chemical structures, additives, sizes, shapes, density, toxicity, behavior in the environment and reactivity. Current literature suggests that plastic particle toxicity depends, in part, on particle size, shape, and concentration (Kögel et al., 2020). Therefore, it is important to consider each of these parameters when assessing risk to aquatic organisms. For example, nanoplastics more readily pass through biological barriers and accumulate in organs. This is due to both passive and active transport mechanisms for handling particulates based on size that are innate in living organisms. As microplastics (MPs) in the environment break down to become nanoplastics, adverse effects to aquatic life and ecosystems can potentially increase or decrease depending on the receptor organism.

There are a great number of experimental studies that show adverse MP effects on aquatic life. However, there is a mismatch between both the types and concentrations of MPs used in most laboratory studies and those that are currently thought to be most abundant in the aquatic environment (Weis and Palmquist 2021). Available technologies are currently not adequately addressing nanoparticles, which makes interpreting exposure risks obtained from these studies very difficult or impossible.

Bucci et al. (2020) performed a meta-analysis of the literature and found a misalignment between field and laboratory studies - different shapes (fibers being the most common in the field; spheres in the lab (Gago et al., 2018)); virgin MPs in the lab versus aged MPs with associated biofilm in the field; and different sizes of MPs (Bucci et al., 2020). They noted that only 17% of the concentrations used in lab studies have been seen in the environment, and that 80% of the particle sizes used in experimental studies are below the sizes observed in environmental samples. This mismatch may be a result of inadequate field sampling techniques to capture the smaller micro and nanoparticles. However, accurately estimating environmental concentrations is also problematic because mesh size of the majority of sampling nets has been too large to capture smaller micro- and nano-sized MPs. Fibers, which are the most common MP type in aquatic ecosystems in coastal areas (Belzagui et al., 2021), are often not captured in sampling with nets. These fibers come mostly from washing of synthetic textiles (Gago et al., 2018) and breakdown of cigarette butts. MP concentrations in marine environments range from approximately 1 mg L^{-1} to 1 ng L^{-1}, often <20 MPs m^{-3} in water (Lenz et al., 2016), but most lab studies use far higher concentrations. Lenz et al. (2016) state that experimental exposures are two to seven orders-of-magnitude higher than those in the environment.

Nano- AND MICRO-PLASTIC TOXICITY FACTORS

The severity of observed effects is due to a variety of factors, including the MP concentration, shape, particle size, polymer type, and compounds associated with the plastic polymer(s). All of these factors would be important for determining uptake and effects.

Chemical Polymer Structure

Plastics are comprised of a very large number of different chemical structures which provide different physical properties for commercial and industrial uses. Therefore, it is not appropriate to consider plastic waste as a single entity. When assessing potential biological reactivity of a specific plastic, it is essential that the chemical structure be considered. Although shape, size and other factors are critical, different plastics will have different reactivities with the various receptor organisms. In addition to biological reactivity, a plastic's chemical makeup will affect several important physical factors such as density, charge, decomposition rates and chemical reactivity (Figure 1.2). The chemical makeup of the plastic is critical when assessing toxicity and potential risk to ecosystem receptors.

Figure 1.2. Polyethylene Terephthalate (PET) vs Polyethylene (PE).

Shape

Microspheres can be purchased in uniform sizes and shapes, with added fluorescence or dyes allowing them to be visualized in tissues (Ward et al., 2019). Therefore, they are useful for laboratory experiments and can give some insight into the chemical classes' reactivity and tissue responses. But they are not representative of the most prevalent MPs in the environment.

Plastic fragments (irregular shapes and sizes) can be obtained through physical means (grinding) and natural photo and biological degradation. Separation by size can be done through sieves or sized filters. The surface irregularity of the plastic particle can result in damage to epithelial surfaces that it contacts, may result in direct cellular damage, and or may be trapped in epithelial folds that would result in an inflammatory response. Smaller particles also generally have increased surface area and charges that other chemicals adhere to, which increases their potential for eliciting a biological response (Mehinto et al., 2022).

Despite not being so easily obtained, researchers have found ways to study effects of microfibers (MF), which are the ***predominant shape found in the environment (***Belzagui et al., 2021; Covernton et al., 2019**).** They can be obtained from ropes, dryer lint, textile samples, microfilaments used to make synthetic textiles, and other sources, and have been used in experimental studies.

Size

Size is a critical factor in uptake and egestion, and differences in size thresholds may affect the impacts of interest or biological effects. Over time in the environment, size is dynamic, with environmental factors potentially reducing microplastics to nanoplastics. Organisms themselves may also reduce MP size via passage through the gut, potentially creating nano sized particles. For small organisms such as zooplankton, particles larger than their mouth opening cannot be ingested, although particles may adhere to the body surface. In experiments, using a size appropriate for a specific organism's feeding range size does not make it unrealistic, and is very appropriate for the organisms which take up particles in that size class. Smaller particles are expected to be better able to penetrate through tissues, and in some laboratory studies smaller particles have exhibited greater toxicity, but that is not always the case. However, using only particles small enough to be eaten is also unrealistic, because a higher surface to volume ratio maximizes exposure (Bour et al., 2021).

MPs in the aquatic environment can also increase in size and density through acquisition of a community of microbes that form a biofilm on the MP surface. Microbial production of extracellular polysaccharides can also trigger formation of aggregates composed of MPs, microorganisms, and organic materials.

Concentration

Horton et al. (2017a) reviewed particle counts in marine and surface freshwaters and found them to be extremely variable. As long as the mass is known, it can be converted to a concentration. MP concentrations in marine surface waters are reported from 0.0005 particles L^{-1} to 16 particles L^{-1} with intermediate concentrations. The concentration depends on whether the samples were taken with nets or not. Freshwater concentrations, in general, are around the low end of marine concentrations. Burns and Boxall (2018) noted that MP concentrations in the environment are orders of magnitude lower than those used in studies of effects on biochemistry, reproduction, growth, feeding, and inflammation in aquatic organisms. In their review, Thomas et al. (2021) found 14 papers where exposure had no or minor effects. Some of these used low concentrations, but some used high concentrations. In contrast they found 81 papers that reported adverse effects, most of which used unrealistically high concentrations. Experiments do not always use unreasonably high concentrations, and some studies have found deleterious effects at ranges close to environmental concentrations. There are also studies that have found no deleterious effects at very high concentrations.

MP concentrations are not necessarily consistent or uniform within a waterbody and have been found to be higher in waters adjacent to highly developed population centers. However, the majority of current sampling methods are likely to underestimate concentrations in the water (Lindeque et al., 2020). Using different mesh sizes for sample collection results in greatly different MP counts. MPs collected with a 100 µm net were 10-fold greater than in samples collected with a 500 µm net. Covernton et al. (2019), comparing samples collected with a 300 µm mesh net versus a bulk sample filtered through an 8 µm filter, found 8.5 times as many MPs in the filtered sample, including a much greater proportion of microfibers. This is important when evaluating surface water samples, since differences in sampling methods lead to a lack of comparability between studies.

Aquatic MP density is determined by the polymer type, the number and type of microorganisms attached to the particle (biofilm), and the potential for aggregation. If MP density is greater than water density, the MP sinks to the bottom, potentially exposing benthic organisms to higher MP concentrations than those found in the water column. A review by Van Cauwenberghe et al. (2015a) found U.S. sediment concentrations of 105-215 MPs L^{-1} (sized 250 µm to 4 mm); studies in the U.K. and Europe of smaller sizes (1.6 µm – 5mm) found 20 to 3320 L^{-1}. Sediment concentrations tend to be higher than in water in UK rivers and coastal Slovenia (Laglbauer et al., 2014; Horton et al., 2017b). High concentrations (thousands of particles kg^{-1} of dry sediment) were found in river sediments in Germany, similar to the 2000–8000 particles kg^{-1} reported in coastal sediments in Canada (Mathalon and Hill 2014; Klein et al., 2015). Over time, many particles that initially float can acquire a biofilm of microorganisms, become heavier, and eventually sink to the bottom where they concentrate in the sediments.

Duration of exposure

The time of exposure is an important factor determining the magnitude of biological effects. Most laboratory exposure studies are short-term (days), which might be thought to be a counterbalance to the use of excessively high concentrations. However, short-term exposures to high levels are not equivalent to long-term chronic exposures at more realistic levels. The duration of exposure is important, especially when assessing several critical life stages (embryonic, larval, juvenile or adult), which may exhibit different responses. Subtle alteration in reproductive success may not be manifested until future generations. A recent paper examined effects of high concentrations (80 g m^{-2}) of irregularly shaped mixed MPs on freshwater benthic communities, with 100 days exposure (Stankovic et al., 2021). No effects on abundance, biomass, species richness, or community diversity were found.

Leached chemicals

Plastics contain additives such as phthalates and BPA (bisphenol a) that can leach out and cause toxicity (Seuront 2018). In addition, environmental contaminants, such as PCBs (polychlorinated biphenyls), metals, and pesticides, adsorb to plastic (Hartmann et al., 2017). Many studies on impacts of MPs use fresh ("virgin") plastic particles, but after weathering in the environment, particles acquire attached chemicals and are potentially more toxic than virgin particles (Figure 1.3). The use of pristine particles in experiments could lead to underestimation of toxicological effects due to environmental concentrations of adsorbed contaminants. MPs are considered "vectors" for moving contaminants into animals and up the food chain (Rochman et al., 2013a; Rochman 2016). Desorption of adsorbed chemicals and leaching of additives makes MPs a "cocktail" of potentially toxic contaminants (Rochman 2015)

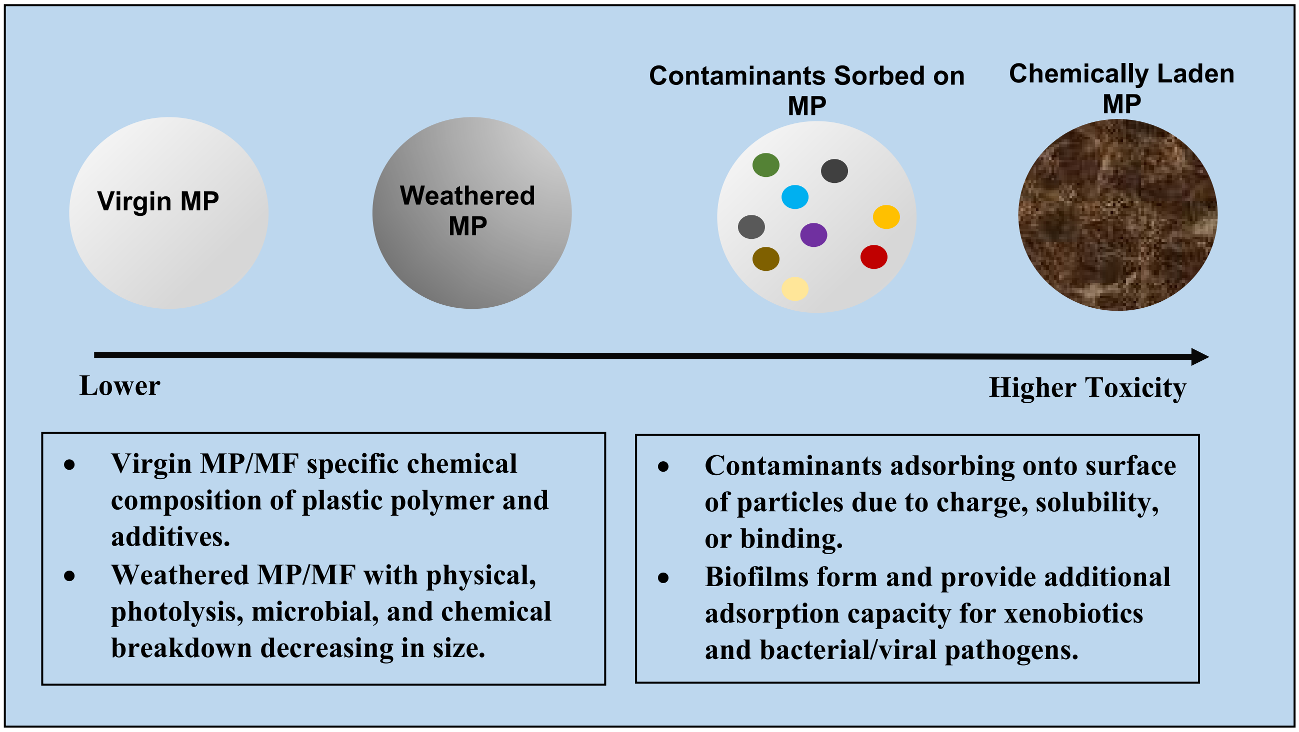

Figure 1.3. Microplastics form nuclear centers for a mixture of chemical and biological components. Over time in the aquatic environment, chemical and physical properties of microplastics change, potentially increasing toxicity. As particle size decreases, availability to smaller organisms increases. *Adapted from Omowunmi et al. 2020.*

Most studies on chemical toxicity of MPs have used microbeads and not MFs, but studies comparing the two indicate that MFs are more detrimental, which could lead to further underestimation of impacts. MFs are rarely studied for the toxicity of associated chemicals, but a study of MFs from cigarette butts indicated toxicity of chemicals from tobacco (Belzagui et al., 2021). MFs from textiles have unique dyes and hundreds of other chemicals used in the manufacture of textiles (e.g., antimicrobials, fire retardants, stain resistance and anti-wrinkle chemicals) which have been shown to be highly toxic (Selvaraj et al., 2015; Athira and Jaya 2018; Carney et al., 2021). Furthermore, chemical additives have been detected in aqueous leachates of virgin and aged (photodegraded) MFs (Sait et al., 2021). These have also been greatly understudied.

A key question is how tightly bound the chemicals are to the plastic, which affects how available they are to biota. Availability depends on the polymer, the chemical, and the environment. After ingestion additives may leach and adsorbed chemicals may desorb (Turner et al., 2015; Suhrhoff and Scholz-Böttcher 2016; Anbumani and Kakkar 2018). Since ingestion is the most often studied type of exposure, it is important to study what proportion of the additives and attached contaminants can be "pulled off" during the time that the MPs are going through the gut. This will vary with the chemical, the type of MP (see Browne et al., 2013), and the complexity, length, and chemistry of the gut (Mohamed Nor and Koelmans, 2019). Bakir et al. (2014) investigated the ability of polyvinylchloride and polyethylene MPs to sorb and desorb various organic chemicals. Desorption measured in seawater and under simulated gut conditions was faster with gut surfactant and in conditions simulating warm blooded organisms. Other studies find that chemicals bind so tightly to MPs that bioavailability is reduced and there is a smaller transfer of toxic chemicals from ingested plastics than from sediments, indicating that plastic is a less important source of contaminants (Beckingham and Ghosh 2017; Kleinteich et al., 2018; Rehse et al., 2018). Mohamed Nor and Koelmans (2019) also examined the transfer kinetics of PCBs between microplastics and gut fluids. Often, clean microplastics could scavenge PCBs from "dirty" food inside the gut, demonstrating that contaminant transfer in these situations is context dependent.

