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Water Pollution by Plastics and Microplastics:

A Review of Technical Solutions from Source to Sea

International Water Management Institute

CGIAR RESEARCH PROGRAM ON Water, Land and Ecosystems

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A Review of Technical Solutions from Source to Sea

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RESEARCH PROGRAM ON Water, Land and Ecosystems

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Abbreviations and Acronyms

2nd Trt	Secondary Treatment
ABS	Acrylonitrile Butadiene Styrene
Conc.	Concentration
EPR	Extended Producer Responsibility
EUR	Euros
GESAMP	Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection
HDPE	High-Density Polyethylene
LDPE	Low-Density Polyethylene
MBs	Microbeads
MFs	Microfibres
ML	Microlitter
MLE	Modified Ludzack-Ettinger activated sludge process
MPs	Microplastics
MSW	Municipal Solid Waste
O&M	Operation and Maintenance
PA	Polyamide
PAI	Polyamide Imide
PBT	Polybutylene Terephthalate
PC	Polycarbonate(s)
PCCPs	Personal Care and Cosmetic Products
PE	Polyethylene
PET	Polyethylene Terephthalate
PLA	Polylactic acid
PMMA	Poly Methyl Methacrylate
POPs	Persistent Organic Pollutants
PP	Polypropylene
PS	Polystyrene
PVC	Polyvinyl chloride
SAPEA	Science Advice for Policy by European Academies
SBR	Sequencing Batch Reactor
TN	Total Nitrogen
TP	Total Phosphorus
USD	United States Dollars
WHO	World Health Organization
WWTP	Wastewater Treatment Plant
XPS	Extruded Polystyrene Foam

Refer to Section VIII, Annex A for more abbreviations of types of plastic.

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Summary[1]

I. Introduction

The world demands and produces more and more plastic every year. In 2018, global production of plastics reached 360 million metric tons. This figure is even higher if we include plastics used in manufacturing synthetic textiles, synthetic rubber, and plastic additives. A very small portion of the plastic so far produced in the world has been recycled. Most of the rest has ended up in landfills, open dumps and the natural environment. Part of this plastic finds its way to rivers, lakes and the oceans. If current consumption patterns and waste management practices do not change, by 2050 it is estimated that there will be approximately 12 billion metric tons of plastic litter in landfills, open dumps and the natural environment.

Once they are in the environment, and with time, plastic items tend to degrade to smaller particles through natural weathering processes and can become microplastics (commonly defined as less than 5 mm in diameter). Other microplastics are directly released into the environment. They may have been intentionally added to products, such as personal care and cosmetic products (PCCPs), or they can result from the abrasion or shedding of objects containing plastic (e.g. tyres and synthetic textiles).

Analysis of water and sediments worldwide indicates that macroplastics[2] and microplastics are ubiquitous in aquatic environments, including freshwater and marine ecosystems.

Macroplastics have serious environmental, health and economic impacts, including (but not limited to) blocking canals and sewers, creating breeding habitats for mosquitoes, lowering the recreational and touristic value of landscapes, and damaging the airways and stomachs of animals.

The risks that microplastics pose to the health of humans, animals and ecosystems are of increasing concern. These risks are a function of both hazard and exposure. Given the ubiquity of microplastics in the environment, exposure of humans and other species through water, air, soil and food is rapidly increasing. Potential hazards associated with microplastics come in three forms: physical hazards from the particles themselves; chemical hazards due to, for example, toxic unbound monomers, additives and sorbed chemicals; and microbial hazards if pathogenic microorganisms attach themselves to and colonize microplastics. However, the human health implications of microplastics are still largely unknown and much remains to be learned about their impacts on mortality, morbidity, and the reproductive success of species.

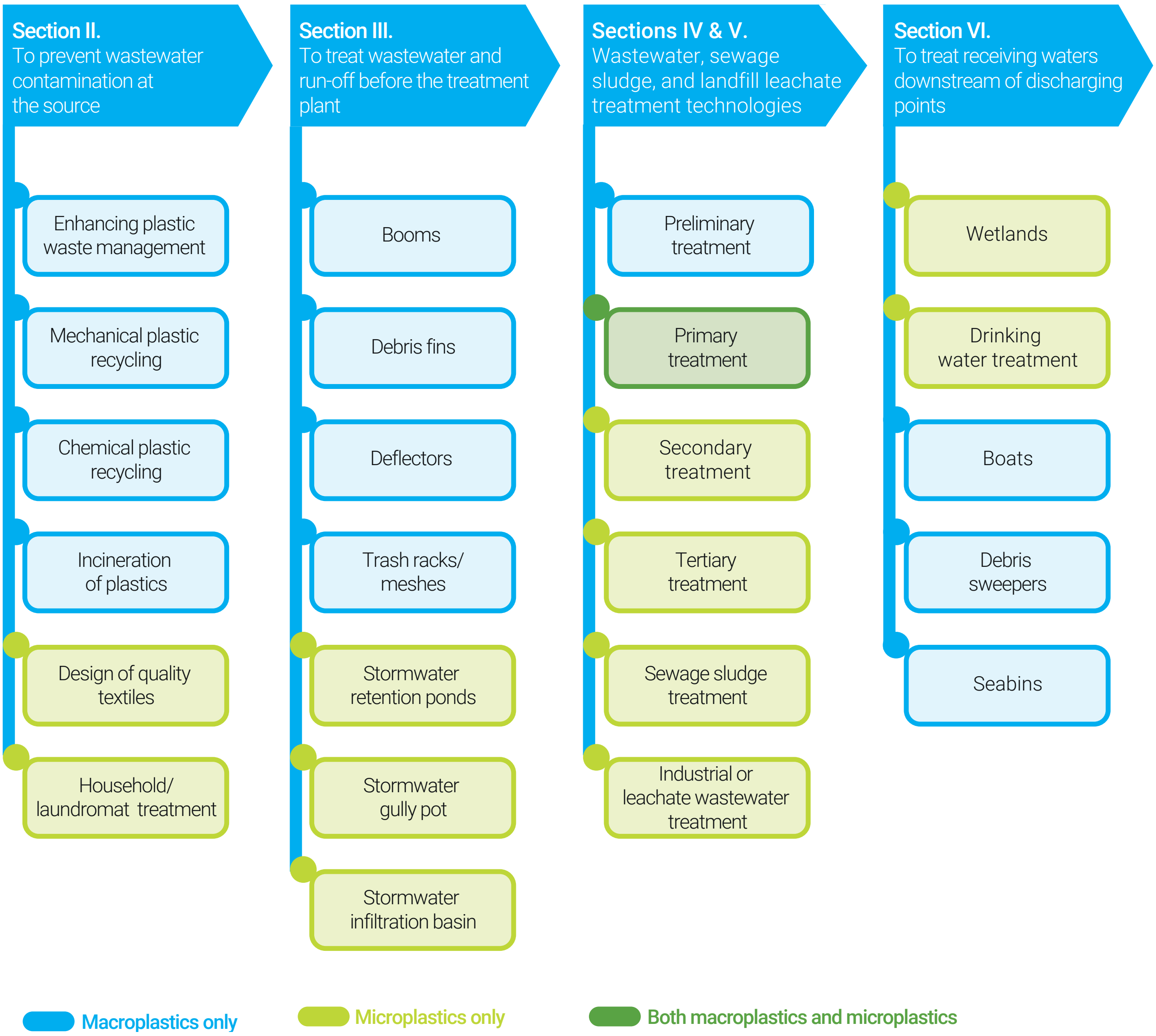
Human risks also have a gendered dimension. Indeed, it is well known that women and men are exposed differently to hazards, e.g. due to biological gender differences such as body size, amount of adipose tissue, reproductive organs or hormones, that can impact the effects and elimination of toxic chemicals and substances. Nevertheless, there is globally a considerable gap of knowledge about the different health effects of microplastics on women and men. The availability of sex-disaggregated data will support the adoption of the necessary policies for adequate safeguards.

The sources of municipal wastewater, in addition to run-off, are residential/domestic, commercial and industrial. Municipal wastewater may be collected through single pipes and channelled to a wastewater treatment plant (WWTP) and/or discharged directly into water bodies. In some cases, separate sewers may exist to carry away run-off. The sources of contaminants in municipal wastewater include plastics and other types of debris. Since plastic waste can be an important source of environmental contamination, it is essential to reduce and remove it before wastewater enters either a WWTP system or freshwater and marine bodies.

The potential and demonstrated risks presented by plastics are high. Meeting the challenge will require urgent preventive action. This report reviews some of the most relevant technologies currently in use and supporting solutions that address contamination by macroplastics and microplastics from source to sea. Where data are available, the report looks at the effectiveness, capital expenditure, and operation and maintenance (O&M) costs of different technologies and their suitability under various conditions. This could help enable policymakers and practitioners to set priorities and select the technologies that would be most cost-effective and suitable in their local context (Figure 1).

1 Sources for data and for substantive statements are provided in the chapters.
2 Macroplastics are plastics larger in size than microplastics. In this report, plastic or plastics generally refers to macroplastics.

Figure 1. Technical solutions for waste management



II. Upstream preventive solutions

Worldwide, daily plastic waste generation per capita has been reported to range from 0.01 kg (in India) to 0.59 kg (in Guyana). Up to 70-85 per cent of plastic waste has been estimated to be mismanaged in Africa and Asia. This report reviews several technical solutions that can be explored to reduce plastic waste at the beginning of the waste management chain and prevent the contamination of water, wastewater and the rest of the environment.

Enhancing macroplastic waste management to reduce impacts. Adequate management of plastic leakage is the first step towards controlling plastic pollution. This requires increasing waste recycling and ensuring the availability of suitable waste handling facilities. Overall, technologies and systems for the collection, storage, transport, recycling and final disposal of solid waste (including waste plastics) must be financially sustainable, technically feasible, socially and legally acceptable, and environmentally friendly.

The waste management sector currently faces numerous challenges, including poor collection systems and road networks, equipment failure, and inadequate waste management budgets. In some locations limited waste segregation at source means that a solution for managing all municipal solid waste (not only plastic waste) is needed to prevent contamination of run-off, wastewater and the rest of the environment.

Improving plastic waste management requires financial investment which is highly context-specific and will allow the enhancement of on-site management, collection (e.g. through acquiring collection trucks which correspond to geotechnical and road network constraints), disposal and recycling. However, in some countries informal plastic collection already plays a key role in the recycling process, leading to the recovery of 10 per cent or more of the plastic waste generated. This job is mostly performed by informal workers who sell the plastic they collect, which is then recycled locally or exported. The informal – and sometimes illegal – nature of this activity makes it difficult to support or to scale up. Within the informal waste economy, studies show that women are often limited to lower-income tasks, such as waste picking, sweeping and waste separation, and could even be displaced by men when informal or voluntary waste-related activities become formalized with pay. In particular cases it may be possible to build on existing value chains to drive the plastic collection sector towards formal establishment, with defined practices and protocols, so as to safeguard the health and livelihoods of these workers.

Plastic recycling (mechanical, chemical and incineration). There are four types of plastic recycling technologies. Primary and secondary recycling, applied respectively to sorted pre-consumer and post-consumer waste, is the reprocessing of a single type of uncontaminated plastic. It produces plastic material of equivalent or lower quality. Secondary or mechanical recycling is the most common plastic recycling technology worldwide. This process depends on the availability of large volumes of single-type selected plastic waste such as polyethylene terephthalate (PET). It may require that specific collection systems are in place, or industrial sorting can be implemented at high cost. The costs of mechanical recycling are typically very low compared to those of other recycling technologies (i.e. United States dollars [USD] 2,000-10,000 to process 1 metric ton/day capacity, while annual operation and maintenance [O&M] costs are USD 500-1,500 to process 1 metric ton/day equivalent capacity).

Tertiary recycling, or chemical recycling, is the most expensive and technically challenging to establish. It involves thermo-chemical degradation of plastics, whether they are sorted or not. This produces products such as liquid fuel or syngas which could be used in different applications, including for virgin plastic production. To achieve financial sustainability, large volumes need to be processed (typically in the range of 50,000-100,000 metric tons per year). Only a few such plants are in operation in the world so far. For chemical recycling the capital cost exceeds USD 385,000 (and can reach USD 857,000) for processing 1 metric ton/day capacity with corresponding annual O&M costs exceeding USD 500-22,000.

Quaternary recycling involves waste incineration for energy recovery. Although incineration takes place in many countries, it is frequently viewed as a non-sustainable solution which is not fully aligned with the evolving principles of a circular economy. Plastic waste has considerable potential for energy generation because the calorific value of plastic is similar to that of hydrocarbon-based fuel. High energy will therefore be released from incinerated plastics. For incineration typical investment costs are USD 260,000-550,000 to process 1 metric ton/day capacity while the associated annual O&M costs are USD 10,800-40,000. It is important to note that chemical recycling or incineration processes release noxious gases, particulate matter and other by-products, (whether intentionally or unintentionally generated) and have gendered health impacts on workers, communities and the environment in general.

Policy tools and behavioural change campaigns. Although they are not a key subject of this report, policy tools and behavioural change campaigns are usually necessary to back up technical solutions. Policy tools entail establishing levies or bans to limit or prevent the use of plastic items such as plastic bags, other single-use plastics, and rinse-off PCCPs containing microbeads (MBs). Typically, total bans can be more expensive to implement than partial ones although the benefits of total bans are more significant. Policy tools may also involve setting effluent quality standards, which helps mitigate the environmental impacts of recycling practices which may be associated with high pollution release in air or water. The success of such measures is variable. For example, lack of awareness, low enforcement, lack of affordable alternatives and non-prohibitive levies have limited their success in many countries. Moreover, it appears that the COVID-19 pandemic has led to a surge in pollution from single-use plastic products including plastic face masks and hand sanitizer bottles.

Consumer decisions affect the volume of macroplastics and microplastics released into the environment. Attitudes and practices may be influenced through behavioural change campaigns. Public education programmes can help improve general understanding of the impacts of macroplastics and microplastics in daily life, as well as encouraging changes in consumer behaviour. Creating gender-sensitive knowledge products highlighting linkages between consumer choices and waste is crucial. Targeted messaging is key. Inclusive stakeholder engagement bearing in mind gendered roles in household consumption and domestic waste management is crucial for introducing new ways of thinking in all sustainable consumption and production practices, as well as in value chain assessments in waste management.

Design of quality textiles. The loss of fibres from textiles is highly dependent on the type of textile. Increased control of production techniques and textile quality, which are related to the manufacturing technology used, could help reduce releases of microfibres (MFs) during textile washing and use. Awareness-raising campaigns, combined with policy measures, can help improve the uptake of innovative and safer (but potentially more expensive) textiles and fashion items. A large percentage of women globally work in at-risk positions in the textile industry. Safeguarding heath in processes is important, as studies have reported that women who work in textile factories and are exposed to synthetic fibres and petroleum products at work before their mid-30's seem to be most at risk of developing breast cancer later in life. Many modern synthetic fibres are basically plastic resin treated with additives such as plasticizers, many of which are recognized mammary gland carcinogens and endocrine disrupting chemicals.

Treatment of effluents from household washing machines or laundromats. Over 840 million domestic washing machines are operated worldwide, using 55 million m^3 per day of water. Around 35 per cent of microplastics in the oceans are estimated to originate from the washing of synthetic textiles. The use of filters to treat effluents from household washing machines or laundromats is being explored by some private companies, but this may not be a cost-effective solution. A washing machine filter typically costs USD 131 per year and per household. Industrial wastewater treatment systems designed for wastewater from large-unit laundromats (which traditionally target removal of contaminants such as oils and suspended solids) can achieve up to 97 per cent microfibre removal. Each of these units costs between USD 5,000 (for a low-cost model made in India) and USD 40,000 (for a European model). Their O&M costs can be high due to energy demand and the use of chemicals in the process. Particularly in water-scarce countries, some of the costs of effluent treatment can be offset through revenues generated from reuse of treated wastewater.

III. Upstream wastewater treatment

Several types of infrastructure can be installed upstream to reduce or remove plastic waste from channelled run-off effectively.

Booms. Booms are logs or timbers that float on the surface of the water. They collect floating macroplastics from wastewater drains. Booms are anchored close to drainage banks (left or right) to allow traffic movement on the water and are cleared using clean-up boats. While they have proven a successful technology for deflecting surface macroplastics, booms do not offer a solution for plastics travelling below the surface. They have the significant advantage of not requiring the installation of permanent structures in a water body bed. Other factors that can influence the efficiency of floating booms are intense run-off of water along the drain, and wind that brings waste back to the banks or causes plastics to escape from the booms. Acquisition costs for booms are USD 485-1,200 per metre of boom, based on length, design and materials. Typical annual O&M costs have been reported to be USD 533 per metre of boom.

Debris fins and debris deflectors. *Debris fins* are barriers built in the drainage channel immediately upstream of an engineered structure to direct plastics away from that structure. They allow plastics to continue travelling in the flow in a directed manner. *Debris deflectors* are triangular-shaped frames placed immediately upstream of a dam or drain to deflect and guide plastics through and/or away from the channel entrance. They are used extensively in bridge construction and are intended to position large plastics to pass in a directed manner. This also facilitates their subsequent removal. The construction costs of debris fins and debris deflectors are part of bridge construction budgets. They both need to be coupled with other systems for effective plastic capture.

Trash racks/meshes. The most common way to deal with plastic waste in traditional facilities is to use a trash rack to keep plastics from entering the WWTP. The rack traps the plastics, and accumulated plastics are removed by manual or mechanical raking (the latter is standard for large facilities). The use of trash racks presents two major challenges: 1) accumulating plastics, which leads to head loss on the racks themselves; and 2) structural fatigue of the racks, which is an important design concern. Typically, acquiring such systems costs USD 3,000-30,000 per unit depending on size and materials. Annual O&M costs for manual and mechanical clean-up units are USD 1,800-9,000 and USD 2,100-9,700, respectively.

Stormwater retention ponds. Microplastics in run-off which is not intended to be directed to a WWTP could be removed before the water is discharged into freshwater bodies. For particulate material such as microplastics, sedimentation and deposition are the main removal mechanisms. In many locations this treatment takes place in artificial basins (water retention ponds). The run-off is channelled into the pond and held there for a period of days to weeks before discharge, allowing microplastics and other particulates to settle. Scientific research has shown that retention ponds are the most effective run-off water management installation for removing microplastics. However, proper design and maintenance are necessary. For example, sediments should be removed from the pond when necessary.

Gully pots. Gully pots (also known as catch basins in North America) are small sumps in the urban roadside drain which act as run-off inlet points. Their main purpose is to retain sediments (e.g. sediments containing microplastics) from road run-off which would otherwise enter drains and sewers. Gully pots are available in a range of diameters and depths and are made from a variety of materials. The most common way to clean them is to use an "eductor truck" that applies hydrodynamic pressure and a vacuum to loosen and remove sediments (including microplastics) and the standing liquids. The costs of cleaning gully pots can differ depending on the methods used, the frequency with which they must be cleaned, the amount of sediments removed, and the costs of disposing of the sediments. Exact cost data could not be obtained for this technology.

Infiltration basin. The infiltration basin is a sedimentation technique whose use in removing or reducing microplastics in run-off is increasingly common. It consists in water impoundment over porous soil. Stormwater run-off is received and contained until the water infiltrates the soil, thereby enriching groundwater reserves. Infiltration basins can provide full control of peak or large volumes of stormwater run-off. If the run-off contains high amounts of soluble contaminants, groundwater contamination can occur. Research has shown that most existing infiltration basins have the highest failure rates of any microplastic removal system. For this system the most critical maintenance item is periodic removal of accumulated microplastics from the basin bottom. If microplastics are allowed to accumulate, the surface soil will become clogged and the basin will cease to operate as designed.

IV. & V. Wastewater treatment plants

Treating municipal wastewater in a plant is the norm in many developed countries. However, only 33 per cent of the population in low- and middle-income countries is connected to a sewer. The wastewater of the remaining 67 per cent is collected and pre-treated in on-site systems or discharged directly to soil and into water bodies. Conventional treatment of wastewater requires **preliminary**, **primary**, **secondary** and **tertiary** treatments. In general, a minimum of secondary treatment is necessary to meet water effluent quality standards for discharges in most countries.

Current knowledge highlights that while plastics are mainly removed during preliminary treatment, microplastics may be removed through fine screening (primary treatment), sedimentation (primary or secondary treatment), flotation (primary treatment) and filtration processes (primary, secondary or tertiary treatment). In addition, coagulation-flocculation (primary treatment) could help facilitate microplastics removal during primary sedimentation (Table 1). The removal of microplastics is not consistent throughout the treatment process. While some stages (e.g. those involving high sludge concentrations) concentrate microplastics in the process, others (e.g. clarifiers) lead to microplastics removal. Therefore, removal of microplastics in a treatment plant is a complex process which does not occur in one step. The wastewater treatment process targets different contaminants, and interactions may therefore be observed.

Although several studies have been published on this subject, data on microplastics removal have usually been obtained using various methodologies. The differences observed from one study to another may be attributed to variations in methodologies (e.g. with respect to sample collection, processing and analysis). This emphasizes the need for harmonization and standardization of analytical techniques. Moreover, one-time measures are taken often and follow-up measurements seldom. There could be wide temporal and spatial variations in influent and effluent wastewater quality between different countries and studies. The conclusions presented here should be viewed as indicative at this time.

Based on those currently studied (in North America and Europe), estimated daily discharges through treated wastewater for a conventional WWTP remain at about 10-60 grams of microplastics per day, mostly depending on the total volume of treated wastewater. During the treatment process microfibres are removed well from the wastewater (i.e. a large percentage removal). However, microbeads and small microfibres could still be released in the treated effluent. Overall in the Unites States, WWTPs contribute less than 0.1 per cent of the microplastics contaminating water bodies and the rest of the environment. The situation should, however, be different in countries where WWTPs are not yet functional or where wastewater treatment coverage is low.

Sewage sludge treatment. While removal of microplastics from treated wastewater can reach 69-99 per cent in a WWTP, it is important to remember that this removal is simply a phase transfer of the microplastics from the liquid to the sludge. Therefore, inadequate management of the sludge will lead to environmental contamination. Sewage sludge contains 4.4-14.9 microplastic counts/gram wet weight or 23-240 microplastic counts/gram dry weight (DW), based on the type of process and the stage at which collection takes place. The average size of microplastics in sludge is larger than in the initial wastewater, demonstrating that the sludge mainly concentrates large microplastics while removal of small microplastics is low. Microfibres typically represent 63-80 per cent of microplastics in sludge.

Once the sludge is stabilized, land application is the main post-treatment process. However, this should be controlled as it contributes to increasing the microplastic content of soils. As a result, the burden of microplastics in soils is greater than the current burden of microplastics in oceans. Microplastics in soils may also be carried by run-off water or wind to nearby aquatic environments and ultimately into the oceans. Microplastics can be found in soils even five to 15 years after application of sludge, although the impacts, including on soil organisms, farm productivity and food safety, are yet to be investigated in detail. To avoid these impacts, all or highly contaminated sludge fractions could be incinerated, which would result in the loss of organic matter value from the application of sludge. The costs of sludge management are usually considered to be part of conventional WWTP costs.