Biofilms may also play an important role in toxic chemical transport by MPs and their effects (Mincer et al., 2016). This is relevant to toxicity, since a biofilm is often the reason that animals are attracted to and consume MPs (Savoca et al., 2017). This study found that microbes in the biofilm produce dimethyl sulfoxide (DMS), which is an attractant odor for plankton and seabirds (Savoca et al, 2017). The community of microbes surrounding MPs (the "plastisphere") is the subject of intense study by microbiologists. A review of information about the plastisphere noted that taxa found include *Pfiesteria*, a dinoflagellate which is often highly toxic (Amaral-Zettler et al., 2020; Kettner et al., 2019). Among the bacteria associated with MPs are *Vibrio*, suggesting that MPs could be vectors for pathogenic microorganisms (Oberbeckmann et al., 2015).

Nano- AND MICRO-PLASTIC EFFECTS ON ORGANISMS

The diversity of organisms inhabiting the aquatic ecosystem indicates the importance of recognizing that each large class of organisms may respond differently to nano and microplastics, due to their organ system structures, physiology and innate biological responses.

Algae and Small Invertebrates

Phytoplankton are primary producers at the base of aquatic food chains. Microalgal phytoplankton are a primary food source for zooplankton communities that range in size from microscopic organisms to larval stages of crustaceans and fish species. Although less studied than higher organisms, MP effects on planktonic communities could potentially contribute to aquatic food web disruption, trophic transfer of nano- and micro-plastic pollution, and bioaccumulation of MPs in higher tropic level consumers. Mendrik et al. (2021) found that fibers, but not spheres, reduced photosynthesis of algal symbionts of *Acropora* species corals (0-10^3 particles/ml), with a 41% decrease in photochemical efficiency after 12 days.

Mao et al. (2018) exposed *Chlorella pyrenodosa* to polystyrene microbeads 0.1 and 1.0 µM in diameter. Response to MP exposure was determined by the algal growth stage (lag, logarithmic, stationary, death) over a 30-day incubation. During lag to early logarithmic stages (day 13), at very high concentrations of MPs (10, 50, and 100 mg/L) dose-dependent reduction of photosynthesis activity, production of oxidative stress, cell wall distortions, and cell wall thickening were observed. However, by day 25 (end logarithmic to stationary stage) cell structure returned to normal, photosynthetic activity and growth increased, and no irreversible negative effects were observed. Production of extracellular polymeric substance (EPS) increased under MP exposure and hetero-aggregations (EPS, MPs, and algal cells) formed.

Prata et al. (2019) reviewed 16 studies of MP effects on microalgae populations and concluded that at current environmental concentrations there are limited or no toxic effects on growth, chlorophyll content, photosynthesis activity, or reactive oxygen species. Most studies have been unable to find an EC_{50} due to the high MP concentrations needed to induce toxicity. Two works reporting EC_{50} include Bergami et al. (2017); reporting EC_{50} = 12.97 mg L−1 for 0.04 µm polystyrene (PS) and Casado et al. (2013) reporting EC_{50} = 0.58 mg L−1 and 0.54 mg L−1 for polyethyleneimine polystyrene (PS-PEI), for sizes of 0.05 and 0.1 µm, respectively. Impairments after MP exposure appear to be temporary, and adaptive responses lead to recovery. Some studies with smaller sized (0.02 – 1.0 µm) MPs or positively charged MPs found impairment of photosynthesis, growth and chlorophyl reduction, and reactive oxygen species production (Prata et al., 2019). However, the majority of these responses occurred at MP concentrations far greater than environmental concentrations.

In addition to attracting chemicals from the environment, MPs also attracted microorganisms that colonize the particle, which can affect the microorganisms as well as the fate and transport of the MP. A review by Nava and Leoni (2021) describes interactions between MPs and microalgae species commonly found in biofilms that form on plastic particles. Eleven studies were reviewed that test microalgal impairment(s) after exposure to various MPs of multiple sizes and concentrations; 19 of 41 end points (46%) showed no significant effects, 3 end points recovered before the study was over, and 2 end points were a significant increase in algal growth. The 17 significant impairments observed were under MP concentrations significantly higher than environmental concentrations. Microalgae colonization of MPs varied based on polymer type, and surface roughness increased microalgal attachment. Common MP colonizers were cyanobacteria, including filamentous *Phormidium* in the North Atlantic, and dinoflagellates, creating potential "hotspots" of toxic compound producers; diatoms are also common members of colonized MP surfaces. Microalgae biofouling can "camouflage" MPs, making these aggregates a potential food source for higher trophic levels. Aggregations of MPs and microalgae also increase MP particle density, contributing to the downward movement of surface particles through the water column and into sediments, potentially creating a sink for MPs and increasing benthic organisms' MP exposure (Botterell et al., 2019).

Laboratory zooplankton studies used microbeads at mostly environmentally unrealistic concentrations. A review by Botterell et al. (2019) found only 10% of the 34 studies reviewed were conducted under field conditions, where the preponderance of MPs ingested were microfibers; in the 24 laboratory studies, all but 1 used only MP beads. Frydkjær et al. (2017) compared spheres with fragments and reported that elevated concentrations of PE MPs decreased mobility of *Daphnia magna*; irregular fragments (10–75 µm) had greater effects than beads (10–106 µm). *D. magna* egested regular-shaped PE faster than irregular ones, which were retained longer and had more severe effects. *Daphnia* inhibition of feeding and growth depended on the size of MP fragments (An et al., 2021). Survival after exposure to 17 and 34 µm fragments was significantly lower than survival after exposure to spheres. Small fragments reduced feeding and retention in the digestive system. Cole et al. (2015) fed the copepod *Calanus helgolandicus* a diet of 20 µm MP beads and microalgae for 9 days and found a 3-fold increase in a downward shift in the size of algae (~12 µm) consumed. No significant differences were seen in egg production rates, respiration, or survival, but there was a 40% decrease in estimated ingested biomass, suggesting potential nutritional impairments.

Lee et al. (2013) fed zooplankton MP beads in combination with phytoplankton. No LC_{50} could be calculated for acute exposures. In two generation chronic tests, larger beads (6 µm) reduced fecundity, but did not cause mortality; small beads (0.5 µm beads at a concentration greater than 12.5 µg/mL) caused F_0 generation mortality of nauplii and copepodites and a significant decrease in survival of the F_1 generation, suggesting nano and small micro sizes were more toxic. ***As particles are worn down to smaller sizes in the environment, they become more available to a larger number of planktonic species if they are in a filterable range for the organism.*** Besseling et al. (2014) found at MP concentrations 10^6 higher than environmental freshwater concentrations and 10^2 higher than marine concentrations, nano MP beads reduced growth and chlorophyll concentrations in the green algae *Scenedesmus obliquus*, and reduced body size and increased neonate malformations 68% in *D. magna*.

There are a few studies that have sampled natural zooplankton populations to determine environmental MP exposures and effects. Based on a limited number of field studies, Botterell et al. (2019) suggest bioaccumulation of MPs varies by taxa, with a trend of omnivore > carnivore > herbivore, potentially influenced by feeding mode. Field sampling by Sun et al. (2018) of 10 zooplankton taxa in the East China Sea found no significant differences in pellet or fragment sizes accumulated by various taxa, but significant differences were found in the lengths of retained fibers (54.6% of total ingested MPS observed), which could be a function of differing prey size; 80% of the ingested MPs were smaller than 330 µm. Two-thirds of the ingested MPS were accumulated in copepods (42.3%) and jellyfish (4.8%).

Desforges et al. (2015) examined MP ingestion by *Neocalanus cristatus* (copepod) and *Euphausia pacifia* (euphausiid), zooplankton found in the north Pacific. Species differences were observed in encounter rates (no. of plankton/plastic particles (1 MP/34 copepods; 1 MP/17 euphausiids); ingested particle sizes (555.5 $\pm$ 148.7; 816.1 $\pm$ 107.7); and % fibers (43.9 $\pm$ 12.3; 68.3 $\pm$ 12.8) in *N. cristatus* versus *E. pacifica*, respectively. Concentrations ingested were inversely related to MP size. Both species are filter feeders; the lower encounter rate for *N. cristatus* may be the result of their higher population density (27.9 versus *E. pacifica* 1.3/m^3) causing more intraspecific competition for food. Sun et al. (2017) conducted field sampling in the South China Sea, collecting 5 zooplankton groups (copepods, chaetognaths, jellyfish, shrimp, fish larvae). Fibers were the largest proportion (70%) of ingested MPs, with the highest proportion (54-79% of the total) found in copepods. The copepod MP encounter rate was the lowest and fish larvae rate the highest, increasing with the increase in trophic level. Steer et al. (2017) conducted a similar study in the English Channel, looking at MP ingestion in 23 species of planktonic wild fish larvae (N =347). Ten larvae (2.9%), consisting of 5 species [*M. merlangus* (whiting), *M. variegatus* (thickback sole), *T. minutus* (poor cod), *C. lyra* (common dragonet), *A. anguilla* (European eel)] were found to contain MPs. The encounter rate (no. of MP ingested/no. fish dissected) was low, ranging from 0.7% – 5.3% (note encounter rate calculated differently in the Desforges et al. study). Ingested MPs consisted of 83% fibers (100 – 1,100 µM length) and 17% fragments (50 – 100 µM).

Larval midges, *Chironomus riparius*, were exposed to PET MFs, and survival, time until emergence, growth, head capsule length, and general stress response were examined after 28 days (Setyorini et al., 2021). They used artificial sediments spiked with MFs at 500, 5000, and 50,000 particles/kg sediment dw, (500 particles/kg sediment was found in Lake Ontario, Canada and 4900 particles/kg sediment was found in the Rhine River). Larvae ingested the microfibers which were later found in the adults, but no significant effects were seen on time until emergence, head capsule lengths, weight, or HSP 70 (stress response). Ziajahromi et al. (2018) exposed freshwater *Chironomus tepperi* midge larvae to environmentally relevant 500 MP PE particles of different sizes (1-4, 10-27, 43-54, 100-126 µm). MPs in the 10-27 µm size range produced significant reductions in size, growth, body length, head capsule size and emerging rate of adults after 10 days.

Sea urchin (*Paracentrotus lividus*) embryos were exposed to PVC leachates from beached (weathered) versus virgin MPs. Weathered beached MPs with polycyclic aromatic hydrocarbons (PAHs) and polychlorinated biphenyls (PCBs) attached produced severe, consistent, and specific developmental abnormalities. Embryos exposed to virgin MP leachates without additives or environmental contaminants developed normally, suggesting that the abnormalities caused by beached MPs are due to exposure to adsorbed contaminants and/or industrial additives (Rendell-Bhatti et al., 2021).

Macroinvertebrates

Polychaetes

Knutsen et al. (2020) found MP concentrations orders of magnitude higher in tube-dwelling polychaetes than in sediments from the same sampling location, and the composition of MPs in the polychaetes differed from those in the sediments. Concentrations of MPs were 6- to 11-fold higher in the polychaete-produced tubes than in the animal's soft tissues.

The polychaetae *Hediste diversicolor* took up a variety of microplastics ranging in size from 100-0.45 µm (PE, PEVA, PP, PA) into muscle and other tissues (Missawi et al., 2022). The MPs were confirmed by light microscopy and Raman spectroscopy. Autophagy, cytoskeletal alterations and altered metabolomics involving amino acids, energy storage and osmoregulation were observed. This is an important finding because it demonstrated that under environmentally relevant sediment concentrations, the observed alterations would impact the polychaeta's survival.

Lugworms (*A. marina*) were exposed to up to 100 g of MP per liter of sediment; reduced feeding was the only effect seen (Besseling et al., 2012). Other lugworm studies included transfer of PCBs (polychlorinated biphenyls) from PS MPs (Besseling et al., 2017). Effects on survival, activity, bodyweight, and transfer of PCBs were studied. The concentration of MPs in spiked sediment was directly related to the uptake of MPs and to weight loss. Reduced feeding was seen at a dose of 7.4% dry weight. There were statistically significant effects on bioaccumulation and fitness, but the magnitude of effects was small. In similar studies, PE MPs were not a measurable vector of PCBs. Browne et al. (2013) exposed *A. marina* to sand with 5% PVC MPs spiked with nonylphenol and phenanthrene (pollutants) and triclosan and PBDE-47 (additives). The pollutants and additives were transferred into gut tissues and caused some effects. However, sand transferred greater amounts of the pollutants to the worms than did MPs. Uptake of nonylphenol from MPs or sand reduced the ability to remove pathogenic bacteria. Uptake of Triclosan from PVC reduced the ability to burrow and caused mortality (both by >55%) while PVC alone made the lugworms more susceptible to oxidative stress.

Mollusks

Bour et al. (2018) exposed two bivalves, the smooth nut clam (*Ennucula tenuis)* and glossy furrow shell (*Abra nitida),* to PE MPs at 1, 10, and 25 mg kg^{-1} of sediment for four weeks. Three sizes (4–6; 20–25 and 125–500 µm) were used. No effects were seen on survival, condition index or burrowing behavior. *A. nitida* had a significant decrease of protein after exposure to the largest particles at all concentrations. No changes were seen in protein, carbohydrate, or lipid in *E. tenuis*, but in those exposed to the largest particles total energy decreased in a dose-related manner.

Microfiber uptake by Asian clams (*Corbicula fluminea*) varied by polymer type and size. Clams took up more fibers when exposed to higher concentrations and were more likely to take up fibers of smaller size (Li et al., 2019). In a more realistic exposure, Alnajar et al. (2021) exposed mussels (*Mytilus galloprovincialis)* for seven days to MFs at 56–180 mg L^{-1} (far higher than environmental concentrations), and observed reduced clearance rate, gill and digestive gland abnormalities, and increased DNA damage. They thought effects were due to both the MFs and the chemicals mobilized from them in the water or digestive tract, since increases in trace elements (e.g., zinc) in the exposure medium were seen with increasing MF concentration. However, no effects were seen in *M. galloprovincialis* mussels exposed to 10 and 1000 MP mL^{-1}, or the oyster *Ostrea edulis* exposed for 60 days to 80 µg L^{-1} (Green 2016; Goncalves et al., 2019). Mussels *(Mytilus edulis*) fed plankton with MF concentrations up to 30 MF mL^{-1} exhibited reduced filtration rates (Woods et al., 2018). In a rare paper comparing effects of PET MFs of different lengths (50 and 100 µm), mussels (*Mytilus galloprovincialis*) were exposed to environmental (0.5 µg L^{-1}) and high (100 mg L^{-1}) concentrations for four days (Choi et al., 2021). Short MFs were found in the lower intestinal organs, while long MFs were seen only in the upper intestinal organs. Both sizes caused necrosis, DNA damage, production of reactive oxygen species, nitric oxide, and acetylcholinesterase. Effects occurred at environmental concentrations for DNA damage (long MFs) and AchE activity (short MFs).

Effects on the clam (*Scrobicularia plana*) of LDPE MPs (11–13µm), with and without adsorbed contaminants (benzo[a]pyrene—BaP and perfluorooctane sulfonic acid—PFOS), were studied (O'Donovan et al., 2018). Environmentally relevant concentrations were adsorbed onto MPs to evaluate their role as a vector for toxic chemicals after ingestion. Clams were exposed at 1 mg L^{-1} for 14 days and were sampled after three, seven, and 14 days. BaP accumulation in tissues, enzymatic biomarkers, genotoxicity in the gills, digestive gland, and hemolymph, and neurotoxicity were evaluated. Results suggested that physical injury of gills by MPs could affect the biomarkers. The digestive gland was less affected by physical damage from virgin MPs than by MPs with adsorbed chemicals, indicating desorption and toxic effects.

Several studies have compared the toxicity of fresh (virgin) MPs with "weathered" MPs that have been in the environment and have had time to adsorb contaminants from the surrounding water, or with virgin MPs that have been spiked with chemicals. Periwinkle snails (*Littorina littorea*) were exposed to leachates from virgin and beached MP pellets (Seuront 2018). Virgin pellets impaired vigilance and antipredator behavior a bit, but beached pellets severely inhibited the behaviors, indicating that MP leachates may have serious consequences for animals that use chemosensory cues to escape predation.

Crustaceans

There are studies that find minimal or no effects from high MP concentrations on crustaceans. Amphipods (*Echinogammarus marinus, Gammarus pulex)* had no impacts after exposure to 100,000 MP L^{-1} and 4,000,000 MP L^{-1} (Bruck and Ford 2018; Weber et al., 2018). However, Chua et al. (2014) exposed amphipods (*Allorchestes compressa*) to MPs spiked with polybrominated diphenyl ethers (PBDEs) and found the amphipods assimilated the PBDEs. Studies have compared responses to fibers vs. spheres vs fragments. The amphipod *Hyalella azteca* egested fibers more slowly than spheres, but eventually both shapes were completely egested (Au et al., 2015). MFs had greater toxicity than spheres, possibly because of slower gut passage. In a 28-day feeding study, Blarer and Burkhart-Holm (2016) studied effects of fibers and spheres on feeding rate, assimilation efficiency and weight of the amphipod *Gammarus fossarum*. Both shapes were ingested and egested, but only the fibers caused impairments. Horn et al. (2020) used polypropylene rope as a source of microfibers, the concentrations of which were based on concentrations found in the beach where the experimental subjects, mole crabs (*Emerita analoga*), were collected. Animals were exposed for 71 days, time for two reproductive cycles. Exposed adult crabs had increased mortality and impaired reproduction and embryo development.

Lobster (*Homarus americanus*) larvae were exposed to 0, 1, 10 and 25 MF mL^{-1} (Woods et al., 2020). Only the highest concentration decreased early larval survival; the timing and/or rate of molting was not altered. Larvae and post-larvae accumulated MFs under the carapace. This phenomenon has been observed in other larval crustaceans following exposure to finely suspended oil. The trapped materials can change buoyancy and swimming ability. Ingestion increased with larval stage and MF concentration; oxygen consumption rates were reduced in later larval stages exposed to high MP concentrations.

Hermit crabs (*Pagurus bernhardus*) living in suboptimal shells were exposed to polyethylene particles at 25 particles L^{-1} (Crump et al., 2020), which is lower than most exposure studies. After five days, crabs showed impaired shell selection. They were less likely to enter optimal shells and took longer than controls to contact and enter an optimal shell. MPs therefore impaired information-gathering and processing, which are essential survival behaviors.

Echinoderms

Mohsen et al. (2021) exposed sea cucumbers (*Apostichopus japonicus)* to MFs for 60days . Food was mixed with MFs at 0.6 and 1.2 MFs g^{-1}, based on environmentally relevant concentrations, and at 10 MF g^{-1} for a worst-case scenario. Exposures did not significantly affect growth or fecal production. However, acid and alkaline phosphatase activity were altered, and total antioxidant capacity was reduced in juveniles and adults.

Fishes

In the marine medaka (*Oryzias melastigma*), body weight, adipocyte size and liver lipids were significantly increased in fish exposed to large (200 µm) PS MPs (10 mg L^{-1} for 60 days), while fish exposed to the same concentration of smaller (2 and 10 µm) MPs had fibrosis and inflammation of the liver (Zhang et al., 2021a). Larger particles did not enter the circulatory system but impacted intestinal microbes. Gut microbial diversity and composition responded in exposed fish, particularly to large particles.

In a study of early life stages, marine medaka (*O. melanostigma*) embryos were exposed to MPs spiked with benzo(a)pyrene (MP-BaP), perfluorooctanesulfonic acid (MP-PFOS), or benzophenone-3 (MP-BP3) for 12 days (Le Bihanic et al., 2020). The MPs attached to the egg chorion (outer membrane) but did not penetrate it. Embryos exposed to virgin MPs showed no effects, but those exposed to MPs with PFOS had decreased survival and did not hatch. Those exposed to MPs spiked with BaP or with BP3 showed developmental anomalies, reduced growth, and abnormal behavior. Compared to similar water concentrations, BaP and PFOS spiked on MPs were more embryotoxic. Data suggest pollutant transfer by contact of MPs on the chorion; smaller particles had more severe effects than larger ones.

Transfer of contaminants from MPs into organisms has been reported. Japanese medaka (*Oryzias latipes*) were fed diets for ten weeks with 500, 1000, or 2000 µg g^{-1} of 10 µm fluorescent PS microspheres during maturation from juveniles to reproductive adults (Zhu et al., 2019). No changes in mortality, behavior, or growth were seen, and MPs were egested after 3–4 days. However, females had dose-dependent decreases in the number of eggs, and histological analysis showed changes in spleen and kidney even though microscopic examination, histologic sections, and scanning electron microscopy found no MPs in any organs. Since MPs were not in tissues, toxicity was attributed to chemicals leached from the MPs. Hu et al. (2020a) found abnormalities in MF-exposed medaka (10,000 MFs L^{-1}) for 21 days), including pathologies in the gills. They acknowledged that their observations (after very high exposure concentration) could have been caused by mechanical damage and/or chemicals. They discussed benzothiazole textile dyes induced gill alterations that resemble those from MF exposure (Avagyan et al., 2015). Rochman et al. (2014) exposed adult medaka (*O. latipes*) to PE MPs and found changes in estrogen receptor-mediated gene expression and testis histopathology, implying altered endocrine function. These effects resemble those caused by the plastic additives di-(2-ethylhexyl)-phthalate (DEHP) and its metabolite mono (2-ethylhexyl)-phthalate (MEHP) (Ye et al., 2014).

Roch et al. (2021) found that rainbow trout evacuated 50% of particles in 12 h for 42.7 µm particles and in 4 h for 1086 µm particles (less than the time for evacuating food). In contrast, the differences between sizes for evacuation by common carp were 7 h for 42.7 µm particles and 4.6 h for 1086 µm particles. They concluded that large particles in rainbow trout must be actively transported out of the stomach, since they had shorter evacuation times than food, while in carp, evacuation rates of all particle sizes were in the same range as food evacuation rates. The digestive tracts of these fish are quite different, and it may not be the size, but the specific organism physiology that determines egestion rates. PET granular particles (approximately 150 µm diam) and fibers (approximately 3–5 mm L^{-1} and 20 µm diam) were compared for effects on zebrafish (*Danio rerio*) embryos (Cheng et al., 2021). Both types accelerated blood flow and heart rate and inhibited hatching.

Goldfish (*Carassius auratus*) exposed to 0.25 and 8 µm polystyrene MPs (environmental concentrations) had enzyme changes and histological lesions more severe than when exposed to smaller MPs (Aborghouei et al., 2021). Bucci et al. (2021) exposed larval fathead minnows to virgin and environmental MPs; environmental MPs caused more drastic changes in length and weight and almost six times more deformities than virgin MPs. Gilt head sea bream *(Sparus aurata)* juveniles were exposed to virgin and weathered MPs for 21 days (Rios-Fuster et al., 2021). Fish were analyzed for enzyme biomarkers and behavior, specifically social interactions and feeding. Results indicated greater stress from weathered than virgin MPs. Fish exposed to either type of MPs were significantly bolder than controls during social interactions.

Herring (*Clupea harengus*) larvae were tube-fed up to 200 MP spheres spiked with ^{14}C-labelled PCB-153; controls were tube-fed an isotonic solution (Norland et al., 2021). Most larvae egested all or almost all MPs by 24 h and tracer levels in exposed fish were not significantly different from controls. There was no significant transfer of PCB into larvae which had gut-transit times of <24 h. Koelmans et al. (2014) assessed leaching of nonylphenol (NP) and BPA in the digestive system of *A. marina* and cod, (*Gadus morhua)*. They used a biodynamic model to estimate the relative contribution of plastic ingestion to total exposure to chemicals in the plastic and concluded that MP ingestion is probably not a relevant exposure pathway for either chemical or for either animal.

Amphibians

Given that amphibians are a globally endangered taxonomic group, understanding the potential impacts of MPs is a relevant research topic for their conservation. The lifecycles of amphibian (terrestrial/aquatic) and reptile species can result in exposure to micro and nano plastics from numerous sources and inputs (Figure 1.1) at different life stages in both aquatic and terrestrial settings. Due to the complexity of field exposures to multiple xenobiotics and parasitic infections, it is difficult to assess a single cause when assessing amphibian impacts observed in field studies (da Costa Araujo et al., 2021). It is likely that the dramatic loss of species in these classes of organisms is a combination of all these factors. There are very few studies examining the biological impacts of microplastics on amphibians, and even fewer studies on terrestrial reptiles.

The field studies examining environmental quantities of micro and nano plastics have provided limited information of concentrations in these smaller ponds and lakes where these species reside (Hu et al., 2018). It has been reported that small freshwater ponds and waterbodies act as sinks for microplastics and microfibers, and the presence of these contaminants has been confirmed (Hu et al., 2020b). The water column was shown to be the primary source of exposure to larval amphibian life stages. Buss et al. (2022) reported MFs concentrations ranged from 4.50 to 12.66 L^{-1}, with fiber lengths of 0.52 ($\pm$ 0.14) to 1.25 ($\pm$ 0.30) mm.

The Buss et al. 2022 study was well designed and utilized appropriate controls for MF exposures that represented environmentally relevant concentrations. The effects were examining the interaction of polyester MFs on parasitic infection of trematode cercaria on the wood frog (*Rana sylvatical*) following MF exposure. It was hypothesized that the MFs would have a deleterious effect on parasite resistance by affecting the host's response. This study demonstrated that that the polyester MFs did not alter trematode survival or amphibian susceptibility to infection, but did reduce the parasite's infection success under environmentally relevant (10 and 20 mg L^{-1}) fiber concentrations.

Schessl et al. (2019) reported sampling GI contents of 31 different amphibian species, all anurans: *Anaxyrus americanus* (Holbrook, 1836), *Lithobates clamitans* (Latreille, 1801), *Lithobates palustris* (LeConte, 1825), *Lithobates pipiens* (Schreber, 1782), and *Lithobates septentrionalis* (Baird, 1854); no plastic was found in their GI tracts. However, MPs were found in the diet of *Triturus carnifex* Laurenti, 1768 (Iannella et al., 2019) and tadpoles (Hu et al., 2018; Karaoğlu and Gül 2020). Further evaluation of the occurrence of MPs in anurans and other amphibians is suggested, especially in areas where a potential of increased input is suspected.

The African clawed frog *(Xenopus laevis)* is a model organism recognized as a good laboratory model for examining chemical effects on amphibians. De Felice et al. (2018) reported that PS MPs did not affect body growth and swimming activity in *Xenopus laevis* tadpoles from stage 28 to 46. Exposure was to PS 3 µm particles in concentrations of 0.125, 1.25 and 12.5 µg/ml. The particles were present in the digestive track but not in the gills. No statistical effects were observed on development or altered swimming ability. In another set of *Xenopus laevis* studies it was shown that exposure to MFs increased the digestive tract gut length and mass. The fibers present were visually observed lining the gut wall. The non-digestible fiber control was cellulose, which produced similar modification of the digestive tract. For both the microplastic fiber and cellulose control the body mass and condition of the tadpoles were similar across all experimental groups. This study suggested the possibility of compensation by larval forms when encountering low food quality. Different effects may have occurred if there were toxic compounds associated with the fibers or if fibers embedded in the wall of the digestive tract. To emphasize this point, Bacchetta et al. (2021) exposed *Xenopus* to polyester fibers taken from a dryer filter. Modified shapes and sizes in a concentration of 6.3 µg/mL caused gastrointestinal damage with an EC_{50} at 96h. Fibers were observed to press against the digestive epithelium, deforming the normal architecture of the gut, sometimes pushing deep into the epithelium until piercing it or obstructing the lumen. In this study mobility was decreased. Heterogeneity of environmental samples may result in effects not observed with pure MP samples. A recent study demonstrated that high-molecular- weight PVC administered orally to adult frogs (2x weekly for 6 weeks 1% BW) could be transferred to eggs and resulted in malformations, altered viability and decreased gene expression in offspring (Pekmezekmek et al., 2021).

In reviewing the available amphibian studies of MP exposure several commonalities appear: 1) effects can be induced using unrealistic fiber and particle counts that are much higher than those found in the environment; 2) studies that have been conducted in the laboratory have shown limited impact on the different stages with larger organism stages showing less effects; and 3) there does appear to be an impact on developing larval digestive tract morphology and altered gene expression. Because of the very limited number of appropriately controlled and realistic exposure scenarios reported in the literature further studies are warranted. The decline in the number of these amphibian and reptilian species worldwide has been attributed to extensive loss of habitat, and widespread parasitic, fungal, and viral infections along with acidification of lakes and chemical inputs. Based on the current literature it is difficult to assess the direct impact of micro and nano plastics/fibers on the ecological health of these species due to the lack of well-designed experiments and the complex effectors in the ecological settings.