Industrial or leachate wastewater treatment. The textile processing and plastic manufacturing industries may release large amounts of microfibres. A typical industrial WWTP can carry 300 microfibres per litre (i.e. 10 billion microfibres per day at a typical Chinese plant), while the treatment currently applied is able to remove 95 per cent of them. Per process, removal efficiency is 76 per cent during preliminary and primary treatment, 32 per cent during the secondary sedimentation stage, and 70 per cent during coagulation and filtration. There are knowledge gaps in regard to how different types of particles and pigments respond to the wastewater treatment process.

Table 1. Description of removal of macroplastics and microplastics (MPs) during wastewater treatment processes

Treatment stage[a]	Preliminary	Primary	Secondary	Tertiary
Sequence of processes and objectives	Screening with metal grids as preliminary treatment to remove fine and coarse debris, i.e. > 10 mm in size	Grit removal (to remove sand, silt and other heavy particles) 1. Skimming tank for grease, oil and fat removal 2. Coagulation and flocculation to create large flocs of heavy metals and phosphorus 3. Primary sedimentation to remove particulate matter and flocs 4. Flotation to remove floating materials, volatile organic compounds (VOCs) (e.g. those which are strong-smelling) and grease	Biological and physical treatment removes: • Suspended particles • Dissolved nutrients • Dissolved and colloidal organics Examples of processes are: 1. Aerobic, anoxic or anaerobic biological treatment, such as • Activated sludge • Membrane bioreactors[b] 2. Secondary sedimentation	It may ensure final effluent meets the required quality standard. Also used to remove nutrients or heavy metals (if necessary) Examples of processes are: • Wetlands • Membrane filtration • Biological aerated filter • Slow sand filtration • Disc filtration
Plastics removal	Removal mainly occurs during this step	Some of the macroplastics are removed during fine screening, skimming, grit removal and other processes if these processes are implemented	Smaller plastic items such as cotton swabs may remain in the wastewater	Not expected because most plastics would have been removed already
MPs removal	Up to 59 per cent	42-82 per cent in general; exceptionally, 78-95 per cent. • (major route) skimming of grease (for floating MPs) • (minor route) filtration and gravity settling for heavier MPs trapped in flocs	86-99.8 per cent, cumulatively • Removal mechanisms are uncertain. • MPs fragments are more easily removed compared with MFs, possibly because MFs were largely removed during the primary step	Typically, cumulative removal is 95-99.9 per cent. Effluent concentrations are 0.01-91 MP per litre.
Cost in Europe or the United States	Not available, as treatment at this level is insufficient to meet the quality standards for treated wastewater		Based on different sources, total capital + O&M costs usually average USD 1,295 to over USD 3,308 per m^3/day (or USD 518 to over USD 1,324 per capita)	• Capital + O&M costs exceed USD 1,717 per m^3/day (conventional units) (or USD 687 per capita) • For wetlands, capital + O&M costs average USD 159 per m^3/day (or USD 64 per capita)
Process costs in developing countries	• Investment costs: USD 3-40 per capita • O&M costs: USD 0.1-2 per capita		• Investment costs: USD 10-150 per capita • O&M costs: USD 0.2-8 per capita	• Not available

a Treated effluent is disinfected to reduce loads of pathogens exiting the treatment plant. This is achieved using chlorine or chlorine dioxide; ozone, peracetic acid or other chemicals; or ultraviolet (UV) radiation. The costs of this step are combined with the costs of earlier treatment stages.
b Achieves secondary and tertiary treatment simultaneously.

Microplastics in landfill leachate originate from the fragmentation of plastics buried in the landfills, but how contamination occurs is not yet fully understood. Typical concentrations of microplastics in landfill leachate in China were 0.42-24.58 per litre, with landfill leachate generation estimated at 1.3-3.2 m^3 per metric ton of waste over a 100-year period. In the leachate 17 types of plastic materials were identified, of which polyethylene (PE) and polypropylene (PP) represented 99 per cent. Shape-wise, pellets (59 per cent), fragments (23 per cent) and fibres (15 per cent) were the most abundant; 78 per cent of the microplastics were between 0.1 and 1 mm in diameter. O&M costs for a landfill leachate treatment plant are USD 1,460-3,240 per m^3/day of leachate treated (i.e. 2-7 per cent of the capital cost).

VI. Downstream water treatment

In cases where upstream strategies for the removal of macroplastics and microplastics have failed or are non-existent, some treatment solutions can be implemented in water bodies to remove plastic waste and reduce risks and human exposure to microplastics.

Wetlands. These nature-based treatment systems are known for their ability to improve water quality while relying on natural processes involving vegetation, soils, and their associated microbial assemblages to filter water as it passes through the system. For conventional contaminants the removal mechanisms are primarily through transformation and uptake by microbes and plants, as well as assimilation and absorption into organic and inorganic sediments. However, the capacity of wetlands (either constructed or floating systems) to reduce microplastics has not been much studied. The very few published studies highlight that constructed wetlands may be able to remove from water and concentrate in sediments high levels of small and rather large microplastics (typically 99.8 per cent of those 20-30 μm in diameter and 100 per cent of those larger than 300 μm in diameter). However, as in the case of all extensive processes, the land requirement is high, which could be a constraint in areas where land is scarce. Capital and O&M costs average USD 159 per m^3/day of treatment capacity in the United States.

Possible uptake of microplastics via drinking water is also of great concern. Microplastics have been detected in drinking water and drinking water sources. They have been detected in bottled water in several countries, including in samples from glass bottles. To date there is no legislative limit for microplastics content in drinking water, nor has any treatment technology been optimized and targeted directly at the removal of microplastics. Gendered effects of drinking water quality on health, with regard to plastics, are also not fully known.

Drinking water treatment. Modern drinking water treatment plants which use processes such as coagulation and advanced filtration could be capable of removing small microplastics (i.e. those smaller than 10 μm in diameter), but less efficient at removing larger particles. The O&M costs of drinking water treatment in the United States are reported to be around USD 1.5 per m^3. These costs vary greatly with a number of factors such as the source of the water, which determines on the contaminants it contains (typically suspended and colloidal solids, silica and colloidal silica, bacteria, hardness, etc.) and the level of treatment required to attain the water quality needs.

Clean-up boats. Clean-up boats are designed to collect plastics from river surfaces. They use skimmers or conveyor belts to collect the plastics as they move along the surface of the water. This is a very practical option for river plastic clean-up. Clean-up boats have been deployed successfully on several rivers in the United States.

Debris sweepers. Sweepers are used to control water upstream of a structure such as a bridge. They are intended to buffer any structure from impact while they steer plastics around a downstream structure. Because sweepers rotate freely, they shed plastics. This greatly reduces the likelihood of accumulation. Experts have expressed disparate opinions on the merits of sweepers. They may be subject to failures due to clogging, which is influenced by the water flow speed.

Seabins. Seabins are floating trash cans placed in the water. Each bin is attached to a dock and powered by a pump that sucks water from the top opening through a filter bag at the bottom to collect particles, including plastics. Seabins are designed to be placed in calm flow waters near a power source (e.g. a dock or marina). Occasionally filter bags filled with waste are removed and replaced. Typically, a seabin is estimated to collect up to 1.4 metric tons per year of floating plastics, ranging from small to large particles. A bin costs some USD 4,000 to acquire and USD 1,200 per year to operate and maintain.

Selecting and combining technologies

When decision-makers and experts plan to address water pollution by plastics and microplastics, they need to agree on the desired water quality in the local context and plan accordingly. Once water quality objectives for plastics and microplastics are established, the most relevant sources of pollutants and pathways to water should be identified. For example, in a given watershed plastic litter could be the most critical source of contamination while, in another, microfibres from synthetic textiles or microplastics from tyre abrasion could be the most relevant sources and pathways.

Based on this understanding, decision-makers can select the most cost-effective and sustainable combination of solutions. To that effect, inclusive stakeholder engagement is necessary to ensure that gender, diversity and inclusion are given the prominent importance they deserve. Therefore, there is need to acquire more insight on the gendered impacts of waste management and the associated impacts. It is crucial to encourage collection of sex-disaggregated data and analysis of these data to support policy formulation. Disaggregated data reveal important gender dynamics and are crucial for gender-sensitive policy formulation. Adequate data enhance understanding of life cycle and intergenerational links with regard to deprivations and support the alignment of actions with needs, leading to better designed policies in specific regional and national contexts.

Technical considerations will also play a role in the decision-making process. For example, to achieve a desired maximum number of microplastics in drinking water, a recycling solution for plastic waste (upstream) could be combined with secondary stage wastewater treatment and conventional drinking water treatment (downstream). The final choice will mainly depend on the combination of solutions which is feasible in the local context and achievable at minimum cost. However, costs and effectiveness will not be the only criteria. The capacities and perceptions of local stakeholders, along with other practical challenges to the adoption of particular solutions in a local context, will also influence the final selection.

Section I

Introduction

A. A growing challenge

The benefits of plastic are undeniable. It is cheap, lightweight, easy to handle, and often the most economical option in certain applications. Because of these attributes, plastic is commonly used, for example, in packaging, building and construction materials, automotive components, electrical and electronic parts, household and leisure products, and agricultural equipment.

The world demands and produces more and more plastic every year. In 2018 global production of plastic products reached almost 360 million metric tons (Mt) (PlasticsEurope 2019). This figure is even higher if synthetic textiles (65 Mt in 2017; Textile Exchange 2018), synthetic rubber (15 Mt in 2016; International Rubber Study Group [IRSG] 2017) and plastic additives (Geyer, Jambeck and Law 2017) are taken into account. This growth will continue at least during the next few decades. Considering estimated worldwide population growth and current consumption and waste practices, plastic production is predicted to double by 2025 and more than triple by 2050 (Food and Agriculture Organization of the United Nations Lusher *et al.* 2017.

According to Jambeck *et al.* (2015), out of 2.5 billion metric tons of solid waste generated by 192 countries in 2010, about 275 Mt consisted of plastic. Only a small fraction of the plastic waste generated is properly collected and disposed, while an even smaller fraction is recycled. As a result of mismanagement, a large portion of this plastic ends up in the environment. Part of it finds its way to rivers, lakes and the oceans. Jambeck *et al.* (2015) estimated that of 275 Mt of plastic waste generated in 192 coastal countries in 2010, 4.8-12.7 Mt ended up in the oceans. More recently Ryberg *et al.* (2018) estimated the losses of plastics to the environment from littering and mismanaged waste treatment to be 4.67 Mt. Plastic waste and pollution have serious environmental, health and economic impacts, including (but not limited to) blocking of canals and sewers, the creation of breeding habitats for mosquitoes (such as the dengue vector *Aedes* sp.) and physical harm to animals.

Once in the environment, and with time, larger plastic items tend to degrade to smaller particles through natural weathering processes and to become microplastics (MPs), generally defined as particles less than 5 millimetres (mm) in diameter. Other microplastics are directly released into the environment. They can be an intentional addition to products (e.g. scrubbing agents in personal care and cosmetic products [PCCPs]) or they may originate from the abrasion of objects made with plastics, such as tyres and synthetic textiles. Losses from clothes do not occur only during washing. Wearing clothes can release even greater quantities of microfibres to the environment (De Falco *et al.* 2020). Boucher and Friot (2017) concluded that, globally, 0.8-2.5 million Mt of microplastics are discharged into the oceans every year

Microfibres (MFs), a subcategory of microplastics that come from the abrasion of synthetic textiles, are of particular concern. They are reported to be the most abundant type of microplastics in wastewater, freshwater and the oceans (Sundt *et al.* 2014; Herbort *et al.* 2018a; Herbort *et al.* 2018b). Ingestion of microfibres by zooplankton, benthic organisms and mussels can be more harmful than their consumption of other microplastics. The characteristic shape of microfibres lends itself to entanglement with other fibres in the intestinal tract, which can result in non-biodegradable gut blockage (Cole *et al.* 2013).

Analysis of water and sediments worldwide (Browne *et al.* 2011; Sundt *et al.* 2014; Driedger *et al.* 2015; Rhodes 2018; Koelmans *et al.* 2019) indicates that microplastics are ubiquitous in aquatic environments, including both freshwater and marine ecosystems. Microplastics have been detected in air, soil, food and drinking water (both bottled and tap water) and even in Arctic and Antarctic,or polar ice.

There is growing concern about the risks microplastics pose to the health of humans, animals and ecosystems. These risks are a function of both hazard and exposure. Given the ubiquity of microplastics in the environment, exposure of humans and other species is rapidly increasing.

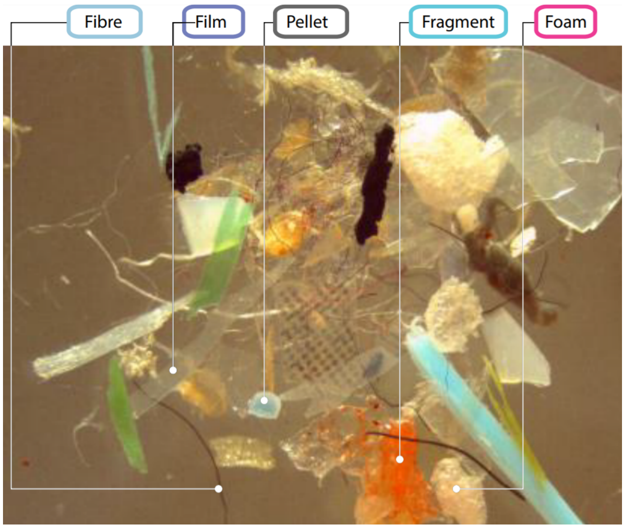
Potential hazards associated with microplastics can be physical, chemical or biological. The particles themselves can potentially cause gut or respiratory blockages in animals. Microplastics may also carry toxic chemicals, including persistent organic pollutants (POPs) and pathogenic microorganisms that may attach themselves to and colonize microplastics (Wright *et al.* 2013; Science Advice for Policy by European Academies [SAPEA] 2019; Wang *et al.* 2019; World Health Organization [WHO] 2019). The impacts of environmental exposure to microplastics on mortality, morbidity and the reproductive success of species continue to be investigated.

Meeting the challenge of plastic pollution will require changes in consumption and production patterns, in line with United Nations Sustainable Development Goal (SDG) 12 (responsible consumption and production). It will be key to improving people's health and well-being (SDG 3) and the health of the planet (SDG 6, clean water and sanitation; and SDG 14, life below water) and will need to include all sectors in society (SDG 5, gender equality and SDG 10, reduced inequalities).

B. Sources of microplastics and pathways to freshwater and the oceans

Microplastics include a wide range of materials with different sources, chemical compositions, shapes, colours, sizes and densities (Figure 2). There is no scientifically agreed definition of microplastics. Different definitions have been proposed in the literature (Lassen *et al.* 2015; Thompson 2015; Verschoor 2015).

Figure 2. Different examples of microplastics



Source: UNEP (2016)

A commonly used definition describes microplastics as plastic particles smaller than 5 mm in diameter. Some definitions propose a lower threshold of about 1 micrometre (μm), which is often chosen solely because of the limitations of the sampling and analytical technique used. Particulate plastics in the order of 1-100 μm in length or smaller are often called nanoplastics, but with an unclear upper threshold.

Microplastics are sometimes categorized as primary and secondary with, once more, different definitions in the literature (e.g. Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection [GESAMP] 2016; Boucher and Friot 2017). In this study the following definitions are adopted:

- *Primary microplastics* are specifically manufactured in the microplastic size range. They can be, for example, industrial abrasives used in sandblasting and microbeads used in PCCPs. They can also be plastic resin pellets (typically 2-5 mm in diameter) or powders used in plastics production (Boucher and Friot 2017). Other examples include plastic coatings for controlled-release fertilizers.
- *Secondary microplastics* originate from the fragmentation or degradation of larger plastic items (including single-use plastics) once they enter the environment. This happens to mismanaged plastic waste (e.g. discarded plastic bags) through photodegradation and other weathering processes, or to unintentionally lost plastic items such as fishing gear. Secondary microplastics can also originate from the abrasion of plastic objects during manufacturing, use or maintenance. For example, 0.033-0.178 grams of microplastics per kilometre (km) have been estimated to be generated as a result of tyre wear (Sundt *et al.* 2014). Secondary microplastics from diffuse sources may enter wastewater through run-off, as well as more directly entering the natural environment.

The main sources of macroplastics and microplastics and their pathways to water are shown in Figure 3 and described in the following paragraphs.

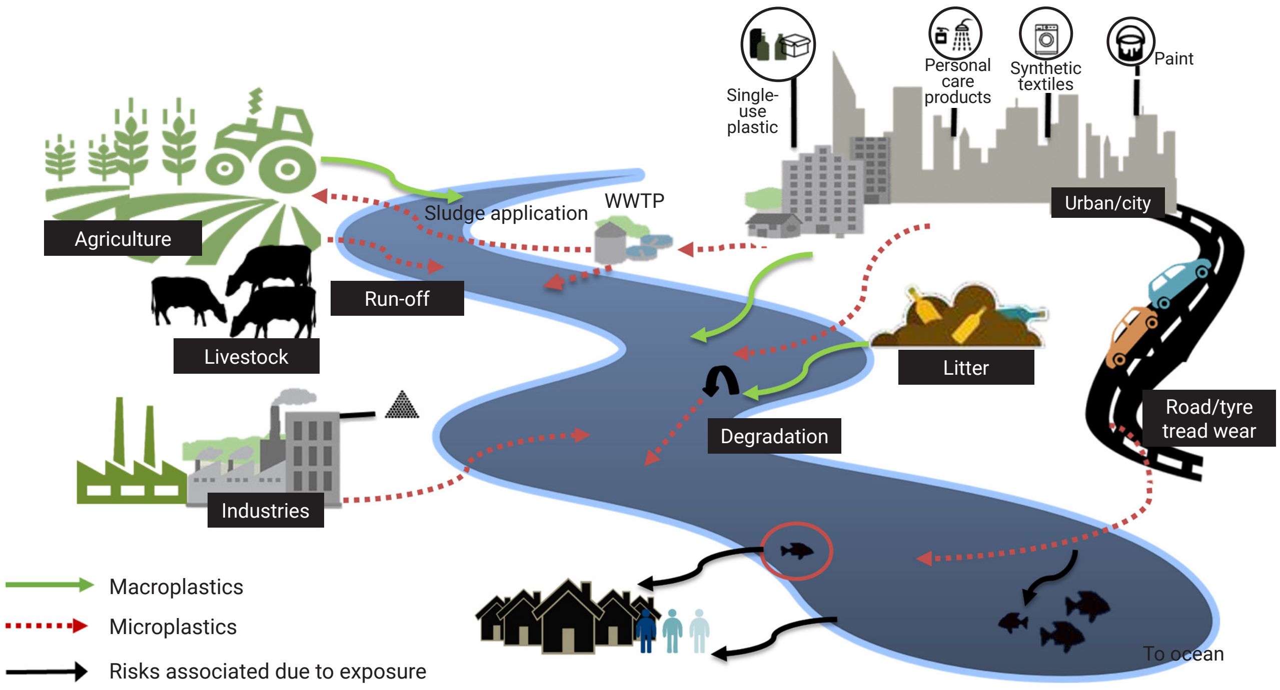
Personal care and cosmetic products (PCCPs) may contain microbeads for a variety of purposes. For example, they may be sorbents used for the delivery of active ingredients, exfoliation or viscosity. Some PCCPs contain several thousand microbeads per gram of product, with microplastics representing up to 10 per cent of product weight, roughly the same amount as in product packaging (Lassen *et al.* 2015; Leslie 2015). Typically, microplastics from PCCPs enter urban wastewater networks from households, hotels, hospitals and other urban facilities. Women are the biggest users of PCCPs and have the highest direct exposure to micoplastics in these products.

Plastic pellets (nibs or nurdles) are used by industries that generate plastic products. During manufacturing, processing, transport and recycling, pellets can be accidentally spilled into the environment and end up in water either directly or through soil erosion and run-off (Essel *et al.* 2015).

Washing *synthetic textiles* in households and laundromats creates primary microplastics through abrasion and shedding of fibres. These fibres are typically made of polyester, polyethylene, acrylic or elastane (Essel *et al.* 2015). Microfibres are discharged from washing machines into urban wastewater networks and sewage systems.

Erosion of *tyres* while driving forms microparticles from the outer parts of the tyres. These microplastics consist of a mix of synthetic polymers (approximately 60 per cent) with natural rubber and many other additives (Sundt *et al.* 2014). Tyre microparticles can be spread by wind or washed off the road as run-off.

Figure 3. Main sources and pathways of macroplastics and microplastics to water



Weathering of *road markings* or their abrasion by vehicles is another source of microplastics. Road markings are used during the development and maintenance of road infrastructure. In most European countries they are commonly made of thermoplastics (Lassen *et al.* 2015). Microplastics from this source are spread by wind or washed off the roads by precipitation before reaching water bodies.

As plastic products and *plastic debris* (including abandoned fishing gear and plastic packaging, bottles and bags) are exposed to UV radiation, they undergo weathering degradation and gradually lose their mechanical integrity (Andrady 2007a; Andrady 2007b). With extensive weathering plastics generally develop surface cracks and fragment into progressively smaller particles (GESAMP 2015). Degradation can occur on land and in freshwater and marine environments. Data are limited on the rates of fragmentation and degradation of macroplastics in the environment, but degradation through weathering generally occurs rapidly on riverbanks and beaches and relatively slowly in debris floating in freshwater and marine environments. It is possible that further fragmentation of microplastics to nanoplastics can occur, but the amount of such nanoplastics in the environment has barely been assessed (Alimi *et al* 2017).

These are examples of the main sources of microplastics, but there are others such as synthetic paints (GESAMP 2016) and controlled-release fertilizers which encapsulate plant nutrients within a coating often composed of a polymer (e.g. polysulfone, polyacrylonitrile and cellulose acetate) (Jarosiewicz and Tomaszewska 2003; GESAMP 2016).

There are limited data with which to quantify the contributions of different sources to water systems. However, a global modelling study (Boucher and Friot 2017) concluded that two-thirds of the microplastics released into oceans come from the erosion of tyres and synthetic textiles This study analysed the sources of microplastics the authors considered potentially most significant (i.e. synthetic textiles, 35 per cent of total releases; tyres, 28 per cent; city dust, 24 per cent; road markings, 7 per cent; marine coatings, 3.7 per cent; personal care products, 2 per cent; plastic pellets, 0.3 per cent). It did not consider secondary microplastics from plastic litter and (primary or secondary) microplastics from other potential sources such as agricultural land and equipment or domestic paints. The relative contribution of different sources of microplastics to freshwater environments remains unassessed at the global level, but country studies (e.g. in Norway) suggest that microfibres from synthetic textiles are the most important source (Sundt *et al.* 2014; Herbort *et al.* 2018a; Herbort *et al.* 2018b).

Both primary and secondary microplastics can enter water bodies through pathways including atmospheric deposition, run-off from land (e.g. from roads or agricultural land), and municipal wastewater. Microplastics enter the environment through deposition to water and soil. Water contamination could be channelled through drains or sewers to WWTPs when they exist, or directly to water bodies. Microplastics that accumulate in or are deposited on soil may contaminate the air via wind and may also contaminate rainwater run-off.