Conclusions

Plastic contamination of the aquatic environment poses a hazard to aquatic organisms. There is exposure through both the water column and sediment sinks coming from multiple sources. The risk of adverse effects is determined by a number of factors concerning the chemical and physical form of the plastic as well as the organisms being exposed. Current literature suggests that plastic particle toxicity depends, in part, on particle composition, size, shape, and concentration (Kögel et al., 2020). At present the degree of risk is uncertain, due to the plethora of unrealistic exposures in the laboratory.

Several studies have demonstrated that plastics in the aquatic environment can pose a significant impact on organisms when the particles are able to be taken up by the organism through the gastrointestinal tract, gills and across epithelia. Epithelial tissues in contact with plastics can exhibit physical damage and irritation that can cause macrophage or inflammatory infiltration and dysplasia of the epithelium. Nanoplastics more readily pass-through biological barriers and accumulate in organs. This is due to both passive and active transport mechanisms innate in the organism for handling particulates based on size. If the plastics are small enough, they can be taken up though phagocytosis, or lymphatics or enter the blood compartment and be deposited in tissues. Alterations have been observed that may interfere with systems involving energy supplies and reserves, intracellular organelles, and cellular responses. If the smallest MPs or nanoplastics affect the smallest organisms (phyto- and zoo- plankton), aquatic food webs could be affected at the community level.

When assessing a specific type of plastic's potential biological reactivity, it is essential that the chemical structure be considered. Although shape, size and other factors are critical, the different plastics will have different reactivity with the receptor organism. The plastics' chemical makeup will also affect several important physical factors such as density, charge, reactivity, decomposition rates and biological reactivity.

Although there are currently a large number of publications dealing with environmental effects of microplastics on living organisms, there are major concerns about the relevance of the majority of the papers to real world scenarios. Cunningham and Sigwart's (2019) meta-analysis points to three issues that need improvement in future research: use of extremely high dosages, incompatible units of measurement, and the lack of good controls. They found that 5% of exposures did not use any control, and 82% used "dramatically elevated" concentrations. They consider 100 MPs L^{-1} to be "high" compared to environmental levels. Only 23 studies tested effects from environmentally realistic concentrations. The studies they considered that used "high concentrations" used MP exposure ranges multiple orders of magnitude higher than environmental concentrations – extreme levels with no environmental relevance.

There are relatively few studies that can capture and/or measure the smallest MPs or nanoparticles. Based on the current toxicity literature it appears that the irregular plastic fragments and fibers appear to elicit the most activity. Manufactured smooth beads seem to be the least concern from a biological perspective.

Recommendations

- Characterization of the amounts and types of MP/MF present in different locations within New Jersey waterways at the atmosphere/surface interface, suspended within the water column and in benthic sediments. To ensure concentration accuracy samples should **not** be taken with nets.

- Examination of accumulation and distribution (toxicokinetics) within a broad array of organisms from zooplankton, macroinvertebrates, and vertebrate organisms that are important in New Jersey ecosystems.

- Examine the toxicodynamic properties of the most prevalent MP/MF on critical life stages.

- Develop analytical tools that can measure the presence and effects of the smallest MPs and nanoplastics in NJ waterways and test organisms.

Question 2. What are the primary sources of these contaminants to rivers and streams specific to New Jersey; which forms of plastic cause the most harm; and which forms are most feasible for prevention, source reduction and/or exposure reduction?

Primary Sources of Microplastic Contaminants to Rivers and Streams

Microplastics (MPs) and nanoplastics enter the environment through multiple physical and chemical processes including abrasion/wear, chemical degradation (e.g., oxidation, loss of plasticity/becoming brittle), photodegradation (UV aging) and combustion. All of these processes generally turn larger plastic pieces into micro ($\leq$ 5 mm to 0.1µm), and then nano (0.001 to 0.1µm) particle sizes. The mechanisms of MP creation are directly related to their point of origin. For example, particles generated through combustion usually start as airborne particles and are carried through the air by wind until they have the opportunity to settle, while those that are shed from clothing might be transported through wastewater, with the primary contributor being washing machines. Microfibers are also shed into the air during textile manufacturing, during wearing clothes and drying them in dryers. From the atmosphere they can enter aquatic systems via wet or dry deposition. Street/city dust and tire dust are most likely transported via storm runoff.

Fate through the *biosphere*

One of the largest sinks for plastic particles, whether initially airborne or waterborne, are surface waters, where they can be ingested by biota within the water system. MPs may bioaccumulate in shell or fin fish and be ingested by humans who include shell or fin fish in their diet. Alternatively, airborne MPs that are inhaled are generally too large to translocate to the circulatory system, although there is evidence that smaller inhaled nanoplastics may reach the circulatory system (blood compartment). While their overall human health effects are still being evaluated, (Vethaak and Legler 2021), MPs have been found in human lungs (Amato-Lourenço et.al., 2021), placenta and meconium (Braun et.al, 2021), providing evidence for *in utero* translocation to the fetus. In addition, Hu et al., 2021 reported that polystyrene microplastics disturb maternal-fetal immune balance and cause reproductivity toxicity in pregnant mice which indicates further concern when MPs translocate to the fetus.

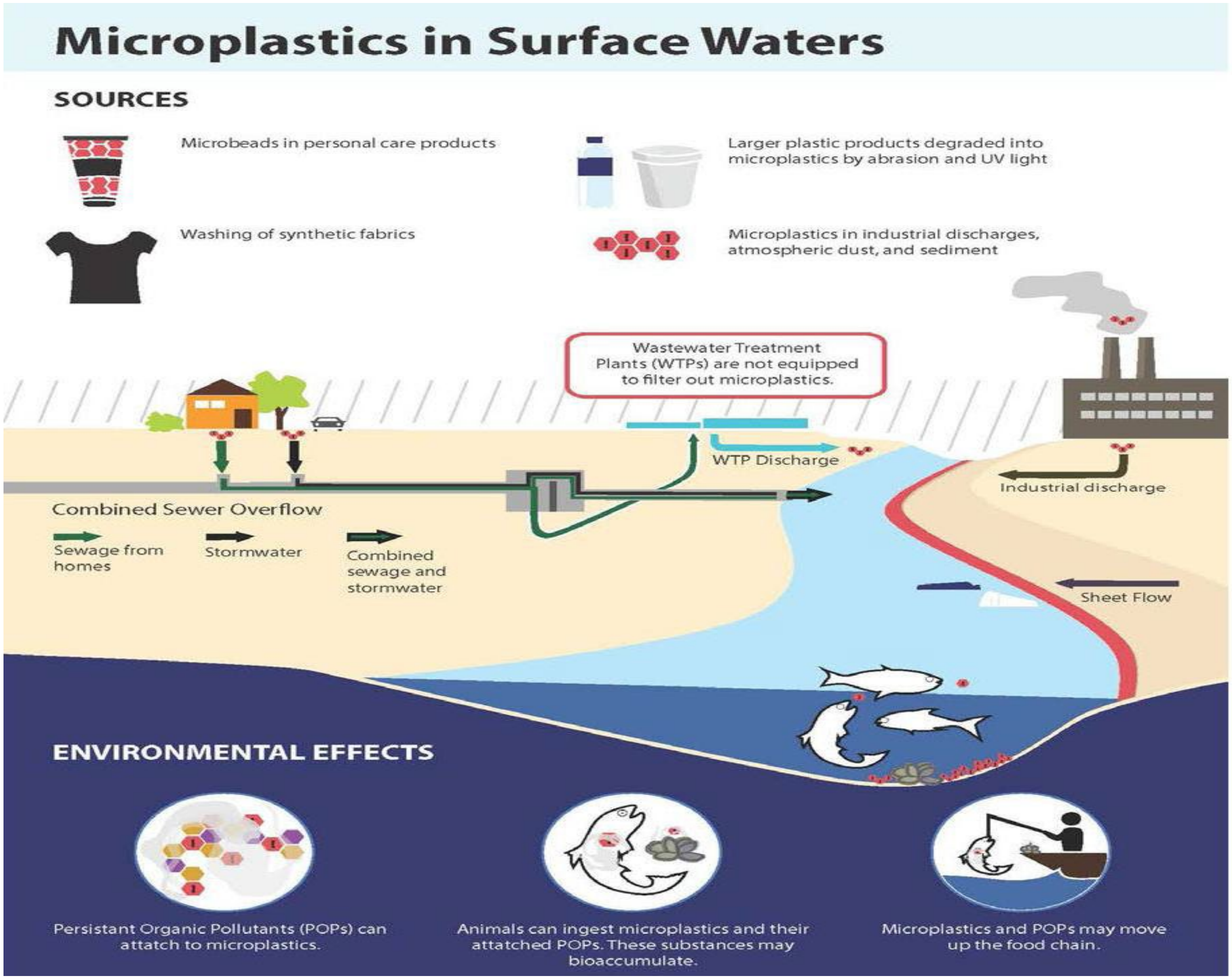

Figure 2.1. Sources, transport, and potential bioaccumulation of microplastic associated persistent organic compounds (POPs) released into the environment. *Adapted from Ravit et al.,2017.*

Microplastics may be large enough to be excluded by many of the membranes (e.g., enterocytes of the gut) but may still become part of the food chain depending on their size and the animal species that may store them (e.g., bivalves). Microplastics that are not ingested within a freshwater system can be either transported to the ocean, settle out in the benthos, or further fragment into nanoplastics. There may also be further biologically driven degradation through microbial intermediaries, as some microbes do attack crosslinking agents as well as specific sites within a plastic polymer. Whether bioremediation of micro and nanoplastics is possible is still being studied.

Sources and Environmental Release Pathways

A management plan for the reduction of MPs into NJ waterways should target both the predominant sources of MPs, as well as focus on opportunities for mitigation within environmental release pathways. When considering potential for mitigation, it is important to understand the pathways of how MPs may enter New Jersey waterways. There may also be some consideration of regional control discussions, especially in shared watersheds and large water bodies. From an environmental point of view, upstream and across state line inputs need to be considered. Although it may be very difficult to identify and control the myriad of potential sources, Figures 2.1 and 2.2 indicate that wastewater treatment plant effluent and sludge are very significant collection points for MPs before entry into the aquatic ecosystem. Hence, treatment plant discharges may be where attention for mitigation strategies should be focused.

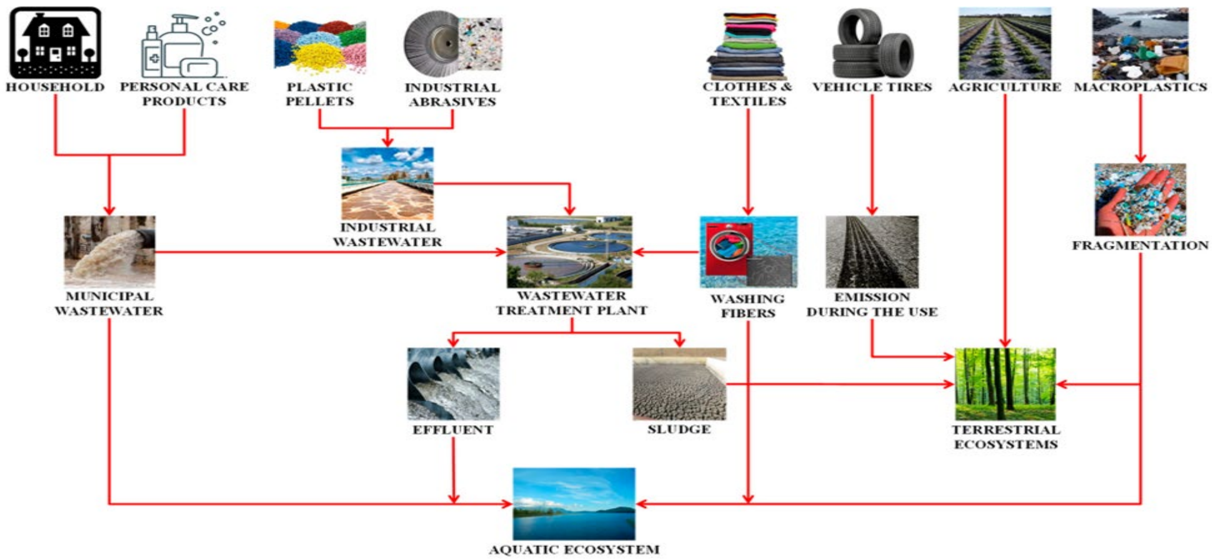

Figure 2.2. Sources of microplastics in the environment. *Adapted from Miloloza et al., 2021.*

<u>Primary Sources</u>

The greatest percent of MPs come from synthetic textiles – 35%, followed by tire wear particles – 28%, and city dust-24%.

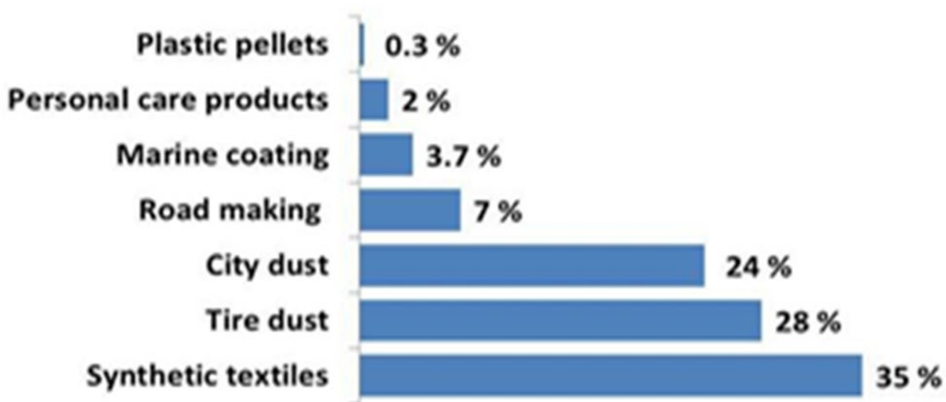

Figure 2.3. International Union for Conservation of Nature (IUCN) primary microplastics in the oceans: A global evaluation of Sources, 2017. *Adapted from Boucher et al., 2017*

Tires and Road Wear Particles

Tires and road wear particles (TRWP) generated on roads during driving processes contribute to airborne non-exhaust emissions of MP pollution. Natural and synthetic rubber account for half of the 1.3 million tons of tire-wear particles generated on Europe's roads (Fig., 2.4). The major amount of TRWP consists of coarser heterogenous particles that are released to road surfaces, soils and aquatic compartments. The extensive compilation of annual emissions of tire wear for numerous countries shows per-capita-masses ranging from 0.2 to 5.5 kg/(capita*area). Contribution of tire wear to PM_{10} (inhalable particles $\leq$ 10µm) accounts for up to approximately 11% of mass. Data on degradation is scarce and most studies do not use realistic materials and conditions. The only published degradation study performed under environmental conditions implies a half-life of tire rubber particles in soils of 16 months (Cadle and Williams, 1980). For truck tires, which mainly contain natural rubber, shorter periods were observed in laboratory tests. Concentrations of tire wear compiled from environmental monitoring studies show highly variable concentrations in road runoff, road dust, roadside soils, river sediments and river water, with a general decrease following the transport paths (Bänsch-Baltruschat et al., 2020). However, the behavior of TRWP in freshwater referring to transport, degradation, and sedimentation is still unclarified. Environmental monitoring of TRWP is still hampered by analytic challenges, and data on environmental concentrations is rare. Further research is needed on emission factors, development of analytical methods for environmental matrices, long-period monitoring, fate in surface waters and soils, (eco)toxicological impacts and degradation under realistic conditions.