Urban wastewater (including urban run-off)

Wastewater from cities and towns, including urban run-off, collects microfibres from the abrasion of synthetic textiles, lost microbeads from personal care products, microplastics from eroded tyres, secondary micro-sized fragments from plastic litter, and other microplastics. While the relative proportion of these microplastics in raw wastewater may vary significantly with local contexts, microfibres are typically the most abundant type of microplastics, followed by microfragments (Gies *et al.* 2018). Urban wastewater and run-off could be channelled through drains or sewers to WWTPs when these exist, or directly to water bodies.

Wastewater treatment removes a large share of the microplastics from wastewater, but most of them accumulate in the sewage sludge produced during wastewater treatment (Zubris and Richards 2005). Similarly, in areas not connected to sewers and treatment plants microplastics lost at household level are collected in on-site sanitation systems such as septic tanks and concentrate in faecal sludge. In many countries both faecal sludge and sewage sludge are used formally or informally on agricultural land.

Agricultural run-off

Agricultural run-off is a significant microplastics pathway to water where sewage sludge has been applied to land or agricultural plastics have been used (Horton *et al.* 2017).

Plastics are used in agriculture for various purposes (e.g. irrigation equipment, greenhouses, mulch). These plastics can be exposed to sun for many months and, when plastic objects are moved, can readily break down into microplastics. One of the newest fertilization technologies, controlled-release fertilizers, encapsulates nutrients within a coating often composed of a plastic polymer which is sometimes called a "nutrient pill" (Jarosiewicz and Tomaszewska 2003; Landis *et al.* 2009). Controlled-release fertilizers improve nutrient use efficiency (Jacobs 2005) and reduce costs and nutrient run-off into water systems (Landis *et al.* 2009). However, they have introduced a new environmental risk in the form of microplastics contamination.

Road run-off

When microplastic losses occur on roads (e.g. from abraded tyres and road markings or lost plastic pellets), a portion is transferred by wind and another portion through road run-off. In the case of urban roads, run-off can be collected in sewers or drains (Boucher and Friot 2017) and transferred to WWTPs (if they exist) or to surrounding watercourses directly, or indirectly if microplastics accumulate first in soils. Outside cities, the latter is the most typical pathway.

Atmospheric deposition

Part of the secondary microplastics originating on land, including those from abraded tyres and road markings, can spread by air as plastic dust. This dust which may travel long distances and enter water systems directly or with precipitation.

C. Occurrence of microplastics in freshwater and the oceans

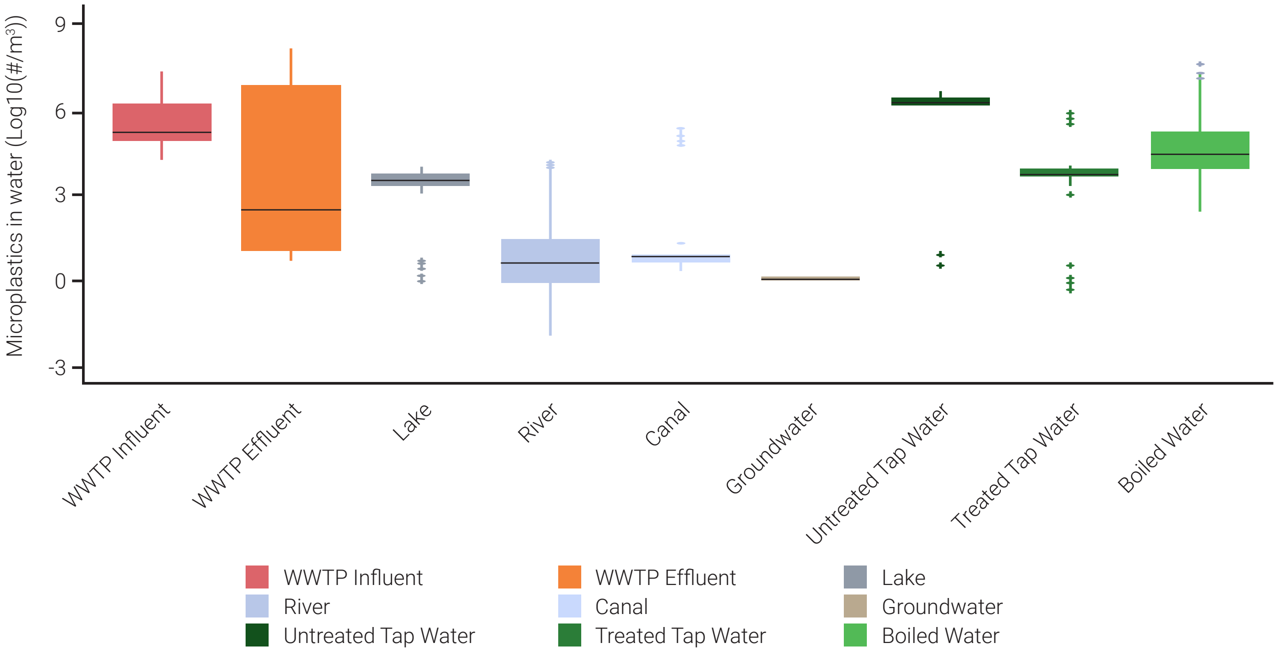
Global estimates of microplastics in the oceans suggest that land-based activities are a significant source, while the rest of microplastics could potentially come from fragmentation of marine plastic litter and lost fishing nets. Considering the estimates by Jambeck (2015), current microplastic releases from plastic waste could be more than 12.7 Mt per year. Considering those of Boucher and Friot (2017), more than 2.5 Mt of microplastics could come from land-based activities.

The global load of microplastics into freshwater systems has not been assessed. However, microplastics in freshwater are increasingly reported, with some studies suggesting high levels of contamination worldwide (Browne *et al.* 2011; Sundt *et al.* 2014; Driedger *et al.* 2015; Rhodes 2018).

The fate of microplastics in freshwater depends on complex interactions between biophysical factors (e.g. run-off, wind, water flow) and the properties of the microplastics themselves. Lighter microplastic particles in fast-flowing rivers may be transported directly downstream and eventually to marine environments. Where flow velocities are low and microplastics are heavier, it is probable that they will sink and be entrained in benthic habitats.

Koelmans *et al.* (2019) carried out a review of recent literature which included 31 freshwater records (Figure 4). The majority of studies in rivers were conducted in Europe and North America and, to a lesser extent, in Asia and South America. Microplastics have also been reported in lakes in Africa. A lack of standard methods for sampling and analysing microplastics in the environment makes it difficult to compare studies. A wide range of shapes and sizes were found in freshwater. Particle counts ranged from around 0 to 1,000 particles/litre in freshwater. Only nine studies were identified that measured microplastics in drinking water; they reported particle counts in individual samples from 0 to 10,000 particles/litre and mean values from 10-3 to 1,000 particles/litre. The smallest particles detected were often determined by the size of the mesh used in sampling, which varied significantly across studies. In most cases freshwater studies targeted larger particles, using filter sizes that were an order of magnitude larger than those used in drinking water studies. Thus, direct comparisons between data from freshwater and drinking water studies cannot be made.

Figure 4. Median and variation in microplastic number concentrations in individual samples taken from different water types



Reference: Koelmans *et al.* (2019)

D. Risks from microplastics

The risks microplastics pose to humans and the environment are a function of both hazard and exposure.

Potential hazards associated with microplastics take three forms: the particles themselves, which may present physical hazards; releases of toxic chemicals from plastics; and pathogenic microorganisms that may attach themselves to and colonize microplastics, known as biofilms (Wang *et al.* 2019; WHO 2019).

Particles may cause impacts in the body depending on their properties, including size, surface area and shape. However, the transport, fate and health impacts of microplastics following ingestion are as yet not well studied (WHO 2019).

Although plastic polymers are generally considered to be of low toxicity, macroplastics and microplastics can release monomers and additives that are toxic (WHO 2019). Five plastic types are currently classified as carcinogenic (category 1A) or both carcinogenic and mutagenic (category 1B): polyurethanes, polyacrylonitriles, polyvinyl chloride, epoxy resins, and styrenic copolymers. The toxicity of these polymers is a result of their monomer constituents (Lithner *et al.* 2011; Lam *et al.* 2018). In addition, hydrophobic chemicals in the environment, including POPs, may sorb to the plastic particles.

Most microorganisms that are part of biofilms which colonize microplastics are non-pathogenic. However, some biofilms can include pathogens such as *Pseudomonas aeruginosa, Legionella* spp., non-tuberculosis *Mycobacterium* spp. and *Naegleria fowleri* (WHO 2019).

Humans are exposed to microplastics in different ways. Exposure can occur through ingestion of microplastics in water or food, inhalation, and dermal or surgical exposure (Murphy 2017). Microplastics have been reported in air, water (including surface water, groundwater and drinking water), soil and food samples. In addition to their presence in seafood, microplastics have been detected in edible consumer products such as beer and salt (Liebezeit and Liebezeit 2014; Yang *et al.* 2015; European Food Safety Authority [EFSA] 2016; Lusher *et al.* 2017; Wright and Kelly 2017; Gasperi *et al.* 2018; SAPEA 2019).

Based on diets in the United States, it is estimated that humans may be consuming anywhere from 39,000 to 52,000 microplastic particles a year via food. With added estimates of how many microplastics might be inhaled, the number increases to 74,000 and 121,000 (Cox *et al.* 2019). Individuals whose drinking water comes only from bottled sources may be ingesting an additional 90,000 microplastics annually compared to 4,000 ingested by those who drink only tap water. These estimates are subject to large variations and uncertainty; however, given methodological and data limitations they are likely to be underestimates (Cox *et al.* 2019) and represent only a fraction of future consumption rates if current trends of microplastic pollution continue.

Although exposure to microplastics through ingestion or inhalation could occur, the implications for human health and exposure levels are still not well known (Wang *et al.* 2019). Chemicals and microbial pathogens associated with microplastics in drinking water, along with the physical hazards associated with microplastics, have not yet been confirmed to be of high concern for human health (WHO 2019). More research is needed to better understand and assess the health risks of microplastics (Box 1).

Aquatic organisms throughout the food chain consume microplastics both directly and indirectly. As in humans, microplastics may pose physical, chemical or biological risks to other organisms. Within the food chain they have been found to have physical and chemical impacts resulting in starvation and reproductive consequences in some species (Wright *et al.* 2013).

Box 1. Priority research needed to better understand and assess the health risks of microplastics (adapted from Wang *et al.* 2019; WHO 2019)

- Development of standard methods for microplastics sampling and analysis;
- More studies on the occurrence and characteristics of microplastics using quality-assured methods to determine numbers, shapes, sizes, composition and sources;
- More data on the significance of treatment waste streams;
- Monitoring programmes on the abundance of microplastics in aquatic products intended for human consumption;
- Evaluation of the synthetic effects of microplastics and environmental toxicants, and identification of the role of microplastics in the trophic transfer of environmental contaminants;
- More studies to understand the role of microplastics as vectors for pathogenic microorganisms and potential ecological risks;
- Better understanding of the toxicological effects of microplastics following ingestion (relevant concentrations should be used in laboratory studies of microplastics exposure);
- More attention to the ecotoxicological effects of microplastics in higher order predators and freshwater organisms;
- More studies on factors that affect the selectivity of aquatic organisms for microplastics, and the toxicity and fate of ingested microplastics in aquatic organisms;
- Better understanding of the different health risks for men and women as a result of different exposure pathways to microplastics (e.g. through the use of PPCPs) or owing to gender differences in the proportion of body fat, which provides a greater reservoir for bio-accumulating and lipophilic (fat-loving) chemicals (Lynn *et al.* 2017).

Limited data from animal studies suggest that microplastics may accumulate and cause particle toxicity by inducing an immune response, while chemical toxicity could also occur due to leaching of additives and adsorbed toxins (Gasperi *et al.* 2018; SAPEA 2019). Ingestion of microfibres by zooplankton, benthic organisms and mussels can result in non-biodegradable gut blockage (Cole *et al.* 2013). Nevertheless, the impacts of environmental exposure to microplastics on mortality, morbidity and the reproductive success of species remain unclear.

Existing studies have mainly focused on possible harmful effects on aquatic fauna, while knowledge about impacts on aquatic primary producers, the trophic transfer process, and the implications for human health of consuming aquatic products remain uninvestigated (Wang *et al.* 2019). Looking to the future, Cole *et al.* 2011 presented a list of knowledge gaps that deserve further attention from the scientific community (Box 2).

Box 2. Microplastics in the aquatic environment: key research gaps (adapted from Cole *et al.* 2011)

- Employ a clear and standardized size definition of microplastics, with further size definitions for nano- and mesoplastics.
- Optimize and implement routine automated microplastic sampling methodologies to better compare results from different study areas.
- Develop appropriate methods to detect microplastics and nanoplastics within the water column and in sediments.
- Expand knowledge of the fate and behaviour of microplastics within the water column (e.g. in lakes), including the effects of fragmentation and biofouling.
- Develop methods to determine microplastic uptake by biota throughout the marine food web and expand the use of sentinel species (e.g. fulmars) to detect microplastic abundance.
- Determine the impacts (i.e. mortality, morbidity and/or reproduction impacts) of ingested microplastics on marine biota, and better understand the transfer of this contaminant within the food chain.
- Determine the impacts (i.e. mortality, morbidity and/or effects on reproduction) of leached plastic additives and waterborne pollutants adsorbed to biota, transferred via microplastics, on marine biota.

E. Macroplastics: a major challenge on their own

Plastic litter is not only challenging because it can fragment into microplastics. Macroplastics are a major challenge on their own. It is estimated that as of 2015, 60 per cent of the 8,300 Mt of plastic ever produced had been discarded and was accumulating in landfills, open dumps and the environment (Geyer, Jambeck and Law 2017). Part of this plastic finds its way to rivers, lakes and the ocean. If current consumption patterns and waste management practices (Notten 2019) do not change, it is estimated that by 2050 there will be approximately 12 billion metric tons of plastic litter in landfills, open dumps and the natural environment (Geyer, Jambeck and Law 2017).

Lebreton and Andrady (2019) calculated the amount of mismanaged plastic waste in 2015. The Asian continent was the leading generating region with 52 Mt, followed by Africa (17 Mt) and Latin America (7.9 Mt). Europe (3.5 Mt), North America (0.3 Mt) and Oceania (0.1 Mt) managed plastic more effectively despite the large amounts of waste generated in these regions.

As mentioned above, Jambeck *et al.* (2015) estimated that of 275 Mt of plastic waste generated in 2010, 4-8-12.7 Mt entered the oceans. This is comparable to the more modest and recent estimate of plastics loss into the environment from Ryberg *et al.* (2018) (i.e. 4.67 Mt). Plastic waste has serious environmental, health and economic impacts, including but not limited to:

- Blocking canals and sewers;
- Creating breeding habitats for mosquitoes (e.g. the dengue vector *Aedes* sp.);
- Damage to the airways and stomachs of animals;
- Loss of the landscape and touristic value of polluted beaches, lakes and rivers.

F. Objective and scope of the report

Water pollution by macroplastics and microplastics is complex and multidimensional. Managing it effectively requires a range of responses. Solutions need to address the design, production, consumption and disposal of plastics that will likely still be used in decades to come. These solutions should reduce pollution by macroplastics and microplastics at the source. Other responses need to 1) limit the export of macroplastics and microplastics from cities and the landscape through treatment of wastewater and run-off and adequate management of sewage sludge; 2) protect water bodies from pollution loads; and 3) restore affected water ecosystems and minimize the exposure of populations at risk. All these efforts must be supported by legislation, economic instruments, education and awareness that bring about real change on the ground.

A large number of potential solutions are available to address the plastic challenge. The enormous number and complexity of options make it difficult for policymakers and practitioners to set priorities and select those that are the most cost-effective and suitable for local contexts.

This report is the first attempt to review systematically some of the most relevant or promising technical solutions for microplastics from source to sea. When data are available, the report describes the effectiveness and costs (including capital expenditure and operation and maintenance [O&M] costs) of different solutions and discusses their suitability in different contexts. This assessment will help identify knowledge and data gaps and point to future research needs. Decision-makers can use the report as a compendium of solutions to choose from, as a starting point for implementing the best combination of instruments in their countries, local areas or river basins.

Throughout the report solutions have been clustered in groups that address the sources and pathways shown in Figure 3: technologies to prevent wastewater contamination at the source (Section II); technologies to treat wastewater and run-off before they arrive at the treatment plant (Section III); WWTP technologies (Section IV); technologies for treating contaminated sewage sludge (Section V); and technologies for treating freshwater (Section VI). Section VII outlines steps to follow in selecting a cost-effective combination of solutions adapted to a given local context.

Section II

Technologies to Prevent Wastewater Contamination at the Source

A. Macroplastics management at source

Several solutions can be explored to reduce macroplastics at source and prevent contamination of water, wastewater or the environment.

Recent information on the 67 plus countries which have adopted policy restrictions and bans on plastic bags and/or single-use plastic items (such as foamed plastic products made of Styrofoam [extruded polystyrene foam, XPS] or other plastics used for packaging) has been published by the United Nations Environment Programme [UNEP] (UNEP 2018) and Excell *et al.* (2020), along with the impacts of these interventions. Based on reported data, only in 60 per cent of cases have there been drastic drops in consumption of plastic bags within the first year (i.e. up to a 50-90 per cent reduction in plastic use) (UNEP 2018) while long-term impacts also appear to be variable.

It is generally accepted that enforcing bans to mitigate plastic pollution is the best solution that can be implemented by the public sector, although it may be expensive overall. A study (Marsden Jacob Associates 2016) for the city of Victoria, Australia,[3] found that the cost of implementation over a 10-year period was about USD 27 per inhabitant for a total ban scenario compared with USD 23-26 per inhabitant for the enforcement of levies and partial bans. However, the benefit-cost ratio was 1.28 for total bans versus 1.01-1.07 for the other scenarios. In addition, if the polluter-pays principle[4] is in place and implemented, this reduces required public investment (e.g. by assigning responsibility to manufacturers and consumers) (Prata 2018; He *et al.* 2018). Where the enforcement of bans has not been successful, reported challenges have included lack of awareness of the policy, low enforcement of the policy, lack of suitable and affordable alternatives that can be substituted for banned plastic-based products, and non-prohibitive levies (Gupta 2011).

Alternatively, measures to reduce dependence on or littering of plastic materials should also be put in place. For example, in early 2018, Indonesia deployed a military clean-up operation on the Citarum River, a vital source of water for 27 million people. With the river acting as a waste receptacle for households and industrial manufacturing plants, over 500,000 m^3 of trash (the equivalent of 200 Olympic swimming pools) flowed downstream each year (Tyler 2011). In the capital, Jakarta, over 4,000 workers were employed to remove litter from the surroundings of rivers and other water bodies (Tyler 2011). However, continued influx indicated that plastic use and waste management infrastructure were challenges requiring long-term strategies and that this problem needed to be addressed at the source (Tramoy *et al.* 2019).

Manual collection of plastics occurs elsewhere at scales large and small. Companies like the for-profit charitable organization 4Ocean prioritize local job creation via manual collection as a pillar of their plastic clean-up efforts (Benioff Ocean Initiative 2019). In August 2018, over 20,000 volunteers participated in a clean-up event at rivers and beaches all over Thailand. Smaller-scale or one-time efforts are under way in cities throughout the world, including in the United States and Europe (Benioff Ocean Initiative 2019). The scale of public involvement in clean-up events demonstrates that there is already significant awareness of and personal value identified in plastic pollution solutions.

This section discusses technologies that could be implemented to prevent or reduce contamination of the environment by macroplastics. In addition, as the large majority (typically 60 per cent or more) of microplastics found in water bodies are of secondary origin (i.e. from plastic degradation[5]) close monitoring of plastics usage will have direct positive impacts on the levels of microplastics in water (McKinsey and Company and the Ocean Conservancy 2015; Rhodes 2018; Magni *et al.* 2019). These solutions and technologies require:

- Enhancing plastic waste management globally;
- Supporting the informal plastic collection value chain and developing it towards a formalized mechanism;
- Implementing plastic recycling solutions.

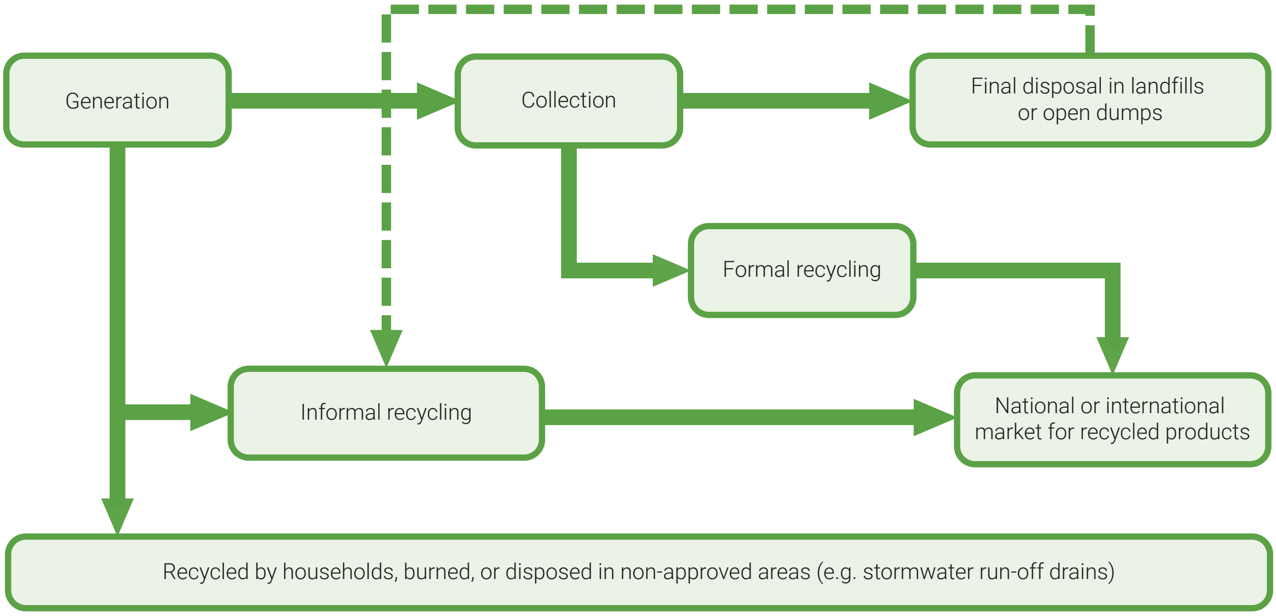
1. Enhancing plastic waste management to enable recycling

Waste management in most locations is an expensive, labour-intensive and low-margin business. Figure 5 illustrates the management scheme for solid wastes in many developing countries. It presents typical challenges along the service chain. Overall, a large share of the solid waste generated is inadequately managed. For example, up to 50 per cent of solid wastes (including plastics) in urban areas may not be collected because of factors including poor collection systems and road networks, equipment failure, and inadequate waste management budgets, often due to citizens' unwillingness to pay waste management charges (Cofie *et al.* 2016). Uncollected waste is burned, recycled informally or illegally dumped, to end-up on land or in run-off drainage channels connecting to rivers and wetlands, thus becoming a source of water contamination.

3 Victoria had a population of 5.88 million inhabitants in 2015 and an annual growth rate of 2 per cent.
4 The polluter-pays principle is based on the concept that those producing pollution should bear the costs of managing it in order to prevent harm to health and the environment.

5 Annex B presents plastic breakdown pathways in the environment. Annex C presents plastic breakdown pathways within landfills.

Figure 5. Typical waste management service chain in developing countries



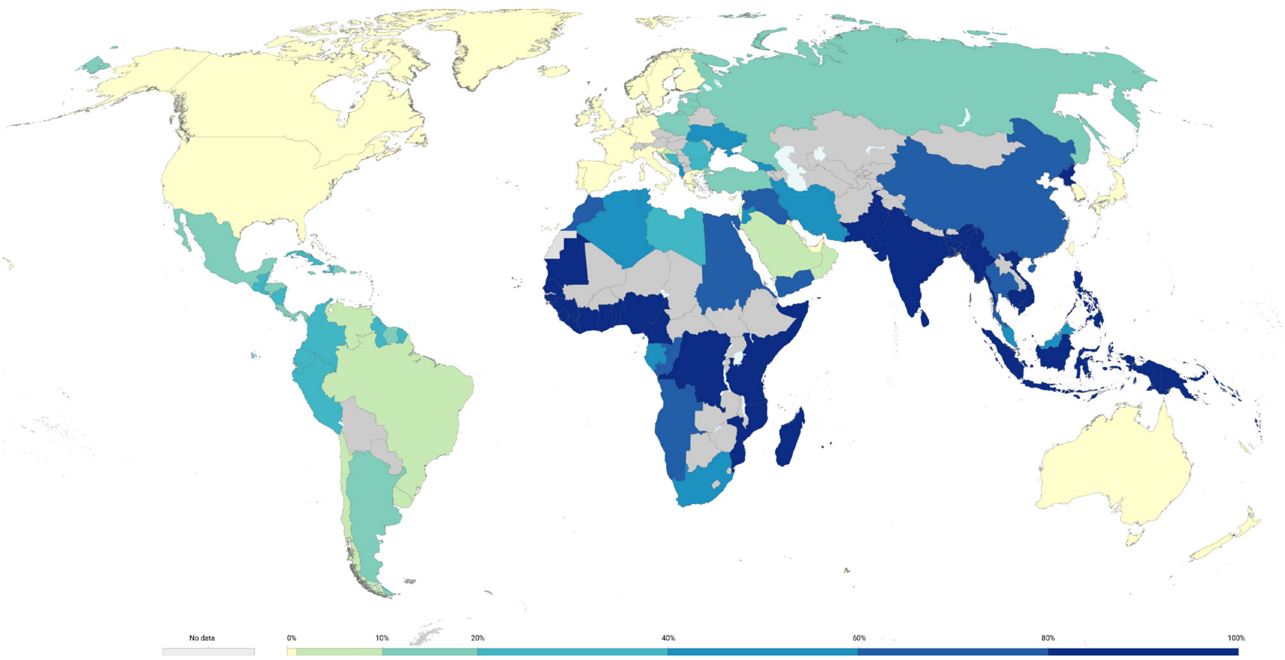
Households have a significant collective capacity to reduce the inflow of waste into the system, both through adapted consumption practices and waste management and recycling strategies. However, many households currently have little formal engagement with the waste sector's power and policy structures, and this has to change since they constitute the pivotal site for reform. Household needs and structures must be included in all waste management plans, and methodologies should be developed to assess the value of contributions to the protection of ecosystem services by women who manage waste in households and communities on an unpaid basis. As of yet, neither the social and monetary value of households' services, nor the unpaid labour of women managing waste within households has been measured or even officially acknowledged. In addition, the alienation of men and boys from domestic and community waste management activities has significant social and economic costs which will undermine any waste sector reforms if left unaddressed. Consideration of these factors will enable policies to be based on a more accurate view of the waste value chain, enhancing its sustainability and resilience (UNEP 2019).

Often waste management is viewed as an essential utility service governed by the public sector. However, it is frequently implemented in partnership with the private sector. Municipal solid waste (MSW) collection and transport is usually contracted to private companies. These companies operate, in principle, under the supervision of local authorities and technical line agencies. However, monitoring is often poor, leading to a focus by companies on profits rather effective performance. Consequently, collected waste is not better managed. Most of it is sent to open dumps. Even when engineered landfills exist, their maintenance could remain poor, leading to waste (including plastic) leakage.

Since there is no or limited segregation of waste at source by households, it becomes difficult to manage plastic waste (which represents 5-15 per cent of overall waste) in isolation from other waste streams (Ritchie and Roser 2018). Therefore, up to 70-85 per cent of plastic waste has been estimated to be mismanaged in Africa and Asia (Figure 6). Daily plastic waste generation per capita has been reported to range from 0.01 kg (in India) to 0.59 kg (in Guyana) (Jambeck *et al.* 2015).

Adequate management of plastic leakage is the first step towards controlling plastic pollution. It requires increasing waste recycling and ensuring the availability of suitable waste handling facilities (McKinsey and Company and the Ocean Conservancy 2015; Eriksen *et al.* 2018). Overall, collection, storage, transport, recycling and final disposal must be financially sustainable, technically feasible, socially and legally acceptable, and environmentally friendly. In both the public and private sectors men hold most upper-level administration roles, from city managers and planners to landfill operators and managers of waste collection companies. Women are more engaged in informal, household and neighbourhood activities related to waste, which are typically voluntary, unpaid or minimally compensated. Table 2 lists some specific actions that could be explored by key parties to improve overall plastic waste management.

Figure 6. Percentage of inadequately disposed plastic waste in the world in 2010



Reference: https://ourworldindata.org/grapher/inadequately-managed-plastic; Jambeck *et al.* (2015)

Table 2. Key actions needed by different stakeholders

<table>
<tr><th>Industry should</th><th>Government should</th><th>Citizens should</th></tr>
<tr><td>• Measure, monitor, manage and report plastic use
• Mitigate environmental risks and increase recycling of plastic products</td><td>• Enforce policies aimed at reducing per capita plastic waste generation, waste mismanagement and landfilling, and policies that promote recycling
• Promote tools that allow consumers (including women and other marginalized groups) to enhance their awareness of the management of plastic and plastic waste
• Openly support alternatives to plastic and encourage industries to move to environmentally friendly packaging
• Create/upgrade solid waste collection and treatment</td><td>• Make sound consumption decisions, e.g. to reduce or avoid plastic waste generation
• Change habits and lifestyles that require plastic usage, e.g. through reducing reliance on single-use plastics or through source separation</td></tr>
<tr><td colspan="2">• Governments/ the private sector should be encouraged to include households/ communities and specifically take affirmative action to ensure that women are invited to discussions as key stakeholders. It is crucial that governments and the private sector promote gender equal employment in the waste sector more actively.</td><td></td></tr>
</table>

References: Eriksen *et al.* (2018); UNEP 2019

McKinsey and Company and the Ocean Conservancy (2015) described 33 potential solutions through which leakage of plastics could be mitigated and modelled 21 of them for five countries: China, Indonesia, the Philippines, Thailand and Viet Nam. It was concluded that the best management initiatives would involve gasification, incineration, setting up of materials recycling facilities (MRFs) or improving haulier systems (although they are not always financially profitable).

2. Supporting informal plastic collection and the recycling value chain

In many developing countries waste recycling is nearly non-existent and is largely informal. Typically, informal waste recycling makes it possible to capture about 10 per cent of the plastic waste produced. It often involves poor and marginalized urban dwellers who resort to scavenging and waste picking for survival. In addition, when sorting at source takes place, the municipal waste collection crew collects sorted plastics which could be sent for further sorting and recycling.

There are at least three main categories of informal waste collectors (Gugssa 2012):

- Itinerant waste buyers who collect particular recyclable items from door to door;
- Street waste pickers who recover valuable plastics from communal bins;
- Waste pickers operating at mismanaged dumps or landfills before a daily or weekly cover is applied.

Roles in waste management are highly gendered, especially within the informal waste economy. This division of labour has implications both for women's opportunities to participate in the sector and for officials seeking ways to improve the system. Studies show that women are often limited to lower-income tasks, such as waste picking, sweeping and waste separation, whereas men are able to assume positions of higher authority, dealing with the buying and reselling of recyclables for example. Therefore, when informal or voluntary waste-related activities become formalized with pay, men often engage in the work, thereby displacing women (UNEP 2019). For example, in Ho Chi Minh City, Viet Nam, one study found this structure:

- Door-to-door itinerant buyers (entirely women) who buy solid waste products from households;
- A range of small, medium and large shopkeepers (men) who purchase waste from the buyers;
- Middlemen who link the shopkeepers with the recyclers;
- Recycling or production units run by men that transform products for sale to consumers.

There are also challenges for women informally collecting waste in landfills. Typically, landfill operators and on-site supervisors are men (UNEP 2019).

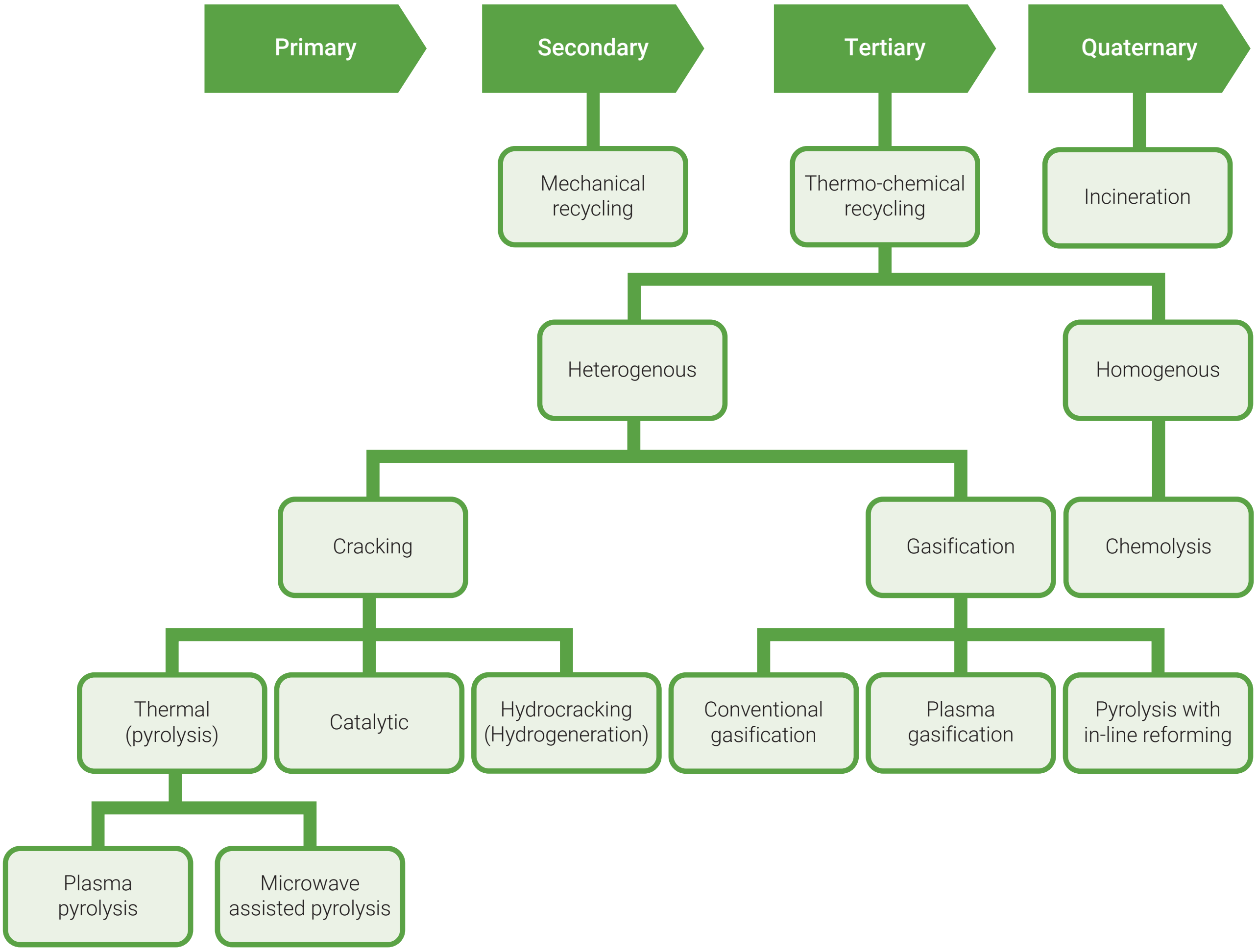
Informally collected plastics are either recycled within the country or exported by private companies to Asia (e.g. Thailand, Viet Nam). Until 2016, China was importing 7-9 Mt of plastic wastes per year.

The viability of informal plastic collection may be questionable in many countries. The informal, and sometimes illegal, nature of this activity makes it difficult to support. However, in particular cases it may be possible to build on these chains to drive the plastic collection sector from an informal and illegal basis to a formal one, with practices and protocols which are comparatively easy to monitor, helping to safeguard the health and livelihoods of informal waste collectors.

3. Implementing plastic recycling technologies

Figure 7 shows the technologies available for plastic waste recycling. Primary recycling is the mechanical reprocessing of a single type of uncontaminated plastic. It produces plastic material of equivalent quality. This could be appropriate for recycling sorted pre-consumer waste, but it is unsuitable for municipal solid waste (MSW) management. Secondary recycling is mechanical recycling that downgrades the recycled material. Tertiary recycling is thermo-chemical recycling (also known as "feedstock recycling") which breaks down polymers into monomers or simpler molecules, later used in producing energy or virgin and recycled materials. Quaternary recycling is waste incineration for energy recovery (Solis and Silveira 2020).

Figure 7. Routes for recycling of solid plastic waste



Reference: Solis and Silveira (2020)

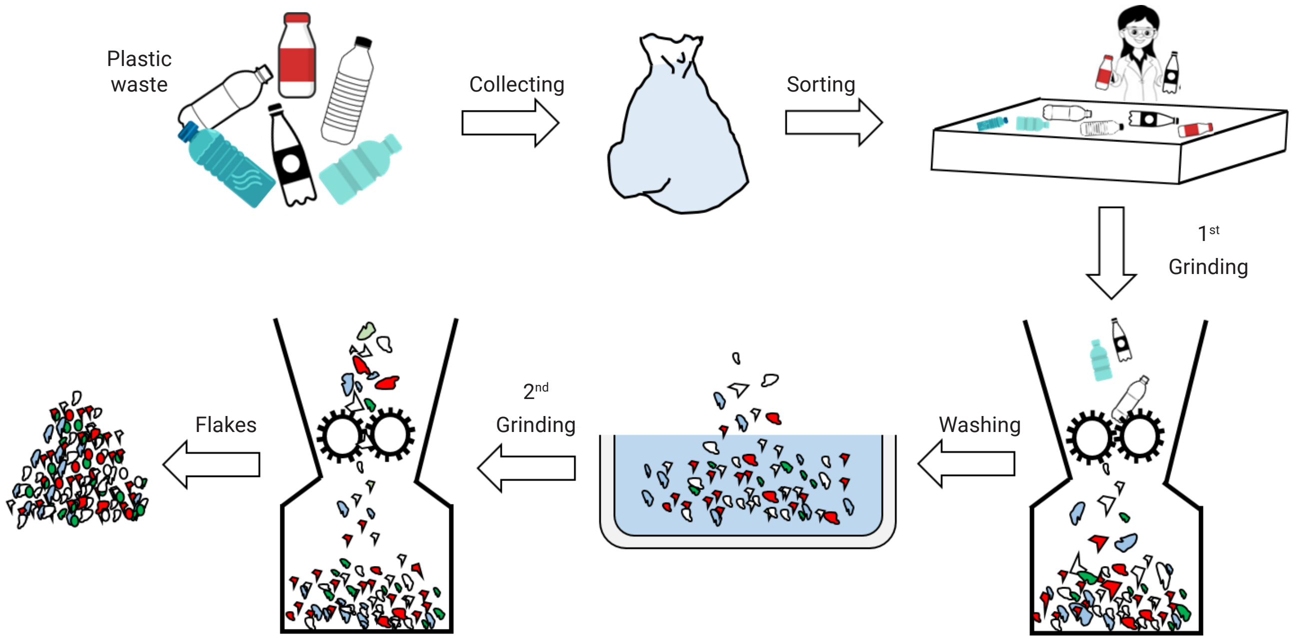
Circular economy models offer a number of environmental benefits such as increased resource efficiency, decreased greenhouse gas emissions, reduction in toxicity risks to human and ecosystem health, and protection of biodiversity and ecosystem services. There are also socio-economic benefits associated with the reduction in ocean plastics-induced loss of marine natural capital, increased efficiency in the informal waste recycling sector and the development of novel livelihoods in circular plastics economy (Wang, Talaue McManus and Xie 2019).

These technologies come with some health risks to workers, communities and ecosystems associated with exposure to contaminants, whether intentionally or unintentionally generated, and this has a gendered dimension. Indeed, it is well known that women and men are exposed differently to hazards, e.g. in the workplace, due to biological gender differences such as body size, amount of adipose tissue, reproductive organs or hormones that can impact the effects and elimination of toxic chemicals and substances. Recently, a Canadian study found that women working in the plastics industry had a five-fold elevated risk for breast cancer and reproductive disorders (Brophy *et al.* 2012). Research on gendered health impacts is scarce and there is need to encourage more scientific work in this area. There is globally a considerable gap in knowledge about the health effects on men and women working in the plastic industry and plastic waste management. The evidence concerning health risks should be investigated and addressed, particularly in less wealthy countries, beginning with implementing existing health and safety legislation. The availability of sex-disaggregated data will support the adoption of the necessary policies for adequate safeguards.

a. Mechanical recycling (secondary recycling)

This technology refers to the processing of plastic waste into a raw material or product without significantly changing the chemical structure. It works well for all types of thermoplastics (as opposed to thermoset plastics), i.e. those made up of linear molecular chains that soften when heated and harden when cooled. There are three types of thermoplastic polymers:

Figure 8. Process leading to mechanical recycling of plastic waste



Reference: Delva *et al.* (2019)

- Crystalline thermoplastics (e.g. PP, LDPE, HDPE, PET);
- Amorphous thermoplastics (e.g. PVC, PMMA, PC, PS, ABS);
- Semi-crystalline polymers, which combine properties of the first two types and include polyester polybutylene terephthalate (PBT) and polyamide Imide (PAI).

Mechanical recycling constitutes the key form of recycling worldwide. In Europe, 99 per cent of the volume of recycled plastics undergoes such a process. It is particularly suited for recycling clean plastic waste with a single composition. This recycling process includes the following steps: collection and sorting, washing, grinding and drying (Figure 8). Granulating and compounding may follow eventually (Ragaert *et al.* 2017). Mechanical recycling is mostly used in recycling PP, PE and PET.

For financial viability it is better to process large volumes of plastic waste (Hopewell *et al.* 2009). In addition, high purity sorting of plastic is necessary to ensure a high quality output. This may cause high material rejection rates.

Mechanical recycling optimizes the use of plastic resources and extends their lifespan (Hopewell *et al.* 2009; Lam *et al.* 2018). However, thermo-mechanical degradation of plastic polymers is observed during mechanical recycling. Other quality degradation may also be observed as a result of exposure to natural light (photodegradation), oxygen or moisture (biological degradation). Another main issue is the presence of additives, fillers or even other polymers in the original plastic; these are hard to recycle, resulting in contamination of the mixture and downgrading of the recycled output quality (Ragaert *et al.* 2017).

b. Feedstock recycling (tertiary recycling)

Plastics and plastic-containing waste which, for health, environmental and economic reasons, cannot be recycled to the required quality standard mechanically provide a valuable input resource for feedstock recycling. Feedstock recycling (also known as “chemical recycling”) is a tertiary recycling method which offers an opportunity to recover more waste than primary or secondary recycling because it allows the breakdown of types of plastic waste usually sent to landfill or incinerated (e.g. shopping and trash bags, retail packaging, food wraps, bubble wrap, carpet fibres and other plastic material). Consequently, higher volumes of plastic waste can be processed.

To achieve tertiary recycling, the chemical structure of plastic waste is transformed through thermo-induced chemical or biochemical reactions into shorter molecules which are readily usable in order to manufacture new products such as fuels, chemicals or virgin plastics. Examples of processes include gasification and pyrolysis

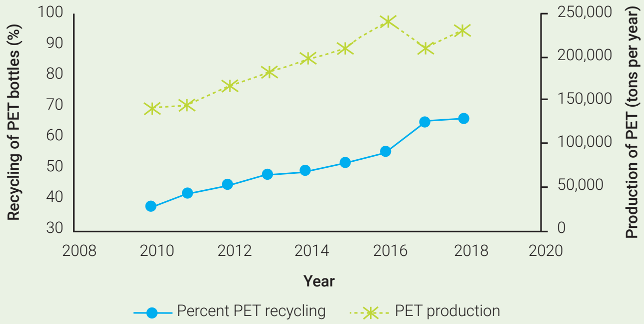
Box 3. When industries forefront post-consumer PET (bottle-to-bottle) recycling PETCO (2018)

PETCO is the trading name of the not-for-profit PET Recycling Company NPC South Africa, incorporated in 2004. It is an industry driven and financed environmental solution for post-consumer polyethylene terephthalate (PET) recycling. The initiative is funded through a voluntary extended producer responsibility (EPR) fee paid by bottle manufacturers which purchase PET resin.

This industrial voluntary engagement helps ensure that the environmental impact of used bottles is minimized while creating jobs and positively contributing to South Africa's economy. New PET packaging can be made of up to 100 per cent recycled PET, recapturing both the material and the energy inherent in the original package. It can also be recycled multiple times. In the last decade recycling of PET bottles (Figure 9) has yielded multiple benefits.

- Recycling a single metric ton of plastic bottles saves 1.5 metric tons of carbon;
- About 2.7 million m^3 of landfill space has been saved;
- Natural resource consumption has been reduced.

Figure 9. PET bottle recycling in South Africa

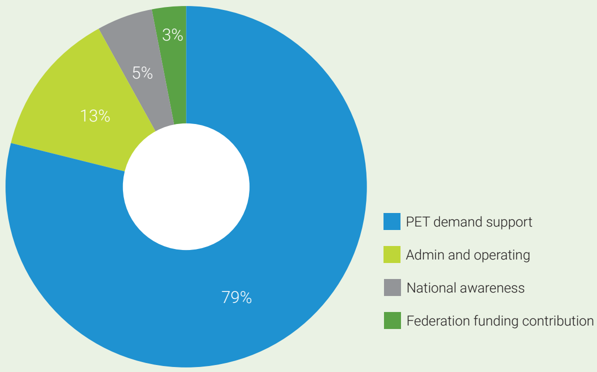


PETCO (2018)

Some factors contributing to the success of this initiative include:

- High awareness of the relevance of recycling among industries and consumers/populations;
- Existence of a clear financing stream, which has enabled equipment support and sponsorship for waste collectors;
- A capacity-building programme implemented in parallel with this initiative, involving key stakeholders, such as municipalities;
- Capacity to source and operate adequate technologies for recycling.