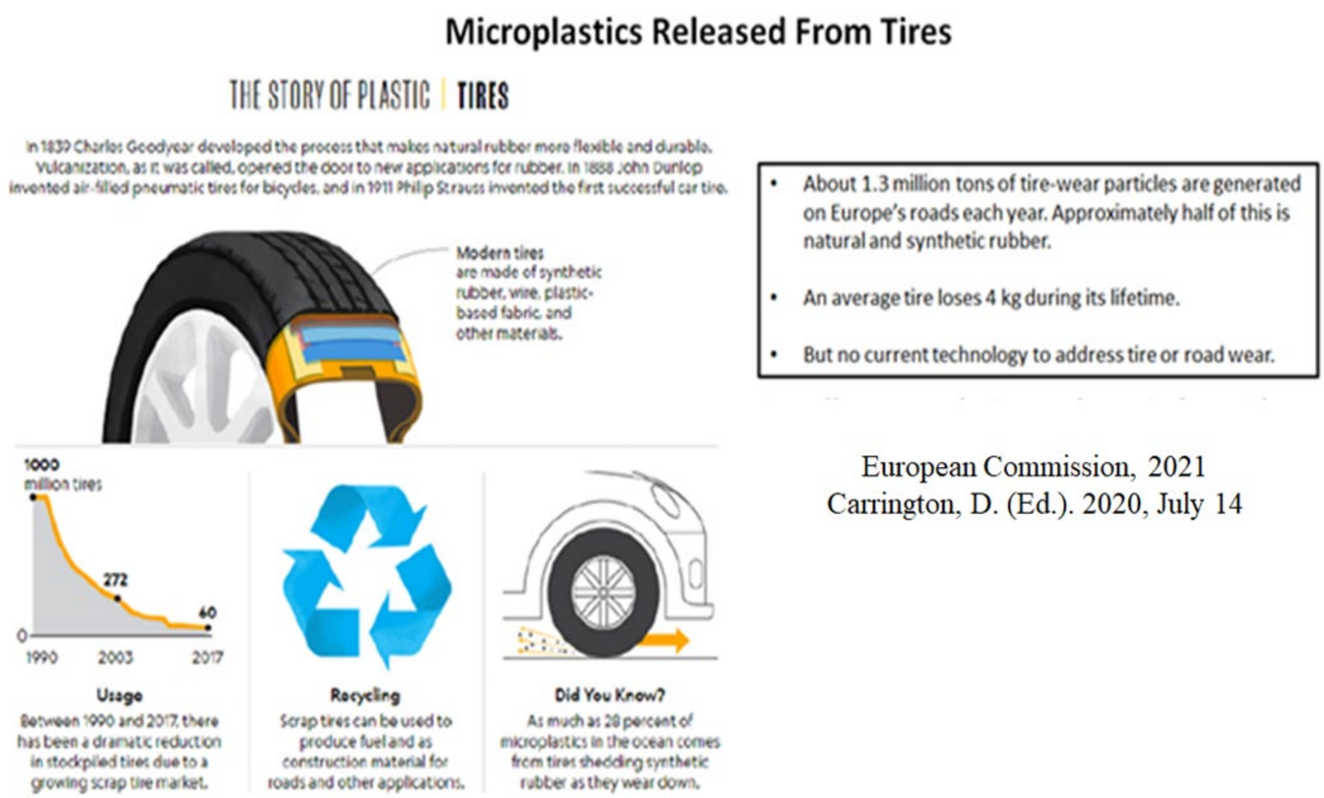

Figure 2.4. European Commission summary of tire wear particles

Plastic Fibers from Textiles

The majority of clothing is made from plastic-based materials like polyester, rayon, nylon, and acrylic. When washed, synthetic clothing sheds tiny plastic fragments known as microfibers (MFs), which are the most prevalent type of microplastic in marine environments (De Falco and Cocca 2022). Plastic fibers from textiles have been indicated as a major source of MPs, entering waterways via wastewater from washing machines and diverse non-point sources. A single garment can produce more than 1,900 fibers per wash. Wastewater treatment plants were not designed to remove MFs, but generally do remove over 90%. Filters for washing machines are a promising prospect for

reducing discharge of MFs, but more research is needed to determine their effectiveness (Dixon et al., 2017).

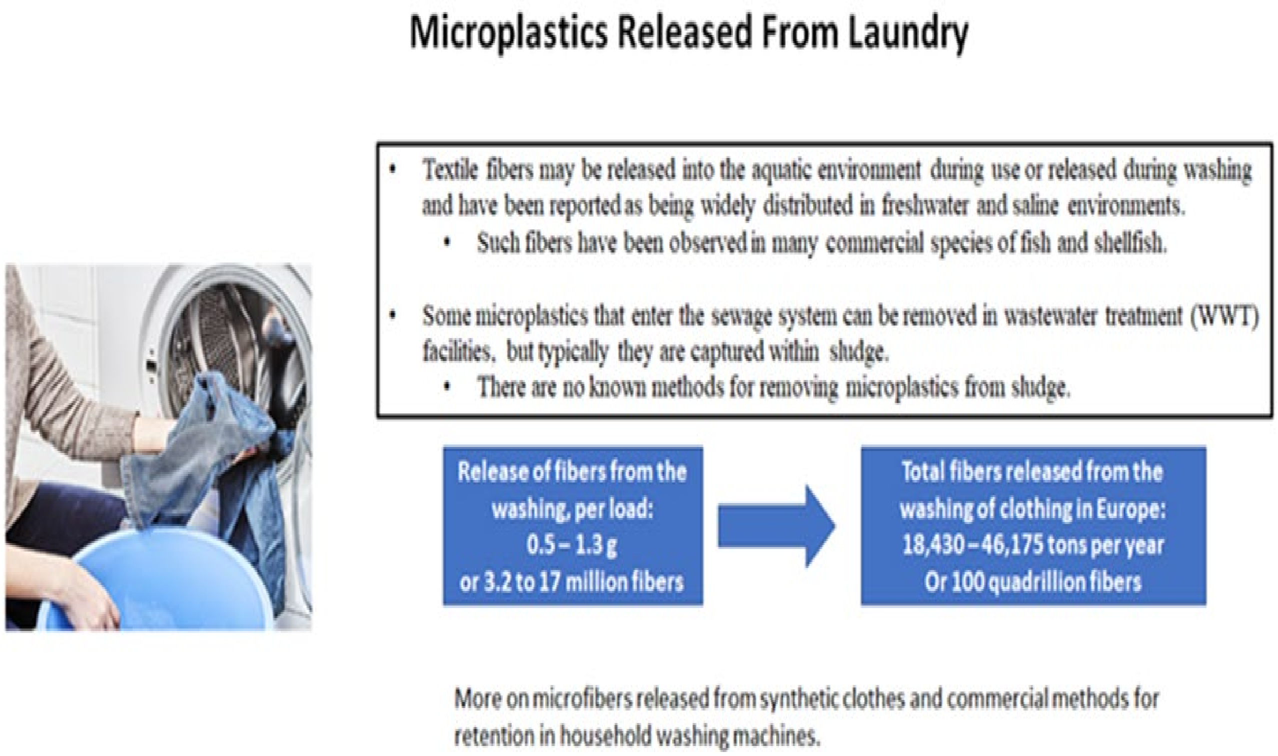

Figure 2.5. Microplastics in laundry. A*dapted from De Falco et al., 2019 and Okamoto 2021.*

De Falco et al. (2019) evaluated the contribution of washing synthetic clothes to MPs in the environment (Figure 2.5). They showed that MFs from washed garments corresponded to 640,000 to 1,500,000 microfibers released per washing. Some textile characteristics, such as the type of fibers constituting the yarns and their twist, influenced the release of MFs during washing. The most abundant fraction of MFs shed was retained by filters with pore size of 60 µm, presenting an average length of 360–660 µm and an average diameter of 12–16 µm, indicating dimensions that could pass through wastewater treatment plants and pose a threat for marine organisms.

Microfiber pollution is a complex problem, there are some simple steps to help washing machine users reduce the number of microfibers that flow from washing machines to waterways (De Falco et al., 2019; Okamoto 2021):

1. **Wash clothing less often.** This is perhaps the simplest and most effective method for reducing microfiber pollution.
2. **Only wash full loads of laundry.** This results in less friction between clothes and reduces shedding of synthetic fibers.

3. **Wash laundry with cold water for a shorter period of time.** A recent study by Northumbria University and Procter & Gamble found that switching to a colder and shorter cycle can dramatically reduce microfiber shedding (Lant et al., 2020).
4. **If possible, use a front-loading washer.** A 2016 study by University of California, Santa Barbara showed that top-load washing machines produced significantly more microfibers than front-loading machines (Hartline al., 2016).
5. **Install an external microfiber filter on your washing machine and/or use a microfiber-catching laundry ball or bag. Dispose of the captured microfibers in the trash.** There are several commercially available external lint filters to choose from. There are also more affordable but less much effective products like wash bags and laundry balls designed to capture microfibers (Pope, 2022).

However, all these suggestions would require educating consumers. Getting the public to adopt these measures would not be likely, so MF filters would need to be a national requirement to be practical.

Clothes dryers can emit masses of MFs directly into the air. Microfiber emissions vary based on dryer type, age, vent installation and lint trap characteristics. Dryer use would be viewed as separate since washing machines release to aquatic systems, whereas dryers release into the air. These releases could contribute to atmospheric deposition of microplastics. Therefore, dryers should be included in discussions when considering strategies, policies and innovations to prevent and mitigate microfiber pollution (Kapp and Miller 2020). A sixth item to the list above could be to dry on a clothesline if it is possible.

MFs can be found in the review by Ramasamy and Subramanian (2021). Their review analyzed the existing literature to identify potential measures to control MF pollution, with a focus on textile properties and laundry filters. The review showed that the use of finer count yarns with filaments and compact structures reduces microfiber shedding. Conversely, mechanical finishes like shearing and raising (as in fleeces) increase MF release because they damage the fabric structure. They also noted a significant MF reduction after chemical (coating) finishing process. In the case of commercial products, the available external laundry filters are reported as much more efficient than the in-drum devices on the market. An analysis of existing regulatory norms showed that very few countries had developed laws addressing environmental MFs, and no global regulations or standards were found to test for MF pollution. In the case of laundry filters, although they filter MFs effectively, they do not prevent them, so can be a control measure, but not a solution to the problem. France has mandated that all new washing machines sold be equipped with a built-in filter by 2025. Controlling textile characteristics appears to be an effective strategy to prevent MF shedding from synthetic textiles. Production methods and textile parameters could yield textiles that shed fewer or no MFs. However, no detailed research was found that correlated these parameters, indicating the potential scope for future research. The recommendations of Ramasamy and Subramanian would require an international effort to change textile manufacturing practices. For the NJDEP, perhaps the best opportunity to reduce MFs would be to identify suitable technology solutions at wastewater treatment plants. However, some states are considering laws requiring filters, similar to that of France.

In a 2021 update to the 2017 EPA report "A Trash Free Waters Report on Priority Microplastics Research Needs" subject matter experts suggested expanding the list of sources to reflect the built environment, including wastewater and stormwater runoff from urban areas. They noted increased interest in identifying where microplastics are introduced into the environment, as opposed to focusing on where they accumulate in waterbodies (From this EPA Workshop, Figure 2.6 below is the updated conceptual model showing microplastic sources, transport, and fate in the United States (US EPA, 2017 and US EPA, 2021).

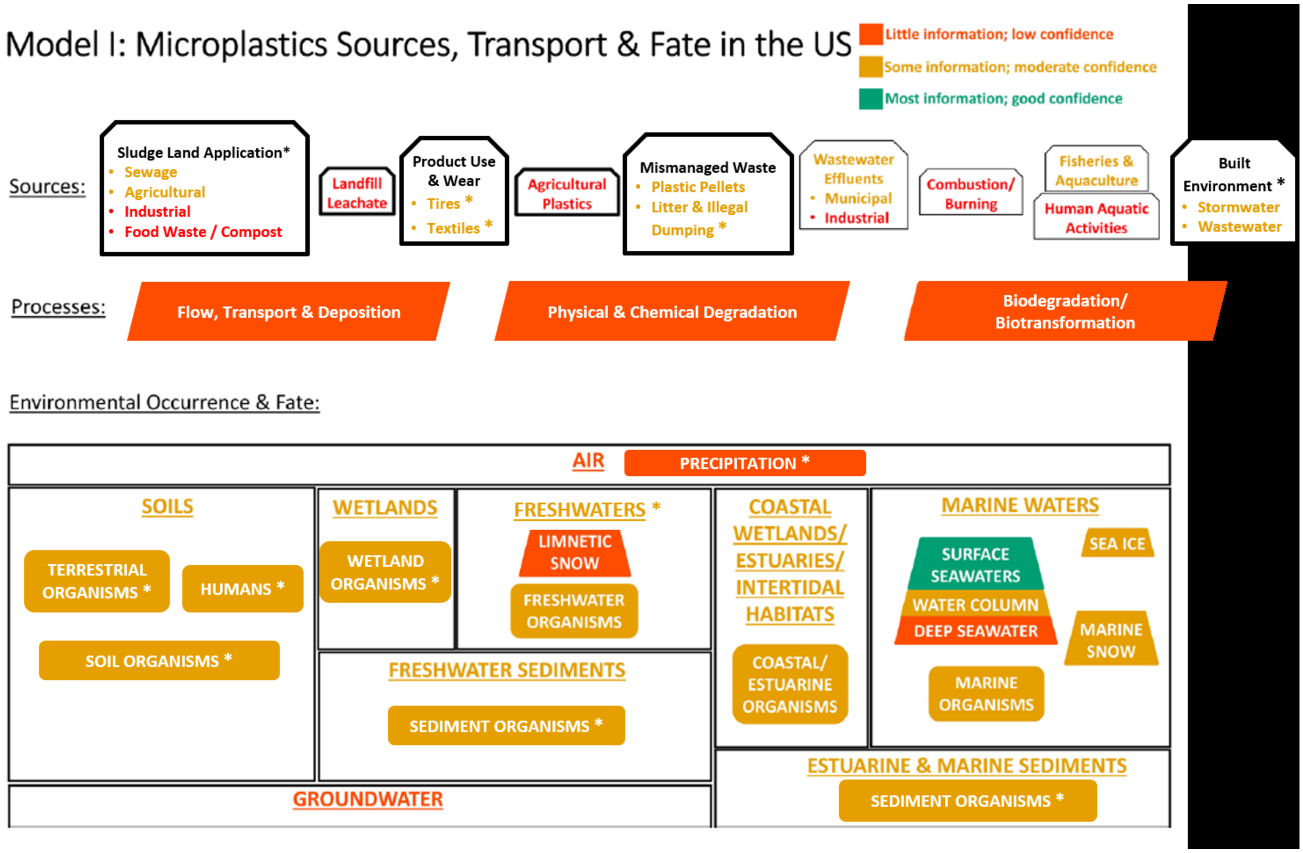

Figure 2.6. Conceptual Model 1: Microplastics Sources Transport and Fate in the US. From USEPA, 2021.