Figure 10. Cost distribution of PETCO operations



PETCO (2018)

The costs of recycling PET in South Africa were about USD 76.51 per metric ton in 2018. The distribution of these costs is shown in Figure 10.

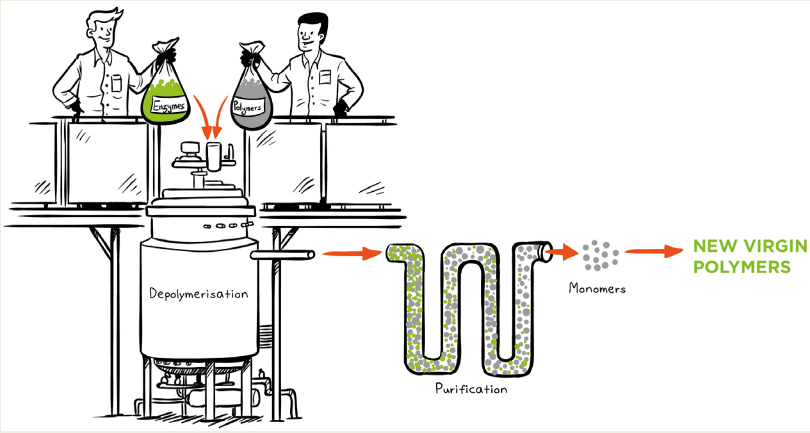
(Table 3), through which plastic waste breaks down to produce synthesis gas (syngas) and oil (fuel), among others (Lam *et al.* 2018). Three technical factors critically affect the cost and attractiveness of chemical recycling of plastic: the process temperature (or energy consumption); the type of plastic feedstock and its level of contamination (particularly how it affects the proposed technical process); and the level of polymer breakdown desired (Solis and Silveira 2020).

In Europe gasification is employed to process plastic waste in blast furnaces into syngas and recover metals (PlasticsEurope 2020). However, this process requires high investment costs, high energy consumption and high input levels, so that that only very large plants (i.e. those able to process over 100,000 metric tons per year) are economically viable (Ragaert *et al.* 2017; Solis and Silveira 2020).

Box 4. Biorecycling by Carbios, France

The CARBIOS technology (Figure 11) targets polyesters such as PET, PA and PLA. It relies on enzymes to depolymerize the plastics. To achieve this, sorted and cleaned plastics are mixed with water and enzymes, heated up and churned. The enzymes decompose the plastic into molecules serving as basic building blocks, which can then be separated, purified, and used to make virgin plastic. With this process there is no loss of quality in the recycled product.

Figure 11. The Carbios technology



Reference: CARBIOS (2020)

According to the company, this technology could be suitable to treat the 1 million metric tons of PET food containers per year in Europe which are not currently recycled since the trays are contaminated with food, while the structure of the plastic means cannot easily be recycled into the form used to make plastic bottles (Koop 2019).

In 2019 Carbios raised 16 million euros to finance construction of a first pilot plant, due to be commissioned by late 2020. The company expects the first industrial plant to be operational from 2023.

Table 3. Comparison of technologies for chemical or tertiary recycling of plastics

Technology	Process outputs	Operating conditions	Benefits	Limits	Scale of operation/ example
Pyrolysis (conventional thermal cracking)	Oil, gas and char[a]	• Key parameters are: temperature (450-700°C), pressure, residence time, catalysts, heating rate and absence of oxygen • Sensitivity to feedstock quality: Medium to high	• Simple and flexible (i.e. it allows varying operating parameters to optimize yields) • Oil is often the most desired product. It has a good calorific value and many applications (e.g. in petroleum blends) after further upgrading	• Process reactions are complex and not fully predictable • High energy requirement • Low tolerance to the presence of certain plastics (e.g. PVC types) • Sensitive to feedstock contamination • Oil often needs upgrading before use	Commercial scale: Mogami-Kiko, Japan, 3 metric tons/day
Plasma pyrolysis	Syngas (a mixture of carbon monoxide and hydrogen) and trace amounts of hydrocarbons	• Temperatures: 1,730-9,730°C • Reaction time: 0.01-0.5 second (based on process temperature and type of waste) • Sensitivity to feedstock quality: low	• Achieves total polymer breakdown • Syngas has low tar content and high heating value. It is used to generate electricity in turbines or hydrogen. • Low emission levels • Forms less free chlorine from hydrogen chloride than other pyrolysis processes	• Well-established technology for metallurgy processing, material synthesis and hazardous waste destruction, but not for plastic waste recycling • High electricity requirement	Laboratory scale

Technology	Process outputs	Operating conditions	Benefits	Limits	Scale of operation/ example
Microwave-assisted pyrolysis	Oil, gas and char	• High conversion efficiencies • Temperature: up to 1,000°C • Sensitivity to feedstock quality: medium	• Even heat distribution • Suitable to treat all solid wastes, including plastics • Compared with conventional pyrolysis: - higher heating rates - better process control - high production speed	• Large fluctuations in waste composition could be a challenge • There are currently knowledge gaps about process efficiency and performance • Technology is not yet commercially feasible • Requires large feedstock volumes to be feasible	Laboratory and pilot scale
Catalytic pyrolysis	Oil, gas and char	• Process temperature: 300-550°C • Reaction time: five minutes • Examples of catalysts: zeolites, metal catalysts, fluid catalytic cracking catalysts • Sensitivity to feedstock quality: medium to high	• Compared with conventional pyrolysis: - catalyst helps optimize product distribution and selectivity, leading to increased oil yield (up to 86-92 per cent) and quality - lower operating temperature (less energy consumed) - shorter reaction time - reduced production cost	• Most tests were done with pure polymers since the process may be affected by contaminants in the mixed waste plastic stream • Chloride and nitrogen components in wastes can deactivate the catalyst • Pretreatment of wastes is required to minimize clogging of catalyst's pores	Commercial scale: Sapporo/Toshiba in Japan processes 14,800 metric tons of mixed plastic waste per year
Hydrocracking	Valuable products are jet fuel, gasoline and liquid petroleum gas. Coke is also produced.	• Involves adding hydrogen to the pyrolysis (cracking) process to increase output quality • Hydrogen pressure: 20-150 (e.g. 70 atmospheres) • Temperature: 375-500°C • Sensitivity to feedstock quality: high	• The plastic waste first undergoes low temperature pyrolysis. The liquid is then sent to a catalyst bed • The catalyst reduces the reaction temperature and increases the oil yield and quality	• The cost of hydrogen is high (typically 2,500 euros per metric ton) • Issues with catalyst deactivation when treating PVC • High investment and operating costs	Pilot scale

Technology	Process outputs	Operating conditions	Benefits	Limits	Scale of operation/ example
Conventional gasification	Mixture of hydrocarbons and syngas, tar[c] and char (the last two are less desirable)	• Temperature: 700-1,200°C • Process time: < 10 seconds • Gasifying agents are air (low quality syngas) and oxygen, steam. • Sensitivity to feedstock quality: Low to medium	• The syngas is used to produce energy, energy carriers such as hydrogen and methane, and chemicals • The steam or oxygen gasification results in syngas with high heating value and high hydrogen concentration. It can be used to produce new plastic products • Suitable for mixed plastic waste processing • Well-established technology	• The gasifying agent determines the composition of the syngas produced and its applications • Syngas quality with plastics reduced due to high tar content. It must be cleaned before use • Requires high feedstock volumes to be feasible • Can be costly and energy-intensive	Commercial scale: Enerkem (Edmonton, Canada): 100,000 metric tons of plastic wastes per year

Plasma gasification	Organic and inorganic matter in the feedstock is converted into syngas and slag,[b] respectively	• Atmospheric pressure
		• Temperature: 1,200-5,000°C (exceptionally up to 15,000°C)
		• Residence time: less than a few minutes
		• Sensitivity to feedstock quality: low
			• Process is not impacted by feedstock quality fluctuations
			• Higher purity of product gas, with reduced level of tars.
			• Compared to conventional gasification:
			  - higher purity of product gas
			  - reduced level of tars
			• Well-established technology for other applications
				• High electricity requirement (15-20 per cent of gross power output)
				• Higher operating costs and larger investments
				• Some industrial plants are in operation in Asia and Europe in other sectors
					Laboratory and pilot scale

Pyrolysis with in-line reforming	Syngas, carbon monoxide, methane and hydrocarbons, tar and char (the last two are less desirable)	• Uses two reactors, connected in series, for pyrolysis and reforming
		• Nickel catalyst can be used for reforming
		• Temperature: 500-900°C depending on the feedstock, reactor configuration and bed material
		• Sensitivity to feedstock quality: medium
			• In comparison with conventional gasification:
			  - lower process temperature
			  - lower production cost
			  - higher hydrogen production from the process (typically 30 per cent more)
			• Syngas is free of tars
				• Lower risk of catalyst deactivation, but this needs more research
				• Current studies limited to laboratory scale
					Pilot scale

References: Garforth *et al.* (2013); Munir *et al.* (2018); Solis and Silveira (2020)

a Char is a solid residue composed of unreacted carbon and ash.
b Slag is an inert glass-like material obtained from the melting of heavy metals.
c Tar is a dark, thick flammable liquid.

c. Energy recovery from plastics through incineration

The quaternary recycling of plastics shortens the material's lifespan. Although it is used in many countries, quaternary recycling is viewed as a non-sustainable solution which is not fully aligned with the evolving principles of a circular economy (Solis and Silveira 2020). Plastic waste has a notable potential for energy generation because the calorific value of plastic is similar to that of hydrocarbon-based fuel (Sun *et al.* 2018). Hence, high energy is released from plastics following incineration of municipal solid waste. When mixed with other organic wastes, the presence of plastic usually increases the calorific value of the waste mixture, making it suitable for combustion (Lam *et al.* 2018). Typically, 70-80 per cent of the energy from waste incineration can be recovered to produce hot water only. If the interest is in electricity only, the energy recovery is 20-25 per cent. In the case of co-generation, both electricity (same amount as earlier) and hot water are produced, for a total energy recovery of 50-60 per cent for both outputs (Planete Energies 2014; Gradus *et al.* 2016). Electricity generation is typically 0.40-0.77 megawatt hour (MWh) per metric ton of input municipal solid waste containing plastics (JFE Engineering Corporation 2018; Tullo 2018). Some microplastics (typically 1.9–565 particles per kg of ash formed) are found in the ashes resulting from the

process. These ashes represent 10-25 per cent of the input mass (Yang *et al.* 2021).

Other innovative technical options for recycling plastic waste include co-processing of plastic waste in cement kilns or foundries (McKinsey and Company and the Ocean Conservancy 2015).

Box 5. Comparing mechanical recycling and incineration of plastics in the Netherlands

In the Netherlands 24 and 27 per cent of municipal waste was recycled and composted, respectively. Most of the remainder was incinerated, allowing energy recovery (mostly in the form of electricity). Incineration facilities in the Netherlands are among the most efficient in the world, with high per cent energy recovery and competitive gate fees. The benefits, limitations and drivers of each process are shown in Table 4.

Table 4. Recycling and incineration in the Netherlands: benefits, limitations and drivers

Solutions	Expected benefits	Foreseen limits	Drivers
Recycling of plastics to produce new plastics for high-quality industrial purposes	• Avoidance of carbon dioxide (CO_2) that would otherwise be emitted during incineration • Production of (new) material	• High collection and recycling costs	• Environmental awareness • Local policy promotes incineration and recycling
Incineration of plastics for energy recovery	• Heat and electricity production leading to fewer emissions in the regular energy production sector • No sorting required	• Requires waste-to-energy plant with associated high capital investments	• Lack of space • Local policy that promotes incineration and recycling

Reference: Gradus *et al.* (2016)

A comparison of net costs in euros per metric ton of plastic, and of net CO_2 emissions (metric tons of CO_2) per metric ton of plastic, for both recycling and incineration, are shown in Table 5. The cost difference is 199 euros per metric ton of plastic in favour of incineration, while the difference in CO_2 emissions is 1.16 metric ton per metric ton of plastic in favour of recycling. There is therefore a trade-off between incineration and recycling. However, the authors concluded that incineration is preferable to recycling when the market value of CO_2 is below 68-172 euros per metric ton.

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Table 5. Net costs of recycling and incineration (euros per metric ton of plastic) and CO_2 emissions from recycling and incineration (metric tons of CO_2 per metric ton of plastic)

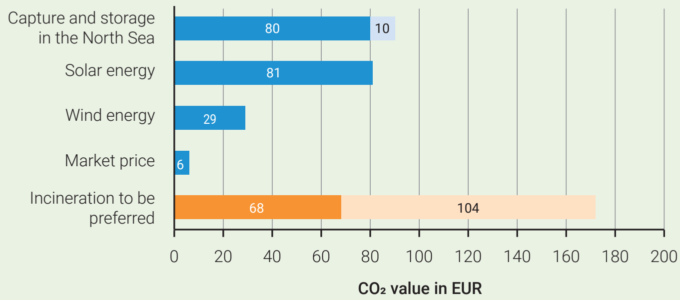
Item	Mechanical recycling		Incineration	
	Costs (euros)	CO_2 emissions	Costs (euros)	CO_2 emissions
Collection and transport	408	0.02	60	0.01
Net – treatment	262	0.85	6	2.6
Opportunity energy production[a]	90	0.78	0	0
Opportunity plastic recycling[a]	0	0	495	0.20
Total	**760**	**1.66**	**561**	**2.82**

Reference: Gradus *et al.* (2016)

a Opportunity cost/emissions means cost/emissions associated with the loss of other alternatives when one alternative is chosen.

In 2016 the market price for CO_2 emissions in the European Emissions Trading System was approximately 6 euros per metric ton. The costs of reducing 1 metric ton of CO_2 of different origins were between 29 euros for wind energy and 90 euros per metric ton of CO_2 for carbon capture and storage in the North Sea (Figure 12). In all these cases, recycling in the Netherlands appeared to be less interesting economically than incineration.

Figure 12. Minimum and maximum value of carbon dioxide (CO_2) in euros



Reference: Gradus *et al.* (2016)

4. Cost comparison

Table 6 presents a compilation of typical capital costs of setting up a plant with the equivalent of 1 metric ton/day capacity. They range from USD 2,000 to USD 10,000 for mechanical recycling to USD 857,000 for chemical recycling. The life cycle would also be different (higher in the case of highly engineered systems). Accordingly, O&M costs vary depending on the technology considered, from as low as USD 500 (including costs of acquiring sorted and clean materials) in a country such as India, in the case of mechanical recycling, to several thousand dollars for gasification or incineration in Europe or the United States.

The obvious conclusion is that mechanical recycling of plastics is more affordable to establish and operate in developing countries because of the cheap labour available to collect, sort and clean plastics. However, in countries with different characteristics (e.g. in Europe) mechanical recycling could be as expensive, or even more expensive than incineration (Gradus *et al.* 2016). In addition, mechanical recycling is highly dependent on feedstock quality and requires strong policy support to ensure the availability of quality plastic waste. A prerequisite for the use of such technologies is enhancement, where applicable, of the collection of solid waste and plastics. The costs of doing this are highly dependent on context, but could typically require increasing collection fees by five to ten times (McKinsey and Company and the Ocean Conservancy 2015; JFE Engineering Corporation 2018).

There are also costs arising from adverse impacts on human health of the adoption of plastics recycling technologies. Pollution deteriorates health, leading to more medical treatment and greater expense', which also has implications with regard to employee sick days and productivity levels (let alone disability/mortality concerns, which represent the highest cost to pay for victims and their families). There are also gender impacts, e.g. increasing numbers of female-headed households. In the case the cost of putting the necessary safeguards in place is often less than that of doing nothing. The lack of disaggregated data in this regard is a gap that needs to be filled (Lynn *et al.* 2017).

Table 6. Costs of technologies used to prevent municipal wastewater contamination

	Capital costs to process 1 metric ton/day capacity	Annual O&M costs to process 1 metric ton/ day capacity	Profitability	References
Mechanical recycling	From USD 2,000 to USD 10,000	Typically USD 500 to 1,500 (in India, including cost to acquire raw plastics)	The plant can become profitable in less than a year if value chains for quality plastic collection and diversification of products and revenues are achieved.	MakeInBusiness 2018
Chemical recycling (pyrolysis)	USD 857,000 Typical output is 224 m^3 per metric ton/day of diesel and naphtha and 73 m^3 of industrial wax	USD 500-1,000	This recycling mode will not be attractive when oil prices are low, e.g. less than USD 100 per barrel. The business is only profitable when large volumes can be processed (50,000-100,000 metric tons/year).	Homolka 2018; Porcu *et al.* 2019; Taullo 2019
Chemical recycling (gasification)	USD 385,000	Labour: USD 4,250 Maintenance: USD 18,100	If energy recovery is carried out, yield is 43.5 megajoules (MJ)/ kg of plastic; hence, an estimated revenue of USD 286/metric ton of plastic. Typically, if hydrogen is purified and marketed, revenue is USD 197/metric ton of plastic. Although the plant can break even, profitability is reduced and net present value (NPV) remains negative even after 15 years of operation.	
Incineration	USD 260,000-550,000 Typical output per year is 269 megawatt hours (MWh) of energy per metric ton/day of waste.	USD 10,800-40,000 (i.e. 56 per cent for maintenance and management, 11 per cent for personnel and 25 per cent for utilities)	In typical developing country, plant is not profitable because it would require five to nine times higher tipping fees, which cannot be implemented in the local context. The situation is different in Europe.	Turchet 2015; Gradus *et al.* 2016; JFE Engineering Corporation 2018; Tullo 2018
Enhancing plastic waste management	Variable, depending on adopted solution	Current investment for O&M in developing countries is insufficient and should be increased considerably (e.g. a several times increase is required for some Asian countries)	This step is necessary for recycling to occur. It is usually not expected that profitability will be reached, but the target is cost recovery.	McKinsey and Company and the Ocean Conservancy 2015

B. Microplastics management at source

Control of microplastics at source would have a direct effect on their release into water bodies and wastewater streams (De Gueldre 2020). To the extent possible, such solutions are therefore preferable. They can also be less expensive to implement than treatment options.

This section discusses solutions that could prevent contamination of water, wastewater and the rest of the environment by microplastics. They include:

- Treatment units that can be used in households and laundromats to remove microplastics from effluents;

- Design of new textiles to reduce the generation of microfibres during washing;
- Policy tools to reduce the use and misuse of microbeads;
- Behavioural change campaigns to reduce the use of microbeads and generation of microfibres at source.

Table 7 shows solutions that could be implemented with respect to specific pollution sources.

Table 7. Sources, measurements and strategies for mitigation of microplastics upstream of water bodies

Sector of activity	Source of plastics and MPs (plastic microfragments, MBs and MFs)	Potential mitigation	Stage of implementation of proposed solution[a]	Remarks	Report section
Production	MBs in cosmetics	Replace MBs with benign alternatives	+++	Several countries have adopted bans on use of MBs in rinse-off PCCPs	II-B3
	Mismanaged MBs in preproduction	Regulate and control pellet handling	++	In most countries there are no specific regulations for management of MBs	II-B3
Commercial	Industrial abrasives (MPs)	• Replace MBs with alternatives • Improve containment and recovery	++	These solutions could be easily explored if regulation integrates these measures	II-B3
	Laundromat exhaust (MFs)	Improve filtration	++		II-B1
Consumption	Tyre dust (MPs)	Technological advances in the design of tyre and road surfaces	+	As pollution with MPs is not yet high on the agenda, solutions of this type are only explored for research purposes. Recently launched initiatives attempt to establish standards to enable improved tyre quality control	III-B
	Fabric use (MFs)	• Choice of single-fibre woven textiles • Adoption of coated textiles	++	Some scientific research on this subject has been done, but there is not yet a strong marketing argument	II-B2/4
	Domestic laundry wastewater effluent (MFs)	• Washing with front-loading (not top-loading) machines • Adequate wastewater containment	++	No control is carried out at this level. Use of filters for each machine could easily be explored if regulations included this requirement	II-B1/4

Reference: Eriksen *et al.* (2018)

a Note: (+) Solution at concept stage; (++) Solution at research stage without any real-life applications; (+++) Solution implemented in some countries.

1. Treatment units for treating pollution at source

a. Washing machine filters

Around 35 per cent of microplastics in the oceans are believed to originate from the washing of synthetic textiles, which releases fibres into the water. A single garment can release between 1,900 and 1,000,000 fibres, with typical average dimensions of 5.0-7.8 mm (11.9-17.7 μm in diameter) (Napper and Thompson 2016; Prata 2018; Yang *et al.* 2019). The amount of microfibres released is related to the type of fabric (e.g. PE fabrics release 8.6 times more microfibres than acrylic ones) and its weathering, but also to washing conditions[6] (temperature, friction, velocity, washing time, detergent used, presence or not of softener, amount of water used) (Napper and Thompson 2016; Salvador Cesa *et al.* 2017; De Falco *et al.* 2017; Prata 2018; Yang *et al.* 2019; Hanning 2020). In addition, more fibres are released after the first wash than during subsequent ones. This is associated with fabric construction, but is also dependent on circumstances during, for example, production and transport (Hanning 2020).

On the other hand, over 840 million domestic washing machines are operated worldwide, using 55 million m^3 of water per day (Salvador Cesa *et al.* 2017). With the projected numbers continuously rising, it is essential to explore solutions to treat the likely contaminated wastewater effluents emerging from these units.[7] One way forward is to develop household-based systems to treat wastewater and microplastics. This approach could be applied to wastewater from washing machines or, more generally, to grey wastewater[8] treatment. It would prevent microplastics being released into sewer lines or the environment (Sun *et al.* 2018). For instance, a filter placed at the washing machine drain will retain microfibres released during washing (Prata 2018).

Some private companies are marketing filters for household washing machines. A typical filter costs 9.95 euros per month for a household (Table 8) and can retain up to 90 per cent of the fibres generated during washing. It is designed for domestic and commercial washing machines, whether already in existence or newly developed, that have a wash load of less than 30 kg and must be replaced monthly (Kržan and Zupan 2020). Filters can be regenerated, and retained fibres are considered for recycling. Aquafin has established that consumer end-of-pipe filtration techniques may not be ideal due to their high cost (which they estimated at 0.08-0.20 euros/m^3) and uncertain performance (De Gueldre 2020).

b. Laundromat effluent treatment

After laundry is cleaned, water becomes polluted to the point that it may not be suitable for discharge into municipal sewers. The composition of wastewater effluent depends mostly on the washing machine and its use, as shown in Table 9.

Based on a study in Sweden, it was established that microplastic concentrations in wastewater effluents originating from such facilities vary depending on the sector, typically from 3,000 to 460,000 microplastics per litre (Brodin *et al.* 2018). Each washing releases 1,000-

Table 8. Costs of technologies used to prevent municipal wastewater contamination

<table>
<tr><th></th><th>Investment cost</th><th>Annual O&M</th><th>Profitability</th><th>References</th></tr>
<tr><td>Household washing machine filters</td><td>None</td><td>USD 131 per household</td><td>Not available</td><td>Kržan and Zupan 2020</td></tr>
<tr><td rowspan="2">Laundromat effluent treatment</td><td>USD 5,000-40,000 per unit (the process is typically operated in batch mode or needs a storage tank)</td><td>Can be high due to energy demand and use of chemicals in the process</td><td rowspan="2">Water can be recycled, leading to some cost savings</td><td rowspan="2">Ahmad and EL-Dessouky 2008; Jafarinejad 2017; Swartz et al. 2017</td></tr>
<tr><td colspan="2">Typically, USD 706/m³ of wastewater treated for both capital costs and O&M. The process was a simple sedimentation and filtration combination. Costs should be higher for conventional treatment based on physical-chemical processes.</td></tr>
</table>

6 There is no information available about the impact of manual washing, which is common in developing countries, on the release of microfibres compared with the use of washing machines.

7 It is understood that the solutions discussed in Section IV.C may not be sufficient to prevent the release of microplastics from washing machines.

8 Grey wastewater is the mixture of all wastewater streams generated in households or office buildings that exclude toilets waste.

Table 9. Composition of laundromat wastewater effluent

	Laundromat (commercial laundry)	Industrial laundry
Water consumption (litres/kg cloth)	15	20-30
pH	7-11	10
Temperature	38°C	45°C
Chemical oxygen demand (COD) (mg/litre)	5,000-18,000	8,000-12,000
Biochemical oxygen demand (BOD) (mg/litre)	250-500	5,000-7,000
Suspended solids (SS) (mg/litre)	400-1,200	1,500-2,000
Grease (mg/litre)	400-600	1,500-32,000
Surfactants (mg/litre)	50-80	100-600
Phosphate (PO_4) (mg/litre)	250-300	300-2,000

References: AZU Water (2015); Swartz *et al.* (2017)

Note: Commercial laundries often work in self-service mode, while industrial laundries usually specialize in providing services to users such as hotels, restaurants, hospitals and nursing homes.

500,000 fibres per kg of fabric, depending on the sector (Table 10).

Some technologies exist to treat effluents from industrial laundries. The focus has not been on the removal of microplastics, but on removal of, for example, oils and suspended solids (SS) (Fijan *et al.* 2008). Typical technologies for treating this wastewater mostly rely on physical-chemical processes such as precipitation/coagulation and flocculation (for SS and colloids), adsorption on granular-activated carbon (GAC), or oxidation with ozone, UV, chlorination and peroxides to remove organic pollutants and possibly also membrane filtration (e.g. ultrafiltration and reverse osmosis) to remove ions, particulate matter and colloids. However, these systems were typically proven to achieve microfibre removal of 65-97 per cent (Swartz *et al.* 2017; Brodin *et al.* 2018). With better awareness of the possible impacts of microplastics, these technologies could also be optimized for microplastics removal. Research in this area is still in its early stages. Other types of low-cost technology (e.g. combining sedimentation and filtration) have also been tested. Costs associated with the treatment of effluents from industrial laundries are shown in Table 8.

Table 10. Concentrations and releases of microplastics and microfibres 100-1,000 μm in size in laundry effluents in Sweden

Laundry	Main type of fabric	Share of MFs compared with all MPs (per cent)	MPs concentration in effluents (per litre)	Total MPs released (per kg of textile)
Hotel	Cotton Polycotton (50 per cent polyester + 50 per cent cotton)	17-50	1,000-3,000	5,000-15,000
Hospital 1	Polycotton	30-68	103,000-235,000	711,000-1,620,000
Hospital 2	Polycotton	28-65	11,500-26,500	106,000-249,000
Mats	Cotton, nylon, rubber	49-83	151,500-254,500	318,000-534,500
Work clothes	Polyester, cotton, polycotton	81-95	385,000-455,500	4,550,000-5,375,000

Reference: Brodin *et al.* (2018)

c. Treatment of tyre particles

Improving the design of tyres and road surfaces to reduce microplastics resulting from tyre abrasion is being explored, currently mainly at research scale. There are also technologies that aim to trap microplastics after they have been generated[9] or minimize their generation through smart tyre design. Research in the latter area focuses on 1) making tyres more resilient to wear and tear, 2) using alternative, environmentally friendly (e.g. biodegradable) raw materials in tyre manufacturing; and 3) better labelling to inform users about the quality of marketed products with respect to this parameter (Quinn 2018). Tyre particles play an important role in atmospheric microplastic pollution, and they may be transported even to remote regions (Evangeliou *et al.* 2020).

2. Design of new textiles to reduce microfibres generation during washing

Fibre shedding from textiles is highly variable (differences of up to 1,000-fold), depending on the type of textile (Ross 2020). For example, thicker fabrics tend to shed more, while nylon fibres, filamentous yarns and woven fabrics shed less (Ross 2020). Increased control of production techniques and of textile quality could help in this regard. Examples of textile manufacturing processes that reduce releases of microfibres during textile washing include (Jönsson *et al.* 2018; Prata 2018; Hanning 2020):

- Improved knitting techniques avoid tight knitting, which increases the concentration of fibres per area and the amounts released during washing.

- Ultrasonic welding of fabrics is better than conventional cutting techniques: microfibres reduction is 70 per cent for particles more than 5 µm in diameter.

- Innovative and quality formulations (e.g. effectively combining synthetic and natural textiles and eliminating loose fibres) could reduce fibre loss during washing by 80 per cent.

- Textile coating (e.g. with silicon emulsion) can also reduce microplastic releases.

The cost of enforcing such measures and practices will ultimately be borne mostly by the consumer; hence, the need to incentivize implementation of these technically advanced solutions with supporting policies and awareness-raising campaigns to achieve greater impact. Safeguarding heath in the textile industry is essential, as studies have reported that women who work in textile factories and are exposed to synthetic fibres and petroleum products at work before their mid-30's seem to be most at risk of developing breast cancer later in life. Many modern synthetic fibres are basically plastic resin treated with additives such as plasticizers, many of which are recognized mammary gland carcinogens and endocrine disrupting chemicals (Lynn *et al.* 2017). To ensure that only good quality textiles are used in that country, the Government of Sweden is considering establishing a tax on harmful chemicals in clothing and shoes. This measure would aim at reducing the occurrence, spread and risk of exposure to harmful substances (Silow 2019).

3. Policy tools to reduce use and misuse of microbeads

According to recent studies, PCCPs may contain 0.5-5 per cent microbeads (with an average size of around 250 µm) (Table 11). Manufacturers of general cleansing products and toothpaste add microbeads to scrub skin or exfoliate teeth. Each time such a product is used, 4,000-94,500 microbeads are released (Chang 2015; Prata 2018).

Table 11. Particle size distribution and concentration of microbeads from selected PCCPs

Product	Concentration of MPs (per cent weight)	Concentration of MPs (number per mg)	Size of MPs (mm)	Plastic types
Face cleanser	0.94-4.2	-	0.1-0.2	PE
Hand cleanser	0.18-6.91	-	0.1-0.2	PE
Shaving foam	0.1-2.0	-	0.005-0.015	PFTE
Toothpaste	0.1-4	-	0.014-0.8	PE, PES
Facial scrub	0.4-10.5	2,185-3,108	0.04-0.8	PE
Body scrub	0.87-11.2	625-1,186	0.07-0.1	PE

References: Sundt *et al.* (2014); Kalčíková *et al.* (2017)

9 For example, The Tyre Collective has designed a device that directs and captures tyre particles based on their electrical charge (The Tyre Collective 2020).

To address the issue of microbeads, several countries have banned their use in selected PCCPs (especially rinse-off products). The European Union has indicated the intention to take similar steps by 2020 (Guerranti *et al.* 2019). The pressure exerted by these countries has convinced some of the largest PCCP manufacturers to phase out microbeads in their products (Prata 2018; Guerranti *et al.* 2019). Microbeads are being replaced by abrasives such as perlite, silica and microcrystalline cellulose.

Related policy tools could also be explored to enhance industrial microbead management and control environmental contamination by microplastics resulting from industrial processes.

4. Behavioural change campaigns to reduce the use of microbeads and generation of microfibres at source

Consumer decisions affect the volume of microplastics (including microbeads and microfibres) released to the environment. The good news is that unsatisfactory attitudes and practices may be corrected through behavioural change campaigns. Previous findings show that environmental messages are more effective if they are tailored to relevant target audiences (male/female). Creating gender-sensitive knowledge products highlighting linkages between consumer choices and waste is crucial. Targeted messaging is key. Inclusive stakeholder engagement bearing in mind gendered roles in household consumption and domestic waste management is also crucial in introducing new ways of thinking in all sustainable consumption and production practices, as well as in value chain assessments in waste management (Woroniuk and Schalkwyk 1998).

Although the amount of microbeads released per individual may seem low, it is significant in view of the large population concerned. Everyone has a responsibility to choose the right PCCPs, provided they are affordable, and to reduce their microplastics footprint. It is therefore useful to educate the general public on the impacts of microbead releases to the environment. To that end, some organizations are promoting adequate labelling to identify clearly, for example, products containing microbeads and their concentration levels. Labelling can help generate additional pressure on manufacturers to phase out the use of microbeads.[10]

Examples of personal decisions that could have an impact on the release of microbeads or microfibres are:

- Use of quality PCCPs that contain no or fewer microbeads. In Europe and the United States, before voluntary and policy bans were enforced, consumption of microbeads in soap was 0.88 grams per person per year. In the case of facial scrub, consumption could reach 80 grams of microplastics per person per year (Kalčíková *et al.* 2017; Prata 2018).

- Use of less powder-type detergent and of adapted softeners. During washing the use of such detergents increases releases of microfibres (due to increased abrasion caused by the inorganic insoluble compounds it contains) compared with a no-use scenario. At the same time, De Falco *et al.* (2017) found that the use of softeners could reduce microfibres release by at least 35 per cent.

- Use of front-loading washing machines. Microfibres generation is greater in top-loading (vertical axis) or industrial washing machines than in front-loaders (Henry *et al.* 2019).

- Reduction of tumble drying. Tumble drivers can be responsible for a 3.5 times increase in the release of microfibres compared with washing only (Hanning 2020).

- Better use of fabrics to increase their lifetime, including through recycling, and use of renewable materials in fabric production.

- Use of innovative fabrics, such as those produced using approaches which reduce fibre release following use (e.g. encapsulation of fibres) (Environmental Audit Committee 2019).

To improve general understanding of the impacts of microplastics in daily life and encourage changes in consumer behaviour, public education programmes are essential. Disseminated messages should highlight the social costs of pollution and extend beyond reuse and recycling of resources to their responsible use and minimization of waste generation. Key cost items to consider in developing such programmes are service contracts (e.g. for media engagement), rental or purchase of essential equipment such as vehicles, and the costs of supplies and personnel (Somda *et al.* 2013).

10 There are online applications that allow users to check the plastic content of certain products, so that consumers can choose less polluting ones.

Section III

Technologies to Treat Wastewater and Run-off Before the Treatment Plant

A. Macroplastics removal in run-off

Municipal wastewater comes from residential and domestic, industrial, commercial and run-off sources. Combined sewer systems may be found in areas with wastewater collection systems that are several decades old. In these systems rainwater run-off and domestic and industrial wastewater are all collected through a single pipe and channelled to a WWTP, where they are treated and then discharged to a water body. However, during periods of heavy rainfall or snowmelt the volume of wastewater may increase beyond the WWTP capacity, resulting in an overflow of diluted untreated wastewater directly into water bodies. The norm is currently to promote the use of separate sewer lines for domestic/industrial wastewater and run-off. Rain or storm run-off water that washes off roads, parking lots and rooftops is channelled through drains and canals into water bodies, often without much treatment. This run-off water is full of contaminants, including plastics and other debris.

The properties of plastics such as size, shape, and polymer type are reported to be the main drivers of their transport in run-off water channels. Environmental parameters such as salinity, wind and flow speeds, nutrients, and temperature affect their transport, creating regional differences (Tyler 2011). Several plastics remain buoyant in water, while others sink. Yet another group has an intermediate density,[11] and their sedimentation behaviour depends more strongly on local environmental factors.

Removal of plastics from wastewater or run-off water usually serves various purposes, such as reducing equipment and piping damage at the WWTP while also avoiding contamination of aquatic ecosystems. Many factors should be considered when identifying a plastic waste removal technology for wastewater and run-off water in drainage systems. They include location, local water dynamics, transport pathways, costs, the surrounding infrastructure, and landscape. The characteristics of run-off or wastewater (e.g. flow rate, width and depth) must also be considered.

Currently, the most used technique for the removal of floating plastic waste is a regular urban clean-up service to avoid accumulation of plastic waste in run-off water. In 2018 the city of New Orleans in the United States reported that 46 tons of plastic beads originating from Mardi Gras celebrations had been removed from stormwater drains during a cleaning exercise (Cherelus 2018). An alternative strategy could involve using a boat to collect plastics floating on the drains. However, both techniques appear to be costly, as well as time and labour intensive, and therefore unlikely to be sustained across all the large drains of run-off water. Setting up infrastructures such as booms, deflectors and meshes, which can operate on their own, is useful to reduce and remove plastic waste from wastewater or run-off water canals[12] before it enters freshwater sources or WWTP systems (Prata 2018).

In implementing removal structures, cost factors include the following variables:

- Structure required to withstand the anticipated debris and stormwater run-off (e.g. debris deflector or debris rack);
- Site location and access;
- Materials required for implementation;
- Number of structures and locations;
- Availability of a knowledgeable crew or contractor;
- Maintenance frequency.

No quantitative data exist on the effectiveness of plastic removal structures. However, subjective information indicates that they can be effective with proper implementation and maintenance. Problems may occur if the structure is too small for stormwater flows and associated (plastic) debris. Effectiveness monitoring of structures is needed, as well as frequent removal of plastics.

1. Booms

Booms are logs or timbers that float on the run-off water surface to collect floating debris, including plastic waste. They generally require guides or anchors to hold them in place. Booms are anchored close to drainage banks (left or right) to allow movement of traffic on the water and are cleared using clean-up boats equipped with a conveyor belt, a coarse shredder and several garbage dumpsters. Booms and collection devices can be designed to account for drainage size and to be climate-specific (e.g. storms result in large fluxes of water and hence plastic pollution). Booms have proven successful in deflecting plastic waste on the surface of the water. They have the significant advantage of not requiring the installation of permanent structures in the run-off water bed (aside from possibly the anchoring system) (Tyler 2011). However, they do not offer a solution with respect to plastics travelling below the surface.

The use of a boom to capture floating plastics has been implemented in Australia (e.g. Adelaide, Melbourne, Sydney and Cairns) (Figure 13). Table 12 shows the costs of technologies used to prevent run-off contamination. Larger booms' (> 30 metres) infrastructure (such as those that

11 Refer to Annex A for the densities of plastics.

12 These solutions could also be used to clean other water systems.

Figure 13. Examples of plastic clean-up efforts; left: combination bin and boom system that captures floating trash; right: a boom

Source: Elastec (2020)

cross entire drain estuaries) in the United States can cost up to USD 36,000, with USD 16,000 in annual maintenance fees (Bauer-Civiello *et al.* 2019). To reduce maintenance costs, a boom could be strategically put in place only during wet seasons, as well as downstream to avoid capturing the bulk of surface vegetation (Benioff Ocean Initiative 2019).

It should be noted that cheap inflatable booms can be degraded in the sun over time (Tramoy *et al.* 2019). Other factors may influence the efficiency of floating booms, such as intense run-off water flow along the drain and wind, both of which bring waste back to the banks or allow plastics to escape from the booms. Waves can also push plastics away from booms.

Table 12. Costs of technologies used to prevent run-off contamination

	Investment costs	Annual O&M costs	Durability	References
Booms	USD 485-1,200 per metre of boom length Depends mainly on type of material used and size	Typically, USD 533 per metre of boom length. To reduce O&M costs, the boom could be strategically put in place only during wet seasons, as well as downstream to avoid capturing the bulk of surface vegetation	Booms can last three to five years in turbulent water, and 10 years or more in calmer locations such as urban drains and creeks	Bauer-Civiello *et al.* 2019; Elastec 2020
Debris fins and deflector	Construction costs of these structures are part of the bridge construction budget	Structures require minimal maintenance	Structures have comparatively low environmental impacts when properly designed and installed. They last as long as the bridges, depending on the material used. Concrete can last a lifetime	Riggs and Naito 2012
Trash racks/ meshes	Typically USD 3,000-30,000 per unit, depending on size and the materials required	Manual clean-up units: USD 1,800-9,000 Mechanical clean-up units: USD 2,100- 9,700	Rack will last 10+ years with proper maintenance. Debris should be cleaned from the rack when required	Keating 2014

2. Debris fins

Debris fins, also commonly called pier nose extensions, are barriers built in the stream or drainage channel immediately upstream of a bridge (Bradley *et al.* 2005). While booms are designed to prevent floating wastes, including plastics, from travelling downstream or to direct plastics away from an engineered structure, debris fins allow waste in water, including plastics, to continue travelling in the flow in a directed manner, facilitating their eventual removal.

Debris fins are vertical walls that extend from internal culvert walls (Figure 14). The fin walls are intended to position large plastics from run-off water to pass through the culvert[13] entrance of the bridge without accumulating at the inlet. They are used extensively in bridge construction in many countries. The fin structure is recommended for the control of medium to large plastics. Debris fins for bridges are conceptually and structurally designed to correspond to the culverts (Bradley *et al.* 2005). They should be carefully aligned with the upstream flow and built with a downward sloping upstream face to limit impact forces and the probability of debris accumulation. Introduction of the slope in the design will make plastics trapped by the fin ride up along the top, allowing smaller plastics to flow underneath. When a piece of plastic strikes the fin, it is turned parallel to the flow, allowing it to flow past the support more easily without being caught (Bradley *et al.* 2005).

The upstream edge should be rounded to minimize the amount of plastics trapped. The fins may not be appropriate if they are too large for the culvert. Therefore, it is recommended that the length of the fins be one and a half to two times the height of the culvert (Riggs and Naito 2012). The culverts with which the fin is used have an opening of four feet or more. Installed fin structures require little maintenance and have comparatively low environmental impacts when properly designed and installed (Sheeder and Johnson 2008). Table 12 shows the costs of this type of system. They are usually included in a bridge construction budget.

Figure 14. Concrete debris fins extending upstream from a bridge pier

Source: Tyler (2011)

3. Deflectors

Debris deflectors are triangular-shaped frames placed upstream of bridge piers to deflect and guide wastes carried by water, including plastics, through the bridge opening and away from the culvert entrance. They are placed immediately upstream of a dam or drain structure in order to direct plastics from run-off water. Debris deflectors for bridges are similar in function to debris fins (Riggs and Naito 2012).

Deflector designs and materials differ significantly. However, they usually consist of either wood or metal within a pair of vertical grids that originate together in a "V" shape, with the apex pointing upstream (Tyler 2011). The apex angle should be between 15° and 25°, while the combined area of the two sides should be at least 10 times the area of the culvert opening (Riggs and Naito 2012). Although Figure 15 shows a vertical member at the apex, a sloping member may be more effective in guiding plastics away from the culvert opening. Storage capacity above the waste rack and the size of the accumulation area should be taken into account.

Unlike booms, these devices have the advantage of being able to deflect plastics throughout the run-off water column and are not restricted to floating plastic waste. The spacing of the horizontal members on the sides is chosen to allow smaller plastics to pass through, but prevent plastics large enough to plug the culvert. A spacing of two-thirds

Figure 15. Upstream view of a steel debris deflector

Source: Tyler (2011)

13 A culvert is a tunnel that carries a stream or run-off water under a road or railway. Traffic may also pass through it.

of the culvert diameter would be appropriate. Although the horizontal bars on the top may be needed structurally, they are only required for plastics if the water level is expected to be higher than the deflector (Jambeck *et al.* 2018). Deflectors have the potential to accumulate plastics. While most plastics are deflected, accumulation can be a problem. Cylindrical pile debris deflectors have been widely used throughout the United States, but their effectiveness as a debris accumulation countermeasure is questionable and they may intensify the problem under certain climate conditions (Sheeder and Johnson 2008). Table 12 shows the costs of this type of system. They are included in the bridge construction budget.

4. Trash racks or meshes

The most common technique for dealing with wastes (including plastic wastes) carried by water in traditional facilities is to use a trash rack to keep them from entering the wastewater penstock.[14] In some cases racks are similar in design to the deflectors used to protect bridge abutments, in that they trap the plastics but do not necessarily redirect the plastics as deflectors do. Traditional trash racks, designed to protect dams, consist of slightly inclined vertical bars (Figure 16) that stretch to nearly the entire height of the dam, typically from the bottom of the intake to above the water surface (Tyler 2011). These vertical bars are spaced according to the minimum size of the waste that needs to be kept from entering the penstock. They are generally made of mild carbon steel. Wrought iron, alloy steel and stainless steel are also used in certain locations. The bars are often attached to the dam by horizontal supports, which can be designed such that removal for maintenance is possible.

The use of trash racks involves two major challenges: 1) accumulating debris (plastics), leading to head loss for the racks themselves, and 2) structural fatigue of the racks, which is a serious design concern. The accumulation of waste (including plastics) is initially addressed by the slope of the rack. Ranging from 15° to 45° for low-pressure systems, the slope of the rack pushes wastes towards the surface and away from intake structures (Tyler 2011). Waste is usually removed from a rack by raking, either by hand or with mechanical rakes. Mechanical rakes are preferable for large facilities. The rake sinks into the water and is pulled up the rack face; once it arrives at the top of the rack, plastic waste is deposited in a collection receptacle.

Plastic removal structures vary in price depending on the materials. Costs for log debris structures are USD 100-4,000 each (Table 12). Log racks constructed with on-site burned logs are economically efficient. Rack structures built with heavyweight rail or steel cost USD 3,000-30,000 or more, depending on their size and the materials required. A heavy rail or steel structure may be worth the investment, depending on the type of materials mobilized and the values at risk (Guide 2020).

B. Microplastics removal in run-off

A key source of microplastics in stormwater run-off from urban areas and highways is the degrading of tyres when vehicles are driven. Microplastics in stormwater run-off which is not intended to be directed to a WWTP could be removed before the run-off is discharged into freshwater bodies. For particulate materials such as microplastics, sedimentation and deposition are the main removal mechanisms. To reduce or remove the volume and load of microplastics transported in run-off water, several processes such as sedimentation, filtration and infiltration are considered.

Figure 16. Debris racks

Source: Bradley *et al.* (2005)

14 A penstock is a sluice, gate or intake structure that controls water flow, or an enclosed pipe that delivers water to hydro turbines and sewerage systems.

Infiltration systems are designed to capture pollutants (such as microplastics) in run-off water and infiltrate the water into the ground, while filtration systems use soil, organic matter or a membrane as media to remove microplastics in run-off. The merits of both processes include water quantity control, but the infiltration process may cause contamination of soil and groundwater and is prone to clogging. Infiltration basins are a common structural tool used in urban areas for microplastics removal.