Wastewater Treatment Plants

Wastewater treatment plants (WWTPs) are receptors for the cumulative loading of MPs from collected landfill leachate, industrial and domestic wastewater. Partitioning of MPs through the settlement processes of wastewater treatment results in the majority becoming concentrated in sewage sludge. Vigilant management of MPs in sewage sludge or biosolids is necessary to reduce environmental releases (Mahon et al., 2017).

The majority of wastewater MPs, 72% on average, are removed during preliminary and primary treatments. During typical WWTP primary treatment, large debris items are eliminated with screen meshes sized 6 mm or larger. Low density MPs tend to collect in the grease layer, where they are skimmed off. Secondary and tertiary treatments remove an average of 88% and 94% of MPs, respectively. Secondary treatment removes suspended and dissolved organic material, through the action of microorganisms within large aeration tanks. Flocculation processes in settling tanks separates sewage sludge from the post-processing effluent. This is followed by disinfection processes, polishing or possibly advanced (tertiary) treatment, such as filtration through sand and/or activated carbon columns before the treated water is discharged into a waterbody. Removal rates can reach 96% - 99% for secondary WWTPs and 90% - 99.9% in tertiary WWTPs. Nevertheless, it has been noted that even the tertiary stage cannot completely remove MPs from final effluent (Iyare et al., 2020; Habib et al., 2020).

Although overall removal is high, the residual MPs in treated effluent (~10% of the MPs in influent wastewater) represent a significant release given the high numbers of MFs shed during washing and the large volumes of effluent involved (Lyare et al., 2020), and so WWTPs are a significant point source of MP entry into aquatic and terrestrial systems. The major concern is small particles (especially ≤150 μm) in discharged wastewater effluent. Estahbanati and Fahrenfeld (2016) investigated the influence of WWTP discharges on surface water MP concentrations. They found Raritan River MP concentrations increased downstream of WWTP discharge locations. However, the presence of MPs at all locations sampled upstream and downstream from WWTP discharges indicated that WWTP effluent was not the only Raritan River MP source. Notably, this study collected samples with plankton nets, so they missed many microfibers of smaller size. Microfibers are the major contribution to the MPs being released by wastewater treatment plants (Habib et al., 2020). In a study of eight WWTPs discharging to San Francisco Bay, 80% of MPs found in the effluent of four WWTPs with secondary and four WWTPs with tertiary treatment were found to be fibers (Sutton et al., 2016).

Because MPs mainly partition to WWTP sludge, sludge management modes for NJ's WWTPs were reviewed. Table 2.1 provides a summary by NJ County that shows how sludge is managed. A description of each mode is shown below the table. Incineration, "beneficial" fertilizer or soil amendments in NJ, "beneficial" use out of state, out of state disposal, and/or in state beneficial use land cover are the current options for NJ sludge management. Knowing that the sludge is highly contaminated with microplastics causes one to question whether applying it to soil is really beneficial.

For Table 2.1, Management mode column headings are defined as follows:

INCINERATION: Generator's sewage sludge is incinerated at an on-site in-state incinerator.

CLASS A BENEFICIAL USE: Generator's sewage sludge is processed in New Jersey by a system designed to prepare Exceptional Quality sewage sludge for beneficial use as a fertilizer or other soil amendment. The process must meet specific pathogen reduction, vector attraction reduction and pollutant standards.

CLASS B BENEFICIAL USE: Generator's sewage sludge is processed in New Jersey by a system designed to prepare Non-Exceptional Quality sewage sludge for beneficial use as a fertilizer or other soil amendment. The process must meet specific but less intense pathogen reduction, vector attraction reduction and pollutant standards which, when combined with site restrictions and management practices, are equally protective of human health and the environment.

OUT-OF-STATE BENEFICIAL USE: Generator's sewage sludge is managed in a beneficial use system located in another state. Generator is required to demonstrate compliance with regulations applicable in the receiving jurisdiction.

OUT-OF-STATE DISPOSAL: Generator's sewage sludge is managed in another state. Generator is required to demonstrate compliance with regulations applicable in the receiving jurisdiction.

I**N-STATE BENEFICIAL USE LANDFILL COVER:** Generator's sewage sludge is beneficially used by a New Jersey landfill specifically permitted by the Department to utilize the material as a landfill cover component.

OTHER DTW: Generator transports sludge to another larger receiving domestic treatment works where the sludge is processed by the receiving domestic treatment works and becomes a part of the quantity of sludge reported by the receiving domestic treatment works.

Knowing about the environmental effects of MPs, it would appear that the various types of "beneficial" uses are not beneficial after all, since the MPs in the sludge are very highly concentrated and likely to accumulate in and damage soil organisms and terrestrial ecosystems.

Table 2.1. 2019 sewage sludge production by management mode in NJ counties. DMT = Dry Metric Tonnes
Adapted from NJDEP, Bureau of GIS 2019. https://www.nj.gov/dep/dwq/pdf/sludgeproductiondata2019.pdf

DOMESTIC SQAR REPORT (Calendar Year 2019)

SUMMARY OF EXISTING SLUDGE PRODUCTION BY MANAGEMENT MODES (DMT/YR)
(For the Calendar Year 2019)

COUNTY	INCINERATION	CLASS A BENEFICIAL USE	CLASS B BENEFICIAL USE	OUT-OF-STATE BENEFICIAL USE	OUT-OF-STATE DISPOSAL	IN-STATE BENEFICIAL USE LANDFILL COVER	OTHER DTW	COUNTY TOTAL
Atlantic	11148.2	255.1	0.0	0.0	0.0	0.0	306.5	11709.8
Bergen	4240.4	0.0	0.0	0.0	0.0	0.0	15623.9	19864.3
Burlington	0.0	7034.6	347.0	0.0	276.7	0.0	1065.4	8723.7
Camden	0.0	813.7	0.0	8983.9	1809.9	0.0	428.0	12035.5
Cape May	0.0	72.9	0.0	347.3	0.0	0.0	3311.7	3731.9
Cumberland	0.0	89.3	843.6	0.0	507.7	0.0	641.0	2081.6
Essex	0.0	0.0	0.0	23549.0	0.0	14517.2	1923.1	39989.3
Gloucester	0.0	84.6	0.0	899.7	682.5	1982.8	10.5	3660.1
Hudson	0.0	0.0	0.0	0.0	0.0	0.0	5541.5	5541.5
Hunterdon	0.0	0.0	0.0	0.0	47.6	0.0	1478.3	1525.9
Mercer	5760.0	0.0	0.0	1305.4	187.3	0.0	2817.3	10070.0
Middlesex	0.0	0.0	0.0	3950.4	0.0	33895.6	400.8	38246.8
Monmouth	2593.1	141.9	0.0	258.2	16.3	0.0	6920.2	9929.7
Morris	0.0	0.0	0.0	0.0	3196.8	0.0	8102.5	11299.3
Ocean	0.0	8397.1	0.0	0.0	2.7	0.0	98.6	8498.4
Passaic	0.0	0.0	0.0	0.0	0.0	0.0	2278.8	2278.8
Salem	0.0	0.0	186.7	0.0	346.6	0.0	75.1	608.4
Somerset	3662.6	0.0	0.0	0.0	373.2	0.0	1155.5	5191.3
Sussex	0.0	0.0	0.0	0.0	0.0	0.0	1252.9	1252.9
Union	0.0	599.4	0.0	3830.6	4426.9	0.0	135.1	8992.0
Warren	0.0	0.0	0.0	0.0	0.0	0.0	2054.1	2054.1

Incineration

Sewage sludge incineration can reduce sludge volume by combustion, and theoretically can result in the conversion of hazardous organic material to inorganic end products. Incineration may also change the plastic's chemical structure, decompose polymers and add or remove chemical structures, such as halogens or oxygen depending on the temperature and mixture being incinerated. The extent of sludge reduction can range to as high as 90% of the input sewage sludge (to a sterile ash) through combustion (dependent on the mineral content of the sewage sludge). Sewage sludge incinerators do not require significant land for sludge disposal, operate in all seasons, safely manage almost 25% of the State's sewage sludge production without nuisance, and are fully regulated by the NJ DEP's Air Pollution Control Program (NJ DEP 2006). Unfortunately, combustion conditions in incinerators may not be ideal, mainly due to non-uniform temperature and mixing conditions inside the furnace generally causing inefficiencies. Cooling of hot gases in the combustion zone or locally reducing conditions can induce partial destruction and pyrolysis reactions. These inefficiencies can lead to formation of toxic organic compounds, referred to as products of incomplete combustion (Braguglia et al., 2003). Incineration of plastic can release a variety of toxic chemicals into the atmosphere.

Vuori and Ollikainen (2002) did a cost-effectiveness analysis looking at three wastewater treatments (activated sludge, rapid sand filtering, and membrane bioreactor) and two sludge management technologies (anaerobic digestion and incineration) in terms of their MP removal capacity for aquatic and terrestrial ecosystems. They found that conventional activated sludge and digestion leads to high MP release to the aquatic environment and extremely high release to the terrestrial environment. If conventional activated sludge, rapid sand filtering, or membrane bioreactors employ incineration of sludge then the terrestrial MP release goes to zero. If incineration is adopted, then membrane bioreactors release the fewest particles into the aquatic environment, whereas conventional activated sludge treatment would release the most particles.

If sludge is applied to soil, it is a major source of MPs in soil ecosystems, so it is not a good disposal option. If applied to agricultural fields, MPs can potentially translocate into the food system via plant uptake. It is recommended that sludges should not be applied to agricultural soils in NJ or at least monitored for microplastics until decisions are made on microplastic regulation.

Stormwater

Stormwater runoff has been suggested as a significant pathway of MP entry into aquatic habitats; this is likely the major pathway for MPs from tire wear. Studies in San Francisco Bay identified urban runoff as a major pathway for MPs, with average concentration and overall load of MPs in stormwater approximately two orders of magnitude higher than in treated wastewater effluent. Trash capture, such as full-capture devices, can be used to interfere and remove debris from stormwater. This strategy can help prevent the flow of large plastic pollution into receiving waters and help mitigate MP pollution from large plastic debris that can fragment into MPs. Various Low Impact Development (LID) practices, such as bioretention rain gardens, high-flow bioretention systems, and infiltration trenches, offer opportunities to capture both large plastic debris and MPs, while providing additional pollution reduction and groundwater augmentation benefits (California Ocean Protection Council, 2022).

Combined Sewer Overflows As a major Source of Microplastics

Another significant source of microplastics in NJ waterways is Combined Sewer Overflows (CSOs) where Combined Sewer Systems (CSS) are present. Original wastewater infrastructure in urban centers was built as a CSS, that collects and conveys rainwater runoff, domestic sewage, and industrial wastewater through one system. Under normal conditions, this infrastructure transports all the collected wastewater to a sewage treatment plant prior to discharge into a receiving water body. However, when the volume of wastewater exceeds the capacity of the CSS or treatment plant (e.g., during heavy rainfall events or snowmelt), this mix of untreated stormwater and wastewater discharges directly to water bodies. These CSOs contain untreated or partially treated human and industrial waste, toxic materials, and debris mixed with stormwater. They are a priority water pollution concern for the nearly 700 municipalities across the U.S. that have CSSs (NJ DEP, Bureau of GIS, 2022).

In NJ, CSOs are mostly found in the older municipalities of Bergen, Hudson, Essex, Passaic, and Middlesex counties. In Southern NJ, CSO's are limited to Camden and Mercer counties (NJ DEP, Bureau of GIS, 2022). Please see Figure 2.7 and Appendix II: Number of Combined Sewer Outflows from NJ cities. for more details on location of CSOs in NJ.

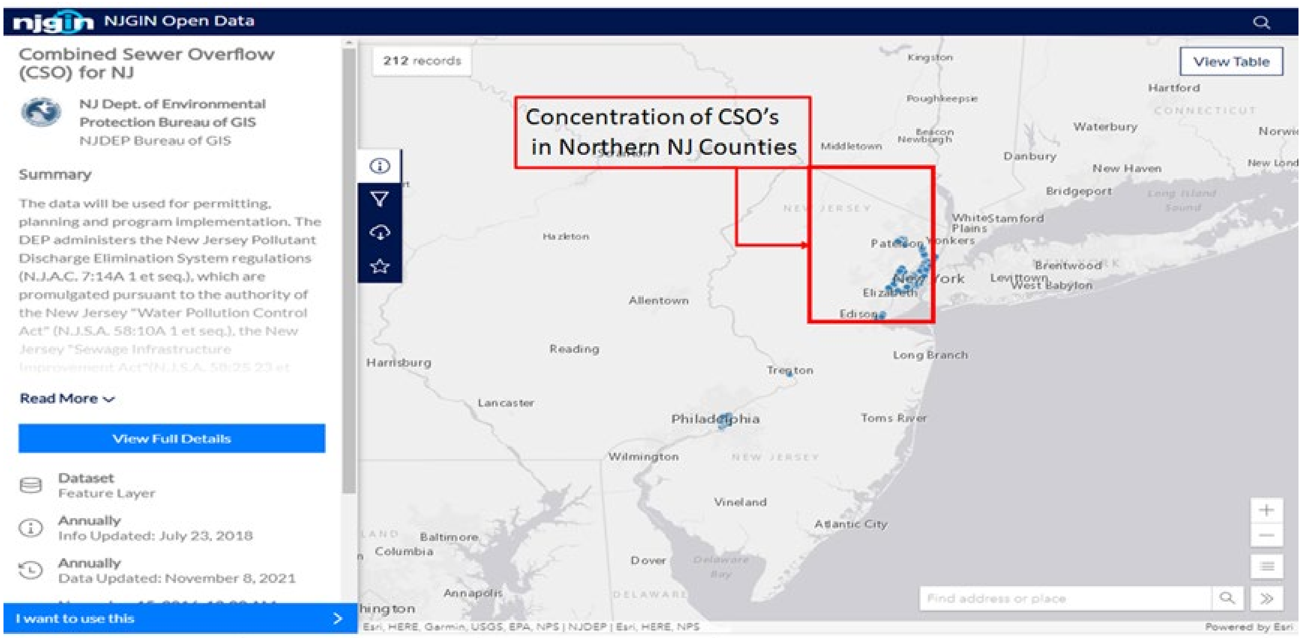

Figure 2.7. Combined Sewer Overflows (CSOs) in NJ. Adapted from NJ DEP, Bureau of GIS, 2022. https://njogis-newjersey.opendata.arcgis.com/

Since the issuance of the first NJPDES permits to regulate CSOs, the total number of CSO outfalls in New Jersey has been reduced from 281 to 210; a reduction of more than 20%. The goal of the CSO permits is to meet the requirements of the Clean Water Act and the National CSO Policy by reducing or eliminating the remaining CSO outfalls in New Jersey. To achieve the reduction or elimination of outfalls, CSO permittees will need to reduce flooding, ensure proper operation, maintenance, and management of existing infrastructure, and provide opportunities for green infrastructure. A major emphasis of the permit process is the development of regional strategies to reduce the amount of storm water that flows into combined sewer systems, through the development and implementation of a Long-Term Control Plan (NJ DEP DWQ, 2022).

The recommendation would be to continue the focus on reducing the number of CSOs across NJ. The solids and floatables controls installed at CSO outfalls prevent solids greater than one half inch from entering NJ waterways. Although CSO solids and floatables controls can eliminate large plastic articles that eventually breakdown to MPs, considering the size of MPs, the controls do not prevent CSOs from discharging MP particles into NJ waterways. Until CSOs are eliminated they will continue to be a source of MPs in NJ waterways, especially in northern NJ counties where CSOs are concentrated.

Atmospheric Transport and Deposition

Allen et.al (2019) observed atmospheric microplastic deposition in a remote, pristine mountain catchment in the French Pyrenees. Their research illustrated the presence of MPs in a non-urban atmospheric fallout and suggested a potential link between precipitation (rain and snow), wind speed, wind direction and the MPs' deposition.

Studies have shown that MPs are ubiquitous in the atmosphere (Brahney et al., 2020). Synthetic textiles are the main source of air-borne MPs. Additionally, the degradation and fragmentation of plastic products, waste incineration, recycling centers, industrial and traffic emissions, and dust re-suspension also increase the concentration of airborne MPs. Wind can transport MPs through the atmosphere over a long distance, and thus MPs are present ubiquitously worldwide, even in remote mountain catchments and polar regions. Therefore, airborne MPs represent a source of contamination in terrestrial and aquatic ecosystems. Human activities and weather conditions, such as rainfall, snowfall and wind, have been confirmed to affect the abundance and deposition of airborne microplastics (Chen et al., 2020). Dris et al. (2016) estimate that between 3-10 tons of MPs are deposited via atmospheric deposition every year in the ~2500 km^2 area of Paris.

Which Forms of Plastic Cause the Most Harm?

Microplastics present in environmental matrices may create risk through either leaching of the plastic components (e.g., crosslinkers or monomers/dimers) or as concentrators and transporters of contaminants within the water supply. For example, in an urban watershed 6PPD-Quinone and 1,3-Diphenylguanidine were present in tire leachate found in road runoff. These compounds are common tire manufacturing additives that are extremely toxic to coho salmon (Johannessen et. al., 2022). Studies have demonstrated that both micro and nanoplastics can concentrate and transport environmental contaminants through multiple media, including surface and fresh water sources (Ravit et al., 2019) as well as through the biota (Katsumiti et.al., 2021).

Aside from their ability to act as contaminant sources, MPs can also act as irritants in the intestinal tract when ingested (DeLoid et al,. 2021; DeLoid et al., 2018; Liu and Kong, 2019). They have also been associated with genotoxic endpoints (Mortezaee et. al., 2019) through redox interactions but Micro Nano Plastics have not yet been implicated. In the lung microparticles are a foreign body likely to cause irritation and perhaps inflammation because they may have a very reactive surface. To date most of the negative health endpoints have been attributed to nano, rather than micro plastics, but a number of possible diseases or disease precursors have been associated with biological exposures in human/human surrogate studies (Campanale et al,. 2020; Prata et.al., 2020).

While a continuum of transport mechanisms exists for micro and nanoplastics that depend on size and chemical/physical characteristics, the smaller MNPs have been observed to translocate across biological membranes by yet to be determined transport processes (Wang et al., 2022). This is significant because translocation would be the first step in circumventing the protective membranes designed to filter contaminants in a biological system (e.g., the epithelial layers of lungs or placenta). If particles can translocate to a protected compartment (e.g., blood) they can create possible negative health outcomes related to leaching of plastic-associated contaminants or the physical presence of foreign bodies, especially those that might act as contaminant transporters, in vulnerable regions (e.g., blood brain barrier). MNPs (primarily nano materials) also have the potential to make the tight junctions within the intestinal tract leakier and potentially more porous, facilitating contaminant uptake (DeLoid et al., 2021; Zhang et al., 2021).

Plastics That Can Be Recycled

Question 2 does ask the SAB to explore what forms of plastics are most feasible for source reduction. To answer this question, we must first discuss what plastics can and cannot be recycled. There is a public misperception with many that all plastics can be recycled. This is an education opportunity with the public. Unfortunately, most plastics are not recycled, and they end up either in landfills, terrestrial or aquatic environments. For the plastics that can be recycled, PET and HDPE, incentives are needed to increase consumer participation in recycling. Increased recycling would reduce MPs in the environment. However, educating the public about what resin types are recyclable is needed because many plastics that end up in recycling facilities cannot be recycled and these facilities become a major source for MPs in the environment. The best opportunities for recycling would be PET and HDPE plastics. However, recycling processes create and release microplastics.

Resin Code 1 (PET): The most commonly used plastics in consumer products (Figure 2.8). It is intended for single use applications; repeated use increases the risk of leaching and bacterial growth. PET is difficult to decontaminate. PET is recyclable; about 25% of PET bottles in US are recycled.

Resin Code 2 (HDPE): The most commonly recycled plastic, considered one of the safest forms of plastic. It is a relatively simple and cost-effective process to recycle HDPE for secondary use. Only 30-35% of HDPE plastic used in the U.S. gets recycled annually.

Resin Codes 3 – 7: These materials are either not recyclable or are not commonly recycled due to limitations of recycling facilities, or there is no consistent demand in the market, or the plastic has resin combinations making it impossible to correctly sort, so they do not get recycled.

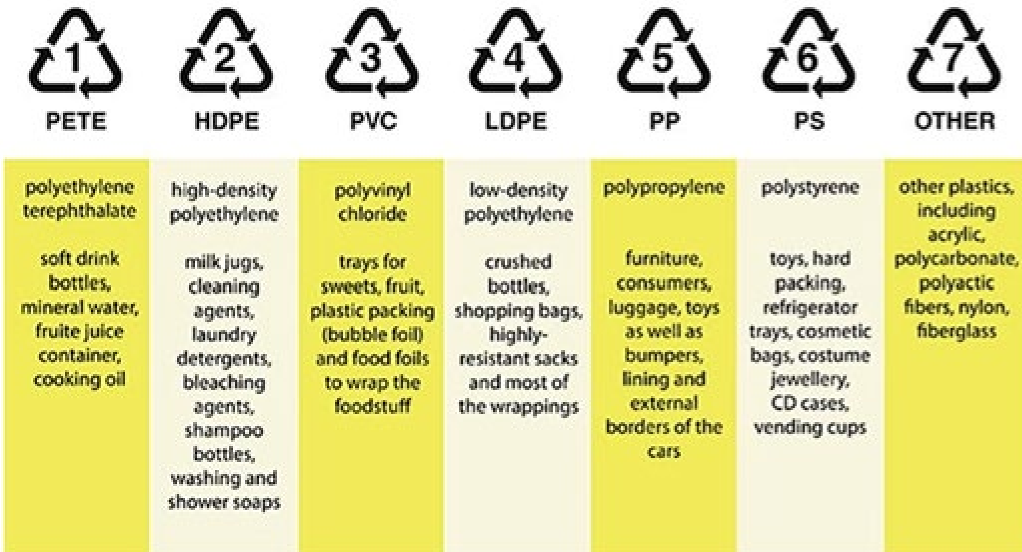

Figure 2.8. Plastic Resin Codes. *Seaman, G. 2022. Plastics by the Numbers | Eartheasy Guides & Articles.*

Plastic Pellets and Personal Care Products

On a percentage basis, plastic pellets and personal care products are estimated to contribute less than 3% of MPs in the environment, but considering sheer volume, it makes sense to look for opportunities to reduce/eliminate. Federal and State regulations have banned MPs from the majority of personal care products. So, although the argument can be made that MPs coming from plastic pellets and personal care products may appear as minimal based on being 3% of the MPs found in the environment, it is of utmost importance to realize that 3% of the total volume of microplastics in the environment is still adversely impactful and these sources should be reduced.

Conclusions

For New Jersey, one of the most significant contribution of MPs to receiving waters of the State, both inland and offshore, are from WWTPs. This is based on the fact that the influent waters come from residential and commercial users that concentrate the micro and nano plastics from a variety of sources (clothing, tire wear, cigarettes, pharmaceuticals, chemical products). In some cases, industrial facilities involved in manufacturing that uses plastics may be MP point sources. Distribution of MPs into the water column or sludge depends on the plastic's physical/chemical properties. Smaller buoyant plastic particles released into receiving waters can be filtered out of the water column or ingested by ecological receptors. Particles may acquire a biofilm and eventually sink, ending up in the sediments and consumed by benthic animals.

Due to the densely populated portions of the State and the extensive use of major interstate roadways, tire particles and road runoff are thought to contribute considerable non-point source MP contributions to receiving waters. Mitigation measures for tire wear particles are greatly needed. Within certain urban areas of the state CSO discharges also contribute to MP mobilization and entry into receiving waters.

Atmospheric deposition may contribute to widespread or local dispersal of plastic particulates and associated contaminants. Comingling of waste material may contribute to some unusual chemical products due to incomplete incineration.

Mechanical and photochemical breakdown of larger plastic products into micro and nano particulates, although the most visible, are likely less of a pollution contributor than other sources. Since micro and nano plastics are not visible to most New Jersey residents involved in recreational activities there may be less interest in pursuing ecological and human health effects.

Recommendations

- Recommend studies to better identify and characterize the various MP source contributions to receiving waters in the state in terms of microplastic concentrations and types.

- Recommend attention to microplastics entry into the environment via monitoring should be given to all potential routes of contamination, including WWTP effluent, land application of wastewater sludge, surface runoff, incineration emissions, and atmospheric deposition. This would also require evaluation of the analytical methodologies used for detection and monitoring of MPs in the environment. This would include standardization of sampling techniques and reporting methods.

- Considering that CSOs are a significant contributor to MPs getting into waterways in northern NJ counties, what would be the feasibility / costs of eliminating CSOs and diverting stormwater to wastewater treatment plants as a means to reduce MPs released directly to waterways in northern NJ?

- Investigate methods of efficiently recycling all plastic types and continue reduction methods. What is the feasibility of developing a circular economy within NJ that would enable more efficient reuse of plastic materials?

- Considering that wastewater treatment plant effluents are among the most significant pathway for MPs getting into NJ waterways, it recommended that NJ DEP considers studies that examine options for more efficient removal of MPs from sewage effluent and biosolids.

- Plants/crops as a problem source: Regarding land application of sludge on agricultural fields, studies are recommended to examine crop uptake of microplastics. Notably, soil types, crop types, physical chemical properties of various types of MPs, size of MPs (e.g., µm vs. nm), shapes of MPs, etc., are examples of variables that need to be considered.

- Plants as a beneficial use: Studies looking at plant uptake (i.e., phytoextraction, phyto-stabilization, phyto-filtration) that may be helpful with stormwater management for storm water that exits directly to waterways.

- Incineration: Recommend studies looking at removal efficiency of MPs from incinerators within the state of NJ that are used for incinerating wastewater sludges. Perhaps, also consider monitoring of incineration emissions for incinerators that burn wastewater sludge in NJ.

➢ Advance programs that educate the public about plastic pollution with a goal of waste reduction. For example, DEP initiate public outreach program about front vs top loading washing machines, cool vs hot setting, less detergent, full loads, availability of filters to add onto washing machines to capture most of the microfibers, etc.

Question 3: What technical steps should the DEP take to understand and manage microplastics and nanoplastics?

At this time, there are a lot of unknowns associated with understanding the fate, transport, uptake, biotransformation and ultimate remediation of microplastics and nanoplastics in the environment. The State has recently banned single use plastic bags and other single-use plastics, which is a first step toward increasing public awareness of the harm of plastics in the coastal ecosystem. However, plastic bags are not a major source of MPs or nanoplastics; also, a feasible alternate solution with thorough lifecycle analysis has not been provided to the citizens. There is significant lack of clarity on certain basic information regarding micro- and nanoplastics, such as how plentiful they are and where, what is their true health and ecological risk in the various environmental systems, and most importantly, how to best test water for MPs, for which there is no existing standard method. (https://calmatters.org/environment/2021/03/california-microplastics-drinking-water/).

However, California has issued the USA's first guidelines for MPs in drinking water and in coastal and marine environments. In 2018, the governor of California signed into law two microplastics bills: Senate Bill (S.B.) 1263 and S.B. 1422. Pursuant to S.B. 1263, in November 2021, California's State Water Resources Control Board circulated a draft Microplastics in Drinking Water Policy Handbook to address concerns over the presence of microplastics in drinking water (https://www.waterboards.ca.gov/drinking_water/certlic/drinkingwater/microplastics.html), and pursuant to S.B. 1263, in February 2022, California's Ocean Protection Council (CAOPC) issued a Statewide Microplastics Strategy in response to concerns over MPs' potential impact on the coastal and marine environment (https://www.opc.ca.gov/webmaster/_media_library/2021/12/Statewide-Microplastics-Strategy_Public-Draft_12.21.2021.pdf).