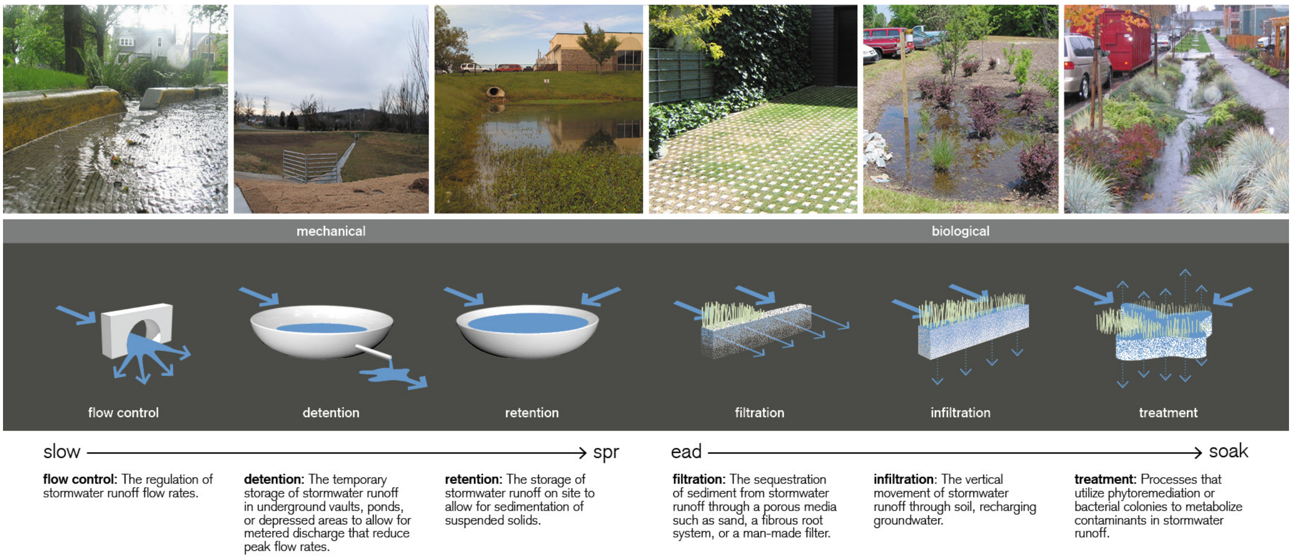
In a detention process, a volume of run-off is captured and retained for a period of time. Clean water is gradually released, but in the retention process the captured run-off water is retained until it is replaced by the next run-off water; thus, the system maintains a permanent pool. Retention systems can provide both quantity and quality control. Constructed wetland systems are also used. Structurally, they present similarities with retention and detention systems, except that major portions of the surface or bottom contain wetland vegetation. Wetlands are discussed in Section VI-A.

1. Retention ponds

In modern cities stormwater treatment often takes place in artificial basins called stormwater retention ponds (Figure 17). Stormwater run-off is drained into the pond and held there for a period of days to weeks before discharge, allowing microplastics and other particles to settle. For this treatment process the size, shape and density of microplastics are critical parameters, as they directly affect particle movement in water and determine final deposition rates. Microplastics 10-2,000 µm in diameter were analysed in stormwater treatment ponds in Denmark (Liu *et al.* 2019a). The results showed that stormwater in the ponds contained 0.49-22.89 items/litre (i.e. about 0.085-1.143 µg/litre). The lowest microplastic concentrations were measured in ponds that collected stormwater from highways and residential areas, while the highest were associated with industrial and commercial areas. Key plastic polymers included PVC (in the case of larger microplastics), PP, PE, PET and PS.

Retention ponds are one of the most effective stormwater management installation and they can remove some percentage of microplastics and particles. The effectiveness of such ponds has been projected in various research, producing variable results. However, it is understood that to improve performance proper design and maintenance are required (Liu *et al.* 2019). For instance, the inlet should be designed to decrease the velocity of the flow entering the system and should not be fully submerged at normal pool elevation. In addition, the inlet and outlet must be distant from each other. To determine the required volume of the pond, the gradual sediment accumulation should be taken into account. Wet ponds need an adequate drainage space to maintain the permanent pool. Construction costs vary considerably in different countries (Rollins 2019). Concerning maintenance of the system, the inlets and outlets of the pond should be checked periodically or after large storms for signs of clogging or the accumulation of debris. Other possible problems that should be looked at include subsidence, nuisance plants, erosion and litter accumulation. Sediments should be removed from the pond when necessary, or when the pool volume has been reduced significantly (Karlsson 2006).

Figure 17. Stormwater management processes (e.g. retention and detention ponds, infiltration)



Source: University of Arkansas Community Design Center (2020)

2. Infiltration basins

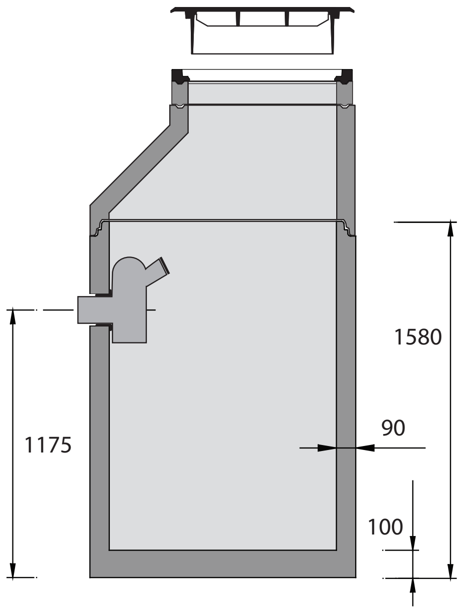
An infiltration basin is another sedimentation technique that can remove or reduce microplastics in run-off. It consists of water impoundment over porous soil. The basin receives stormwater run-off and contains it until the water infiltrates the soil (Figure 17). Infiltration basins can provide full control of peak or large volumes of stormwater run-off. If the stormwater run-off contains high amounts of soluble contaminants, groundwater contamination can occur. If soluble contaminants are known to be present, source elimination of these contaminants should be pursued. Research has shown that most existing infiltration basins have the highest failure rates of any microplastics removal system (Karlsson 2009). The main reasons are lack of pretreatment for removal of substances which can clog the basin and lack of maintenance. Maintenance is crucial for long-term use. For this system, the most critical maintenance item is periodic removal of accumulated sediment and microplastics from the basin bottom. If sediments are allowed to accumulate in excess, surface soil will become clogged and the basin will cease to operate as designed.

Sediment should be removed only when the surface is dry and mud-cracked. To avoid compacting soils, light equipment must be used. After the removal of sediment (microplastics), the infiltration zone should be dug deep to restore infiltration rates. Additional maintenance items include mowing buffer/filter strips, side slopes, and the basin floor. Exact cost data could not be obtained for this technology.

3. Gully pots

Countries such as United Kingdom extensively use roadside gully pots (“gullies”) in their drainage networks to remove microplastics and other pollutants (Karlsson 2006) (Figure 18). Also known as catch basins in North America, they are small sumps sited in the urban roadside drain which act as run-off inlet points. Their main purpose is to retain sediments such as those containing microplastics from road run-off water that would otherwise enter drains and sewer systems. This is in order to avoid blockages or hydraulic restrictions in the systems (Scott 2012). Gully pots are available in a range of diameters and depths and are made from a variety of materials (Scott 2012).

Figure 18. Concrete gully pot design



Source: Norwegian Water Institute (2020)

Gully pots accumulate significant amounts of sediments and require regular cleaning to prevent blockage. Blocked gullies may cause flooding. Leikanger and Roseth (2016) suggested that they should be emptied when a pot is 50 per cent full of sediment. They are normally cleaned with an “eductor truck” which uses hydrodynamic pressure and a vacuum to loosen and remove sediment (including microplastics) and the standing liquids from the gully pot (Karlsson 2009). A gully pot is quick and easy to install, reusable and cost-effective. It reduces or removes microplastics from road run-off typically with up to 80 per cent efficiency if well maintained. Operational costs include gully cleaning and sediment (incl. microplastics) removal, maintenance costs for the sump and trap, and O&M costs of the cleaning equipment. The cleaning costs can differ depending on the methods used, the frequency with which the pots need to be cleaned, the amount of sediment removed, and the costs of disposing of the sediment. Exact cost data could not be obtained for this technology.

Section IV

Wastewater Treatment Technologies

A. Description of processes and costs for municipal WWTPs

Treating municipal wastewater in a plant is the norm in many developed countries. However, in low- and middle-income countries only 33 per cent of the population is connected to a sewer. Wastewater for the remaining 67 per cent is collected and pre-treated in on-site systems or discharged directly to soil and into water bodies (WHO 2019). Conventional wastewater treatment requires preliminary, primary, secondary and tertiary steps. Table 13 describes the main objectives to be attained during each stage of the wastewater treatment process in a conventional set-up. Low-cost technologies commonly used to treat wastewater, especially in developing countries, include waste stabilization ponds (or lagoons), wetlands and anaerobic processes. The costs of WWTPs are highly country-specific. In Annex E several cost functions are proposed, which could be used to tentatively estimate the costs of construction, operation and maintenance of WWTPs. Beyond the cost of technology, it is also important to consider human health costs in the decision to treat or not wastewater. This might help increase the uptake of more sustainable technologies as opposed to the cheapest options. However, there is an important knowledge gap on the amounts of these human health costs.

Table 13. Conventional treatment of wastewater: objectives, fate of microplastics and costs

	Preliminary treatment stage	Primary treatment stage	Secondary treatment stage	Tertiary treatment stage and disinfection
Sequence of processes and objectives	Screening with metal grids to remove fine and coarse debris (i.e. > 10 mm in size)	1. Grit removal to remove sand, silt and other heavy particles (mandatory) 2. Skimming tank for grease, oil and fat removal (common) 3. Coagulation and flocculation to create large flocs of heavy metals and phosphorus [optional] 4. Primary sedimentation to remove particulate matter and flocs leading to removal of heavy metals, organic matter and phosphorus (common) 5. Flotation to remove floating materials and volatile organic compounds (VOCs) (e.g. those which are strong-smelling) and grease (optional)	To achieve biological and physical treatment removing: • Suspended particles • Dissolved nutrients (mainly nitrogen, possibly phosphorous) • Suspended and dissolved organic material • Colloidal material Processes are: 1. Aerobic, anoxic or anaerobic biological reactor (mandatory); examples are: a. Suspended growth biological treatment • Activated sludge (common) • Membrane bioreactors (achieves secondary and tertiary treatment simultaneously) b. Attached growth biological treatment • Trickling filters • Rotating biological contactors c. Combined growth biological treatment 2. Secondary sedimentation	Tertiary treatment processes are selected to ensure final effluent meets the required quality standard. It is not always absolutely essential. However, it is used to ensure adequate nutrient removal as well as removal of heavy metals (if not removed earlier). • Wetlands (low-cost) • Membrane filtration • Biological aerated filter • Slow sand filtration • Disc filtration • Dissolved air flotation • Adsorption • Gas stripping • Ion exchange • Advanced oxidation Disinfection is applied to a treated effluent to reduce loads of pathogens exiting the treatment plant. It is achieved with: • Chlorine or chlorine dioxide • Ozone, peracetic acid or other chemicals • UV radiation

	Preliminary treatment stage	Primary treatment stage	Secondary treatment stage	Tertiary treatment stage and disinfection
Performance achieved	Debris and floatable materials (based on design target)	• BOD: typically 20-30 per cent • Suspended solids: typically 60-98 per cent • Phosphorus: typically 60-95 per cent • Other pollutants, including heavy metals (based on design target)	Typically 85-95 per cent removal for BOD and TSS	• Typically, 90 per cent N removal • Other pollutants including heavy metals (based on design target)
Fate of macroplastics	The major part of macroplastics removal occurs during this step	Smaller plastic articles such as cotton swabs may remain in the wastewater	Minimal removal	Minimal removal
Fate of MPs	No removal	The major part of MPs removal occurs during this step, through: • Skimming of grease (for floating MPs) (major route) • Filtration and gravity settling processes for heavier MPs trapped in flocs (minor rout)	Exact removal mechanisms for MPs are unknown. Sludge flocs and microbial secretions help the accumulation and removal of MPs in sludge. This phenomenon is aided when the contact time is high. Nutrient level could also impact on the fouling behaviour. MPs may also be ingested by protozoans and metazoans. During this step the percentage of microplastic fragments removed is higher than that of MFs. This could be due to the fact that fibres were largely removed during the preceding treatment step.	The item number concentration per litre may increase during the process, while the concentration in mass per litre may be reduced. Effluent concentrations range from 0.01 to 91 MPs per litre.
Process costs[a] in developing countries[r1]	Not available	• Investment costs: USD 3-40 per capita[b] (2013) • O&M costs: USD 0.1-2 per capita[b] (2013)	• Investment costs: USD 10-150 per capita[b] • O&M costs: USD 0.2-8 per capita[b]	Not available

	Preliminary treatment stage	Primary treatment stage	Secondary treatment stage	Tertiary treatment stage and disinfection
Process costs[a] in Europe[r4] or the United States[r2,5] (excluding sewer)	Acquisition costs per m^2 for total screen area: USD 1,500 for the larger screen to USD 1,980-2,240 for the smaller screen area.[r4] Full construction costs (design, fabrication and installation) were USD 44,000-190,000 in the United Kingdom (UK).[r4]	Not available	• In the United States, average total (capital + O&M) costs: USD 1,295 per m^3/day treated, or USD 518 per capita.[r2, c] • Another source reports costs between USD 880-2,650 per m^3/day treated (or USD 352-1,060 per capita) for the United States.[r5]	• For conventional systems, capital + O&M costs average USD 1,717 per m^3/day (or USD 687 per capita)[r2, c] • For wetlands, capital + O&M costs average USD 159 per m^3/day (or USD 64 per capita)[r2]
Typical costs[a] for the full process (including sewer) in the Untied States[r2,r3,r5]	Not available, as treatment at this level is insufficient to meet the quality standards for treated wastewater		• Investment costs per m^3/day is: - USD 399-9,246 with an average of USD 3,308 (2017)[r2] (or USD 1,324 per capita) - USD 1,300-11,900 per m^3/day (2014).[r3] • O&M costs per m^3/day: USD 29-1,321 with average of USD 437 (or USD 175 per capita) (2017).[r2] • Between 4 per cent (per cent is lower for larger plants) and 25 per cent of investment costs (13 per cent on average) • USD 124 per m^3/day treated in Jaen, Spain.[r6] Note: the costs of O&M include sludge management.	Conventional systems • Investment costs per m^3/day: USD 984-144,224 with an average of USD 57,534 (2017) (or USD 23,000 per capita) • O&M costs per m^3/day: USD 76-21,804 with an average of USD 6,168 (2017) (or USD 2,768 per capita). • O&M costs: Between 1 per cent and 33 per cent of investment cost (10 per cent on average) Wetlands[r2] • Total costs: USD 379-11,016 with an average of USD 3,441 (2017) (or USD 1,377 per capita)

r References: [1]Drechsel *et al.* 2015; [2]Hunter *et al.* 2018; [3]Guo *et al.* 2014, [4]Keating 2014, [5]SAMCO 2016, 2019; [6]Pajares *et al.* 2019. Please refer to Annex E for detailed costs. For references discussing treatment at each process stage, refer to the respective sections.
a Investment costs include engineering, at 10-15 per cent of the total cost. They are also affected by the level of automation needed for the treating system.
b Wastewater generation per capita varies per country. It is 0.186 m^3/day (Iran), 0.098 m^3/day (India), 0.200-0.300 m^3/day (Australia), 0.455 m^3/day (Canada), and 0.400 m^3/day (United States).
c Costs depend on various parameters such as the type of process implemented, the treatment level required, the level of automation of the plant, etc. Typically:
- Investment in the United States (2019): USD 4,400 per m^3/day treated for an aerobic fixed-bed bioreactor wastewater treatment system; similar for membrane bioreactors; -20 per cent in the case of a moving bed bioreactor. Respectively, annual O&M costs per m^3/day treated are: USD 485, +25 per cent, +100 per cent.
- Investment in the United States (2019): USD 5,300-7,100 per m^3/day treated for an anaerobic wastewater treatment system; Annual O&M per m^3/day treated: USD 288-387 per m^3.
- In Iran, investment is USD 2,600-3,000 (or USD 484-550 per capita) for a wastewater treatment system using either activated sludge, extended aeration activated sludge, or sequencing batch reactor. Annual O&M costs are USD 111-147 per m^3/day capacity.

B. Macroplastics removal at municipal wastewater treatment plants

Before wastewater enters any plant for treatment it must flow through a debris removal structure which removes large floating debris, sticks or rags to protect the WWTP (including the mechanical equipment and piping) from blockage and/or damage. Then preliminary treatment (or pretreatment) of the wastewater is carried out through screening to retard the accumulation of solids and reduce abrasion of mechanical parts, which could help to extend the life of the infrastructure.

Screening is a structural unit installed to separate debris in and/or on water, which may include plastics, from entering the WWTP. Screening is made up of parallel bars or rods that can have a circular or rectangular opening. Screening units are categorized into two kinds based upon the opening size provided: coarse screens (bar screens) and fine screens. The size of a screening unit refers to the size range of the particles it removes. Coarse screens remove large plastics from wastewater and are typically made of woven wire cloths with openings of 6-20 mm or larger. Common types of coarse screens are bar racks (or bar screens) and coarse woven-wire screens. Some modern WWTPs use both coarse screens and fine screens (Tiwari 2018). Fine screens with openings of 0.2-1.5 mm are practically placed after coarse screens to reduce smaller plastics.

As shown in Figure 19, screens are generally placed inclining towards the flow of the wastewater in the wastewater inflow channels. Design considerations for screens include the depth and width of the channel; the approach velocity of the wastewater; the discharge height; the screen angle; wind and aesthetic considerations; redundancy; and head loss (US EPA 2020). Cleaning of accumulated wastes on coarse screens can be done either manually or mechanically, while fine screens are usually cleaned mechanically. One major advantage of using manually cleaned screens is that this requires little or no equipment maintenance although it requires frequent raking to avoid clogging and high backwater levels to avoid the build-up of waste. Nevertheless, increased raking frequency increases labour costs. Alternatively, mechanically cleaned screens tend to have lower labour costs than manually cleaned ones. The removal of the screen mat during manual cleaning may cause flow surges. This can reduce the solids capture efficiency of downstream units. Mechanically cleaned screens are not subject to these problems, but have high equipment maintenance costs (US EPA 2020). The costs of screen units used for plastics removal in a WWTP varies, depending on the type of technology used and its applicability in diverse situations. Suggested costs include construction, operation and maintenance. Currently in the United States, contractor bids on screenings removal in a wastewater project were USD 150,000-400,000 (US EPA 2020). Graphs can be used to relate average wastewater flow through a plant to a specific technology.

Figure 19. Typical screen

Source: Bradley *et al.* (2005)

In Greece it has been demonstrated that plastic waste is not totally removed during preliminary and primary treatment, leaving materials such as cotton swabs in the wastewater treated subsequently (Morgkogiannis *et al.* 2018). These could continue to disintegrate through mechanical shearing during the process, leading to an increase in the number of microplastics in the wastewater being treated.

C. Microplastics removal at municipal wastewater treatment plants

Microplastics from other sources contaminate wastewater, which then acts as a route to aquatic environments of microplastics discharged into the system upstream. Annex A presents the existing types of plastics, their uses/applications, and their relative abundance in wastewater. Annex D presents the characteristics of microplastics found in wastewater.

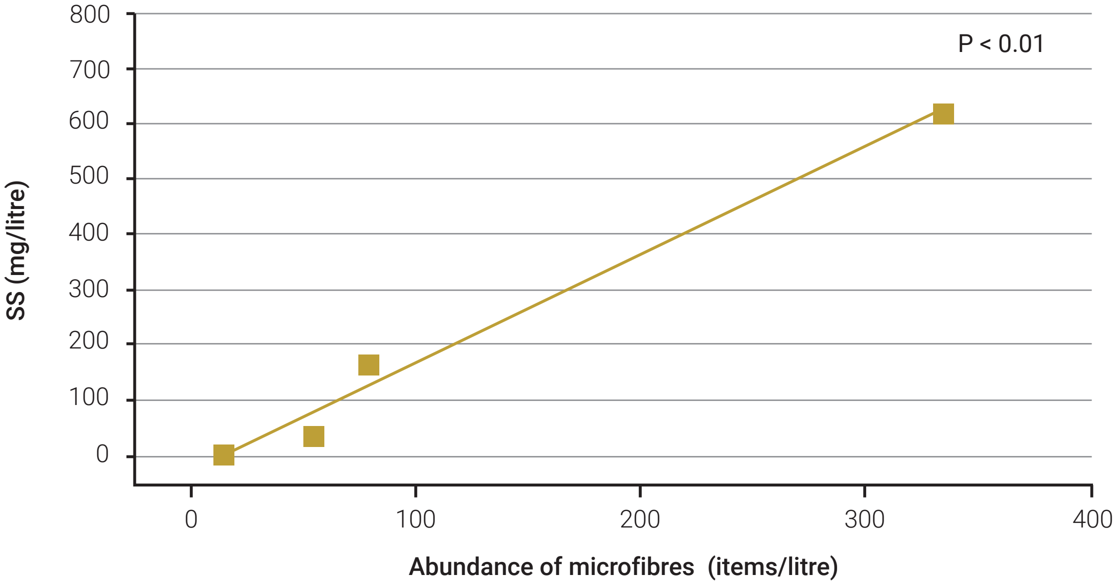
1. Key parameters impacting municipal WWTP performance

Several parameters which influence the removal of microplastics by WWTPs are presented and discussed in Table 14.