In the opinion of this group, the strategy adopted by CAOPC – a two track approach – to manage MP pollution in California or a similar approach should be adopted by the DEP and applied to New Jersey conditions for protection of both human and ecological health (https://www.natlawreview.com/article/california-and-world-move-toward-cleaning-microplastics-what-you-need-to-know-now).

The first track outlines immediate, "no regrets" actions and multi-benefit solutions to reduce and manage MP pollution, and includes:

- *Pollution Prevention*: Eliminate plastic waste at the source (products or materials from which MPs originate).
- *Pathway Interventions*: Intervene within specific pathways (stormwater runoff, wastewater, aerial deposition) that transport MPs into California waters.

- *Outreach and Education*: Engage and inform the public and industries of MP sources, impacts, and solutions.

The second track outlines a comprehensive research strategy to enhance the scientific foundation for microplastic monitoring, source identification, risk assessment, and development of management solutions, and includes:

- *Monitoring*: Standardize a statewide monitoring approach. Understand and identify trends of MP pollution statewide.
- *Risk Thresholds and Assessment*: Improve understanding of impacts to aquatic life and human health.
- *Sources and Pathways Prioritization*: Identify and prioritize future management solutions based on local data.
- *Evaluating New Solutions*: Develop and implement future pollution prevention and pathway intervention solutions.

Although we endorse adopting similar strategies to address the issue of micro- and nanoplastics pollution in the State of New Jersey, providing any further details is beyond the scope of this preliminary analysis. As an extension of the recently appointed Governor's Plastic Advisory Council, we strongly recommend that a Microplastics Task Force (MPTF) consisting of various key stakeholders, such as academic and regulatory professionals as well as those from the industry and the community be assembled by the DEP, codified by the New Jersey State Legislature as soon as possible, and tasked with developing a strategy handbook appropriate for New Jersey conditions. Funds need to be identified and allocated for the MPTF to promote appropriate and adequate monitoring and research initiatives that would generate the information that is essential for developing a practical strategy document that can be properly implemented by the DEP. Monitoring and research are the needs of the day for the State to understand and manage microplastics and nanoplastics.

Question 4. To what extent does this issue currently affect New Jersey's water quality/designated use support, tourism, economy, and Environmental Justice communities? The Department is interested in any research findings and papers that could support the Department in developing policy related to the above.

Marine plastics can produce economic costs through reducing the profitability or viability of economic activities -e.g., through the reduction in harvestable marine resources, negative effects on aquaculture or agriculture, or reduced marine ecotourism due to the aesthetic and ecological impacts of plastic pollution. However, since microplastics are largely invisible, aesthetics would not be relevant. To affect tourism or the economy, the public would have to believe that MP exposure in coastal areas is a serious hazard to their health, and therefore avoid going to NJ coastal areas. Since there is no evidence that MP exposure in coastal areas is indeed a health hazard such a public response is unlikely. In a publication by Dowarah and Devipriya (2019), the prevalence of microplastics in the sediments of six beaches in India was reported and its correlation to fishing and tourist/recreational activities was analyzed. They found a strong positive correlation between fishing activity and MP abundance and a weak correlation between recreational activities and MP abundance. The microplastic abundance is considered an effect, rather than a cause of the fishing and recreational activities. Tourism has been considered a source of plastic pollution in the U.S. (Kleinschmidt and Janosik 2021), Europe (Grelaud and Siveri, 2020), and China (Wei et al., 2022).

The Clean Water Act Section 101(a)(2) goals for the protection and propagation of fish, shellfish, and recreation in and on the water should be considered for impacts, as well as the use as a potable water supply after conventional treatment, per the New Jersey Surface Water Quality Standards at N.J.A.C. 7:9B-1.12. Any studies related to the impact of microplastics on the fishing or shellfishing industry would be valuable in assessing impacts to New Jersey's economy.

To affect fish and shellfish resources, there would need to be evidence that current levels of MPs in fish and shellfish are reducing their growth, health, and/or reproduction. The fishing or shellfishing industry could be negatively impacted if the presence of MPs in the fish or shellfish posed a food safety issue causing people to avoid eating them. There is no such evidence available.

Gammaro et al. (2020) reviewed and evaluated information on microplastics in fish and shellfish as a possible threat to their growth and to seafood safety. Bivalves are probably the main source of MPs when humans consume seafood since the whole animal, including the digestive tract which contains the most MPs, is eaten. Current assessments also suggest that the contribution of hazardous chemicals from MPs to human consumers of bivalves is very small compared to other sources, and there is no indication that the safety of such food is compromised. There have been some publications quantifying MPs in both farmed and wild bivalves. Van Cauwenberghe and Janssen (2014) determined the amount of microplastics (>5 µm) in farmed blue mussels (*Mytilus edulis*) from Germany and cultured oysters (*Crassostrea gigas*) from France. They found that the average content was 0.36 and 0.47 particles per gram wet weight soft tissue, respectively. Based on these figures, they estimated that consumers of bivalves in the European Union might ingest 11,000 MP particles per capita annually. There is concern about the toxicity of additives (e.g., phthalates) or environmental chemicals (e.g., PCBs) adsorbed on the MPs. Many experimental studies have been done on the effects of MPs on aquatic organisms, including bivalves, and negative effects have been observed. However, the concentrations of microplastics used in most of these investigations have been orders of magnitude higher than levels found in the marine or freshwater environment. Rochman et al. (2013b) investigated the effects of LDPE microplastics (<0.5 mm) contaminated with environmental levels of POPs fed to Japanese medaka (*Oryzias latipes*) for 2 months. They found that PAHs, PCBs, and PBDEs increased in all groups of fish (control feed, virgin plastic feed, and marine contaminated plastic feed), but only the PBDE concentration was significantly higher in the fish fed the contaminated microplastic.

Microplastics might potentially harm humans by both physical and chemical means. Possible impacts are inflammatory response, disruption of the gut microbiome, tissue damage by the particles, and transfer of adsorbed environmental chemical pollutants. Some preliminary calculations have been made on how MPs in bivalves might contribute to human exposure to contaminants. The general conclusions are that the effects are very low (EFSA, 2016; Lusher et al., 2017). Van Cauwenberghe et al. (2015b) investigated MPs in *M. edulis* from the French, Belgian, and Dutch coastline and found an average concentration of 0.2 particles per gram of tissue. Using this amount and data from the literature on concentrations of PCBs adsorbed to plastic particles collected from the environment and tolerable daily intake (TDI) of PCBs, they calculated that a meal of mussels (300 g of meat) would contribute only 0.06% of the TDI of PCBs. Li et al. (2015) analyzed commercial bivalves from Chinese markets and detected a higher average number of 4 particles per gram of meat. Using the same assumptions and figures as Van Cauwenberghe et al. (2015b), a large meal of bivalves would still make only a very small contribution to the TDI of PCBs. Similar preliminary calculations have been made for the contribution of BPA in the human diet from MPs in consumed bivalves. The estimation was that this was small or negligible, which is also expected for other additives (EFSA 2016; Rist et al. 2018). The amount and availability of chemicals contained in MPs in the ocean and transferred to food webs is currently considered to be small or negligible compared to the chemical concentrations found in food organisms (Bakir et al., 2016). More research is needed to develop risk assessments of the impacts of MPs on seafood species and on human health. It is recommended to monitor MPs and the related chemical concentrations in seafood, particularly shellfish, and to get information on human consumption rates of seafood, particularly bivalves.

Additionally, any information on identifying impacts to Environmental Justice communities (e.g., data showing increased exposure to microplastics through higher rates of fish/shellfish consumption) would be beneficial for the Department to perform an impact analysis.

Historically marginalized groups that may rely on subsistence harvesting or small-scale fisheries, tend to be disproportionately exposed to and affected by marine chemical contamination (Landrigan et al., 2020; Liboiron, 2021). People living in Environmental Justice communities in NJ may be at greater risk if they are more likely to collect and eat fish and shellfish from areas which are not safe (e.g., Passaic River). This practice is likely to lead to higher body burdens of toxic chemicals, but previous studies (see above) indicate that the bioaccumulation of such toxicants is much more likely to occur directly from the tissues of the consumed fish/shellfish than from MPs in the consumed seafood.

Question 5. Are there any research initiatives that the SAB would recommend pertaining to the contaminants of emerging concern in addition to the ongoing initiatives (e.g., municipal wastewater treatability, analytical methods for municipal wastewater)?

Department is requesting guidance on next steps for New Jersey and is interested in research initiatives pertaining to micro- and nanoplastics as CECs. As part of this, the Department requests that the SAB review work done so far on the California drinking water regulation proposals. In 2018, Senate Bill No. 1422 was filed requiring the State Water Board to adopt a definition of microplastics in drinking water on or before July 1, 2020, and to adopt a standard methodology to be used in the testing of drinking water for microplastics on or before July 1, 2021. Information on other states' research initiatives or policies would be helpful for the Department to develop possible actions for New Jersey.

SB 1422, Portantino. California Safe Drinking Water Act: microplastics.
Existing law, the California Safe Drinking Water Act, requires the State Water Resources Control Board to administer provisions relating to the regulation of drinking water to protect public health, including, but not limited to, conducting research, studies, and demonstration programs relating to the provision of a dependable, safe supply of drinking water, enforcing the federal Safe Drinking Water Act, adopting implementing regulations, and conducting studies and investigations to assess the quality of water in private domestic water supplies. Under the act, the implementing regulations are required to include, but are not limited to, monitoring of contaminants and requirements for notifying the public of the quality of the water delivered to customers. *This bill would require the state board, on or before July 1, 2020, to adopt a definition of microplastics in drinking water, and on or before July 1, 2021, to adopt a standard methodology to be used in the testing of drinking water for microplastics and requirements for 4 years of testing and reporting of microplastics in drinking water, including public disclosure of those results.*

Suggested Research initiatives:

1. We recommend that the NJDEP form a working group to coordinate with other States that are currently working on plastic pollution, microparticles and nanoparticles to share research and coordinate efforts at the State level.
2. We recommend NJ should identify and map statewide primary sources based on sewage dischargers, manufactures, high density population centers, plastic incineration sources, traffic/road density as it relates to surface water sources including potential atmospheric deposition and microplastic content in precipitation.
3. We recommend that studies be carried out to determine the distribution of nanoparticles and MPs throughout the water column and receiving water sediments. Studies should include groundwater.
4. We recommend developing new or incorporating sampling methods which better reflect the total nano and MPs present in different media (i.e. **not** sample with nets).
5. Sorting methods should be followed by analytical techniques for appropriate characterization: Raman or Infrared spectroscopy, which can also identify the chemical polymer and help determine sources. We do not recommend the use of net-based nano or MP based sampling methods.
6. We recommend the incorporation of specific species to represent Classes of organisms to examine effects of nanoparticles and MPs on life-stages at greatest exposure and risk. These organisms could be currently required organisms used in water quality assessments. This would allow comparison with traditional endpoints and plastic impacts. This could be both in field and laboratory based.
7. We recommend examining the rate and percentage of MPs (of different shapes and sizes) that pass through the gut/gills/epidermis at sublethal levels.
8. We recommend examining what fraction of the contaminants can be desorbed from the gut of different animals during the time the MPs are passing through.
9. Experimental laboratory studies on effects of MPs should focus on microfibers, which are the predominant shapes in the environment, and not use spheres. Since microfibers from textiles contain a unique set of chemicals such as dyes and finishers, studies are needed on these kinds of chemicals, which are quite toxic and greatly understudied.

10. Experimental studies should use environmental concentrations and long-term exposures.
11. Prioritize shellfish (mollusks) particularly for body burden studies to understand microplastic exposure in higher trophic levels including humans.
12. We recommend that sludge applied to terrestrial environments should be sampled and utilize earthworms or other appropriate species for soil to organism transport determinations and toxicity.
13. We recommend examining tire MPs for their fate, transport, distribution and mitigation methods to reduce impact on nearby waterways.
14. We recommend the development of engineering technologies that could be used to reduce environmental inputs.
15. We recommend that there be an education and outreach effort to inform the public of the concerns and how they could play a role in reducing inputs, including (but not limited to) modifications to laundry practices.

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APPENDIX I.
Abbreviations

BPA Bisphenol A
BaP benzo(a)pyrene
EPS extracellular polymeric substances
HD high density
LD low density
MF microfiber
MP microplastics
NP nanoplastic
PA polyacetate
PAH polyaromatic hydrocarbons
PEVA polyethylene vinyl acetate
PCB polychlorinated biphenyl
PE polyethylene
PET polyethylene terephthalate
PFAS Per and polyfluorinated alkyl substances
PFOS Perfluorooctane sulfonic acid
PBDE polybrominated diphenyl ether
PP polypropylene
PVC polyvinyl chloride
HDPE high density polyethylene
LDPE low density polyethylene
SAB science advisory board
ST styrene

Appendix II

Number of Combined Sewer Outflows from NJ cities. *Adapted from NJ DEP, Bureau of GIS*; Updated 2022. https://njogis-newjersey.opendata.arcgis.com/datasets/njdep::combined-sewer-overflow-cso-for-nj/explore?location=40.414950%2C-74.554350%2C9.85

Number of Combined Sewer Overflow (CSO) from NJ Cities to NJ Waterways

	Number of CSO Outfalls to NJ Waterways from Each City																
Cities with COS	Arthur Kill	Cooper River	Cromakill Creek	Delaware River	Elizabeth Channel	Elizabeth River	Hackensack River	Hudson River	Kill Van Kull	Newark Bay	Newton Creek	Overpeck creek	Passaic River	Penhorn Creek	Queen Ditch	Raritan River	Wolf Creek
Bayonne						1		2	10	18							
Camden		9		12							1						
Collingswood				1													
East Newark													1				
Edgewater								2									
Elizabeth	4				2	21				1							
Gloucester				7													
Guttenberg								2									
Hackensack							2										
Harrison													6				
Hoboken								4									
Jersey City							8	10		2				2			
Kearny													5				
Newark					4								13		1		
North Bergen			3														1
Paterson													23				
Perth Amboy	8															8	
Prospect Park													1				
Ridgefield Park							4					2					
Secaucus			2														
Trenton				1													
Weehawken								3									
West New York								2									
TOTALS	12	9	5	21	6	22	14	25	10	21	1	2	49	2	1	8	1