Table 14. Operating parameters which could affect WWTP performance in removing microplastics

Parameter	Impacts recorded on untreated wastewater quality	Impacts on treatment or surface water quality	References
Population size and preferences in terms of, for example, clothing and washing practices	• Total particles increase with increased population due to increased total number of plastic fibres. The number of plastic fragments is not affected • The profile of MPs in influent seems to be correlated with community textile laundering practices	A greater number of people served leads to higher concentrations of MPs in sludge.	Mason *et al.* 2016; Li *et al.* 2018; Sun *et al.* 2018; Conley *et al.* 2019
Combined sewers	• High flow rates of wastewater must be treated because run-off and domestic/industrial wastewater are combined • The number of plastic fragments in wastewater increases, but there is no change in the number of MFs. The increase in the number of fragments could be linked to contamination from adjacent land use (run-off) as well as land-related emissions, e.g. from tyres and brakes • MPs concentration increases with increase in the share of industrial wastewater in WWTP influent	• MPs concentration is reduced in surface water due to WWTP treatment • WWTP reduces MPs concentrations in the treated wastewater, but the amounts of microplastics released could remain high due to the large volumes treated • Higher concentrations of microplastics will be found in the WWTP sludge	Mason *et al.* 2016; Li *et al.* 2018; Sun *et al.* 2018
Climate	Seasonality seems to affect concentrations of MPs or MFs in wastewater, especially when run-off is combined with domestic wastewater	• Average temperatures and rainfall seem to follow a pattern similar to MPs concentration in sludge • In the United States, neither MP concentrations nor MP removal efficiencies followed seasonal trends	Li *et al.* 2018; Conley *et al.* 2019
Type of process	Not applicable	• Morphotypes of the MPs found in treated wastewater depend on process stage and type • High capex (which means high flow of industrial wastewaters and a better treatment technology) correlates with high concentration of microplastics in sludge and improved wastewater treatment • Removal performance of wastewater treatment processes depends on the shape of particles in the influent wastewater	Li *et al.* 2018; Sun *et al.* 2018; Xu *et al.* 2018; Lv *et al.* 2019
Shape and polymer type (properties) of MPs	There is a correlation between the concentration of suspended solids and the concentration of MPs > 300 μm in effluent, as shown in Figure 20	The size distribution of microplastics changes during the treatment processes. Overall, removal of MBs or microfragments is more difficult than removal of MFs	Lee and Kim 2018; Xu *et al.* 2018; Long *et al.* 2019; Lv *et al.* 2019

Figure 20. Correlation between MFs and suspended solids (SS) in industrial wastewater



Reference: Xu *et al.* (2018)

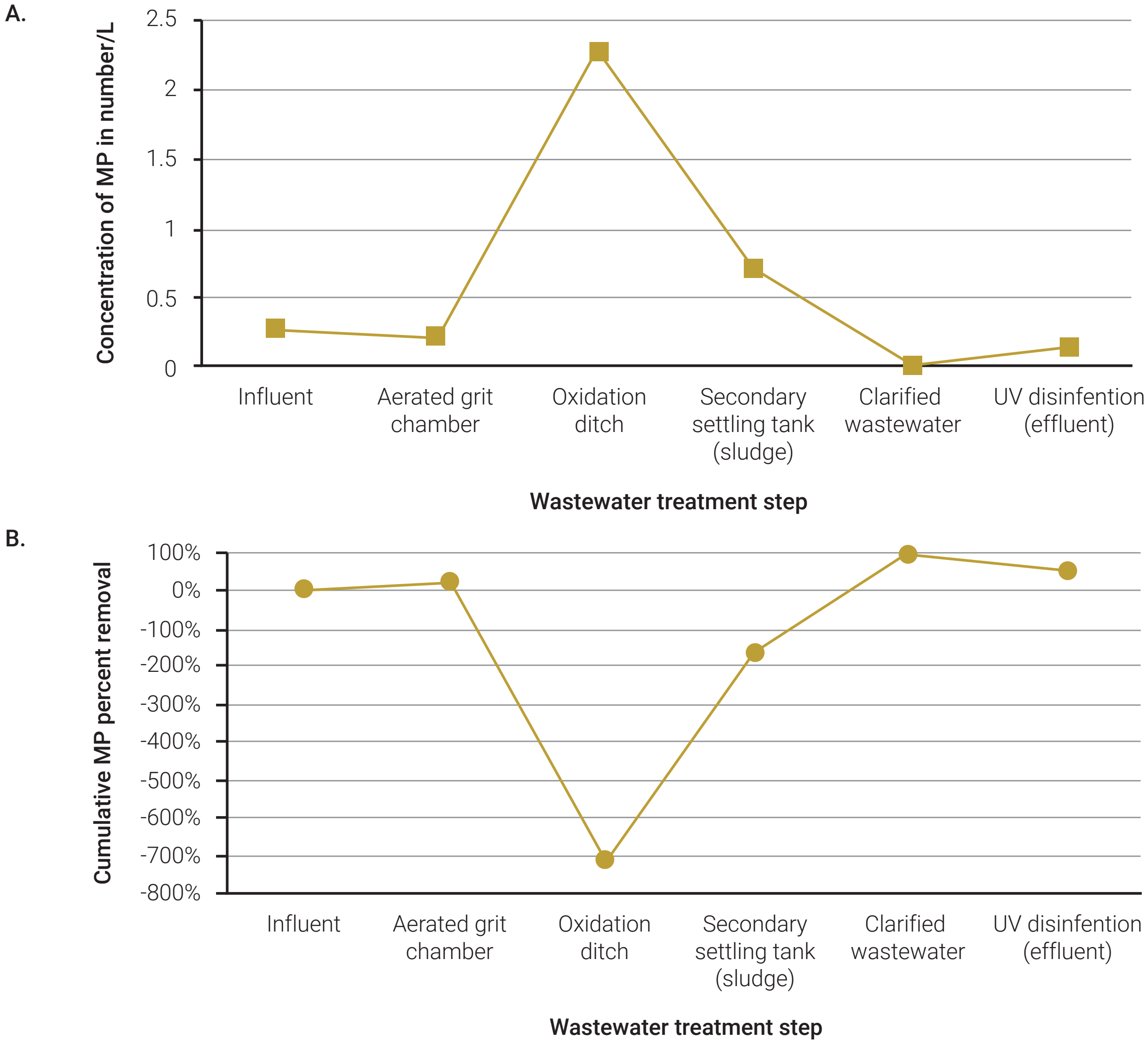
On the other hand, the removal of microplastics is not stable throughout the wastewater treatment process. While some individual stages concentrate the microplastics in the process, such as those involving high sludge concentrations, others lead to the removal of microplastics (e.g. in clarifiers). Figure 21A shows typical profiles of microplastic concentrations and Figure 21B presents the cumulative removal rates of microplastics, both during a given treatment process.

Figure 21 shows that microplastics removal within a treatment plant is a complex process which is not solely determined by one step. This is because the wastewater treatment process targets different contaminants, and therefore interactions can be observed.

2. Treatment performance per stage within a municipal WWTP

Although several studies have been published on this topic, the data were usually obtained following different methodologies. Therefore, differences observed between one study and another may be attributed to variations in sample collection, processing and analysis. This emphasizes the need for harmonization and standardization of analytical techniques. Moreover, data may vary even within the same study, leading to the recommendation by Mason *et al.* (2016) that large volumes be sampled. Often, only one-time measurements are taken while further measurements seldom are. This is a challenge, as there could be wide temporal and spatial variations in influent and effluent wastewater quality among different countries and studies. The reasons could also be linked to the points made above (Sun *et al.* 2018). In most of the studies considered in this section the lower limit for microplastics detection and quantification was 0.025 mm. The analysis and conclusions presented here should therefore be viewed as indicative. In addition, many authors have reported their data in numbers per volume. While this is convenient to collect, conclusions may be affected by intermediate processes leading to the shearing and shredding of particles. In this case, the number of particles may vary according to a pattern different from that of the mass concentration, which should also be monitored.

Figure 21. (A) Profile of microplastic concentrations and (B) cumulative microplastics removal efficiency during treatment in a typical WWTP in China



Reference: Lv *et al.* (2019)

Note: In this figure the aerated grit chamber is the primary treatment; the oxidation ditch and secondary settling tank constitute the secondary treatment, and finally disinfection is achieved using UV radiation.

On the other hand, even the reliability of the data presented could be questioned in some instances. Recently, Koelmans *et al.* (2019) applied nine quality control criteria adapted from criteria developed for biota samples to determine the reliability of studies on drinking water quality. Only four of 50 studies (8 per cent) received positive scores for all criteria (WHO 2019), while the others were not considered reliable for at least one crucial criterion.

a. Microplastics removal Performance following preliminary and primary treatments

Table 15 presents selected cases to illustrate how preliminary and primary treatment influence the removal of microplastics.

Current knowledge highlights that microplastics may be removed through fine screening (primary treatment step), sedimentation (primary or secondary treatment step), flotation (primary treatment step), and filtration processes (primary, secondary or tertiary treatment step). In addition, coagulation-flocculation (primary treatment step) could help facilitate microplastics removal during primary sedimentation.

Overall, primary treatment is the main step during which the largest amounts of microplastics are removed from wastewater (Raju *et al.* 2018; Saur 2020). However, the performance achieved during this step usually remains insufficient. For instance, microplastics removal

Table 15. Selected cases of preliminary and primary treatment performance with respect to microplastics removal

Process in use	Country	MP removal [b]	Inlet conc. (MP per L)	Outlet conc. (MP per L)	References
Screening, grit removal, skimming and primary sedimentation	-	78 per cent [m]	-	-	Mason *et al.* 2016
Screening, grit removal, primary sedimentation (no chemical used)	France	80.6 per cent [n]	1,737	337	Saur 2020
Screening, grit removal, physic-chemical lamellar settling (no chemical used)	France	78.6 per cent [n]	183	43	
Screening, grit removal, pre-aeration and sedimentation	Finland	82 per cent [m]	567.8	11.7	Talvitie *et al.* 2017b
		Estimated at 55-60 per cent[a,n]	180.0 MFs and 430.0 MPs per litre	14.2 MFs and 290.7 MPs per litre	Talvitie *et al.* 2015
		99 per cent [n]	57.6 per litre	0.6 per litre	Lares *et al.* 2018
Screening, aerated grit removal chamber	China	21-30 per cent,[n] 3 per cent [m]	0.28 (or 5.60 mg/litre)	0.22 (or 5.43 mg/litre)	Lv *et al.* 2019
Screening, rotary grit removal chamber		-371 per cent,[n] 1 per cent [m]	0.28 (or 5.6 mg/ litre)	1.32 (or 5.54 mg/litre)	
Screening, grit removal and primary sedimentation		41.7 per cent [n]	2.06	1.2	Ruan *et al.* 2019
Screening, flocculation + sedimentation		78.2 per cent [n]	1.01[c]	0.22	Ruan *et al.* 2019

a Assuming that 80-90 per cent of the inflow goes out after the primary treatment.
b The removal efficiency can be obtained on a percent mass basis or on a percent number basis. To differentiate between the two cases, [m] is for removal efficiency on the basis of the mass concentration while [n] is for removal efficiency on the basis of the item number concentration.
c Some pretreatment of the effluent has taken place prior to this step.

performance[15] attains 59 per cent [n] after preliminary treatment, while it is 42-82 per cent [n] after primary treatment in the majority of cases reported. In the United States, where wastewater treatment and monitoring is the norm, microplastics removal efficiency during these stages is reported to be 78-95 per cent. The key removal mechanisms are linked to (Carr *et al.* 2016):

- Grit and grease removal (typically, up to 45 per cent of microplastics are removed at this stage). In particular, microbeads are easily removed through skimming along with the fat, grease and oil. The microbeads removal rate is high if the level of fat, grease and oil is high.
- Sedimentation in the primary clarifier (typically, 34 per cent of microplastics are removed at this stage). It is important to note that this treatment step removes microfibres more easily than microfragments. Typically, the removal efficiency for microfibres is 93 per cent vs. 88 per cent for fragments.

The overall removal of microplastics during treatment is mainly determined by the removal performance achieved during this stage. In addition, while it may be difficult to improve removal of microplastics during the secondary and tertiary treatment stages, it appears easier to maximize microplastics removal during the primary treatment.

15 The removal efficiency can be obtained on a percent mass basis or on a percent number basis. To differentiate between the two cases,[m] is for removal efficiency on the basis of the mass concentration while [n] is for removal efficiency on the basis of the item number concentration.

b. Microplastics removal performance following secondary treatment

Table 16 presents selected cases to illustrate how secondary treatment influence the microplastics removal.

Performance of a secondary treatment plant concerning microplastics removal is better than that of primary treatment only. Additionally, it could be similar or better than that of tertiary treatment plant. In this summary table (Table 16), the high removal efficiencies are obtained with WWTPs located mostly in Europe and North America. In those cases, following the secondary treatment, 86-99.8 per cent[n] of microplastics in raw wastewater may be removed. This means the secondary treatment process adds 5-20 per cent extra removal compared to primary treatment only (cumulated) (Mason *et al.* 2016). Typically, removal at this stage could mean +5.6 per cent for microfibres versus +9.5 per cent for other microplastics. Particles more than 0.5 mm in size can be removed almost totally during this step, although exceptions were observed during some reported studies.

In other countries, such as China, removal performance of microplastics by WWTPs is reportedly lower, typically 64 per

Table 16. Selected cases of secondary treatment concerning microplastics removal

Secondary treatment variant	Country	MPs removal (cumulated with primary treatment)[b]	Inlet concentration (MPs/litre)	Outlet concentration (MPs/litre)	References
Membrane bioreactor[a]	Finland	99.4 per cent 99.7 per cent	0.6	0.004	Talvitie *et al.* 2017a' Lares *et al.* 2018
Activated sludge		88 per cent of microlitter (ML) [c] (99.98 per cent) [m]	11.7	1.4	Talvitie *et al.* 2017b
		Estimated at 75 per cent g (90.2-92.4 per cent) [m]	14.2 (MFs) 290.7 (MPs)	13.8 (MFs) 68.6 (MPs)	Talvitie *et al.* 2015
		Around 66 per cent [n] [98 per cent]	0.6	1.0	Lares *et al.* 2018
	Turkey	(74 per cent) [n] Note: the WWTP treats domestic wastewater	26,555 (average MP size: 1.57 mm)	6,999 (average MP size: 1.15 mm)	Gündoğdu *et al.* 2018
		[79 per cent] [n] Note: the WWTP treats domestic & industrial wastewater	23,444 Average size of MP: 1.68 mm	4,111 Average size of MP: 1.39 mm	
	China	77.5 per cent [86.9 per cent] [n]	1.2	0.27	Ruan *et al.* 2019
Oxidation ditch[d]		95 per cent [96 per cent] [n] 76.5 per cent [96 per cent] [m]	0.22 (or 5.43 mg/L)	0.01 (or 0.22 mg/L)	Lv *et al.* 2019
A2O process[e]		17 per cent [-293 per cent] [n] 15 per cent [16 per cent] [m]	1.32 (or 5.54 mg/L)	1.1 (or 4.70 mg/L)	Lv *et al.* 2019
7 WWTPs		[90.5 per cent] [n]	[6.55]	0.59	Long *et al.* 2019
Conventional activated sludge	France	85.2 per cent [97.1 per cent] [n]	337	50	Ross 2020
		- [87.8 per cent] [n]	[Inlet: 210] [f]	16	
Biofiltration		72.1 per cent [92.7 per cent] [n]	43	12	

a With membrane bioreactors, secondary and tertiary treatments are achieved in a single stage process.
b The removal efficiency can be obtained on a percent mass basis or on a percent number basis. To differentiate between the two cases,[m] is for removal efficiency on the basis of the mass concentration while [n] is for removal efficiency on the basis of the item number concentration.
c Microlitter is a mix of microparticles, mainly plastics, but could also include glass, metals, rubber, wood, paper, textile, such as cotton fabric.
d This process is a variation on the conventional activated sludge treatment process. It relies on long solids retention times for the treatment.
e A2O is Anaerobic – Anoxic – Oxic process. It is a variation on the conventional activated sludge treatment process. The biological reactor comprises three separate sections operating under anaerobic, anoxic and aerobic conditions.
f The process is composed of a screening unit (6 mm mesh), a grit and sand removal unit, and an activated sludge unit.
g The amount of wastewater passing through the system was reported to be 80 per cent of the inflow in some studies. In other cases, the authors calculated the removal based on the concentration only (i.e. assuming there is no change in volume flow going out after primary treatment).

cent (Liu *et al.* 2019b). The reasons could be linked to the operation and maintenance of the plants, which represent a high cost burden for emerging countries.

It is not evident which secondary processes are better than the others. In general, the WWTP was not designed to optimize the microplastics removal during the process. However, it appears that any process which removes particles can remove microplastics. Therefore, all secondary units of WWTPs are able to achieve notable removal of microplastics (Saur 2020).

c. Microplastics removal performance following tertiary treatment and disinfection

Table 17 presents selected cases to illustrate how tertiary treatment and disinfection influences microplastics removal.

Examples of tertiary treatment processes tested for their microplastics removal performance include: membrane filtration processes such as reverse osmosis, (rapid) sand filters, disc filters and dissolved air flotation (Booth *et al.* 2020). However, to date there is no information on how microplastics are/could be transformed during oxidative processes such as ozonation or advanced oxidation (WHO 2019).

Given that more stringent regulation are adopted in some parts of the world, many WWTPs are retrofitted to also enable tertiary treatment to take place, usually to remove additional fractions of nutrient or heavy metal levels and yield treated water effluents that are below standard values. Cumulatively, WWTPs which implement tertiary treatment are able to remove 95-99.9 per cent[n] of the microplastics in the raw wastewater. This is not a sign that tertiary treatment

Table 17. Selected cases of tertiary[a] and disinfection[b] treatment performance, concerning MP removal

Process variant[a,b]	Country	MP removal[c] (cumulated with preceding treatments)	Inlet conc. (MPs per litre)	Outlet conc. (MPs per litre)	Impact on MPs removal	References
Biological aerated filter (BAF)[a]	Finland	Up to 53.8 per cent (99.9 per cent)	1.4 (1-2)	2.5 (0.7-3.5)	Removal is occasionally negative, leading to an increase in ML release. This could partly be due to a buffer effect in the filter	Talvitie *et al.* 2017b
		85 per cent [d] (98.6-98.9 per cent)	13.8 (MFs) 68.6 (MP)	4.9 (MFs) 8.6 (MPs)		Talvitie *et al.* 2015
Sand filter[a]	France	-58.3 per cent (90.2 per cent)	12	17		Saur 2020
Filtering disks[a]		68.8 per cent (97.1 per cent)	16	5		
Membrane filtration[a]	China	95 per cent (79 per cent) [n] 83.5 per cent (99.5 per cent) [m]	1.1 (or 4.70 mg/litre)	0.06 (or 0.03 mg/litre)	The concentration in the membrane sludge is 4 MPs/ L (or 4.54 mg/litre)	Lv *et al.* 2019
UV disinfection[b]	China	-1,300 per cent (50 per cent) [n] 1 per cent [97 per cent] [m]	0.01 (or 0.22 mg/litre)	0.14 (or 0.17 mg/L)		
Chlorination[b]	China	-81.8 per cent (60.4 per cent) [n]	0.22	0.40	Redox during the process may cause bigger particles to be reduced in size.	Ruan *et al.* 2019

a Tertiary treatment process
b Disinfection process
c The removal efficiency can be obtained on a percent mass basis or on a percent number basis. To differentiate between the two cases, [m] is for removal efficiency on the basis of the mass concentration while [n] is for removal efficiency on the basis of the item number concentration.
d Assuming that up to 85 per cent of the inflow goes out after the primary treatment.

is beneficial to removal of microplastics in wastewater. In fact, the impact of tertiary treatment seems to be inconsistent from one study to another and also depends on the type of process implemented. While in some cases only WWTPs with tertiary treatment performed better than those ending after secondary treatment (e.g. when using membrane filtration processes or filtering disks), it was noted in other cases (e.g. biological aerated filter (BAF) and rapid sand filters) that the tertiary treatment itself led to an increase in the concentration of microplastics, expressed in number per litre (Mason *et al.* 2016; Sun *et al.* 2018; Saur 2020). As many authors do not report the concentration of microplastics in mass, it is difficult to confirm whether the same trend will be maintained on a mass basis. On the other hand, it appears obvious that the incremental benefit achieved with tertiary treatment compared to secondary treatment may not be financially justified when considering microplastics only. However, tertiary treatment aids in removing other pollutants and therefore may still be essential for adequate treatment of a wastewater stream.

d. Overall performance

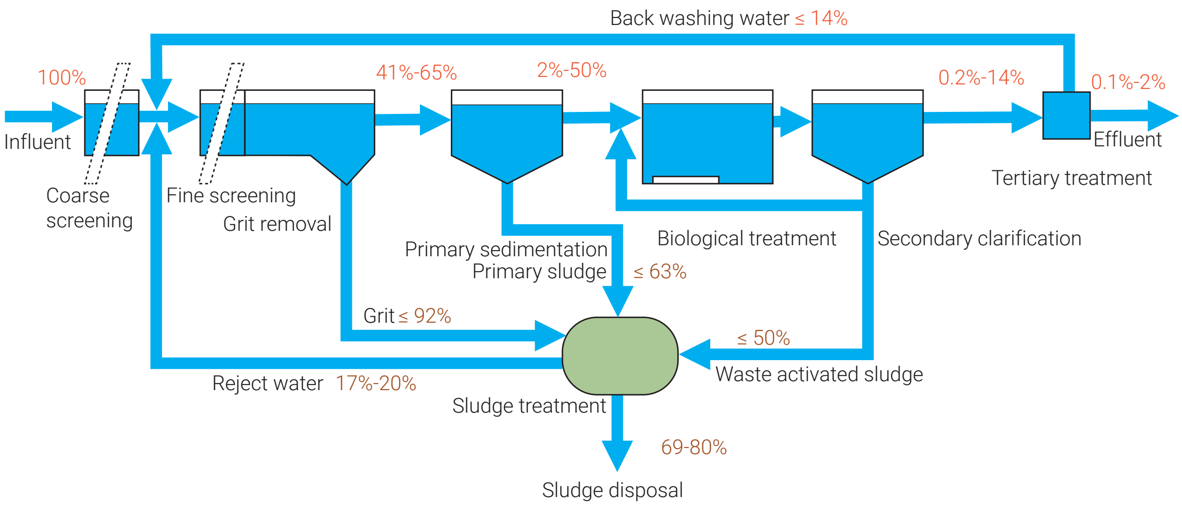
Overall, removal of microplastics can attain 99 per cent in a WWTP. However, this removal is simply a phase transfer of the microplastics from the liquid to the sludge. Inadequate management of sludge will lead to contamination of soils, the environment and natural systems. The estimated daily discharge through treated wastewater for a conventional WWTP among those currently studied (North America and Europe) remains about 10-60 grams of microplastics per day, depending mostly on the total volume of treated wastewater (Herbort *et al.* 2018a; Herbort *et al.* 2018b). During the treatment process microfibres are well removed from the wastewater, but microbeads and small microfibres could still be released in the treated effluent (Xu *et al.* 2018).

In the United States, Conley *et al.* 2019 established that emissions of microplastics through wastewater per capita and per year were 0.34-0.68 grams. This represents < 0.1 per cent of the microplastics contaminating the environment. WWTPs are therefore not always microplastics' main entry point into the environment. The situation could, however, be different in countries where WWTPs are not yet functional or wastewater treatment coverage is low.

Figure 22 presents the overview summary proposed by Sun *et al.* (2018), showing concentrations of microplastics at different stages of a conventional WWTP. In addition, two examples show typical distributions of microplastics during treatment by WWTPs operating in Canada (Figure 23) and China (Figure 24). Overall, WWTPs can be globally efficient for removal of microplastics and traditionally-targeted pollutants. Although some advocate for extra treatment through the addition of new effective treatment systems able to retain more microplastics, it appears that such extra treatment would have limited environmental benefits while the cost of implementation would remain high (De Gueldre 2020).

Microplastics removal performance remains uncertain with respect to several wastewater treatment processes used in developing countries, such as waste stabilization ponds. As these systems rely on extended residence times (several days) and display good sedimentation performance, it is anticipated that notable removal through sludge and scum may be achieved. The performance of anaerobic processes

Figure 22. Average microplastics flow in liquid and sludge within a WWTP with primary, secondary and tertiary treatment processes



Source: Sun *et al.* (2018)

Note: Numbers in red represent the microplastics content of liquid streams. Those in brown show the microplastics content of sludge streams.

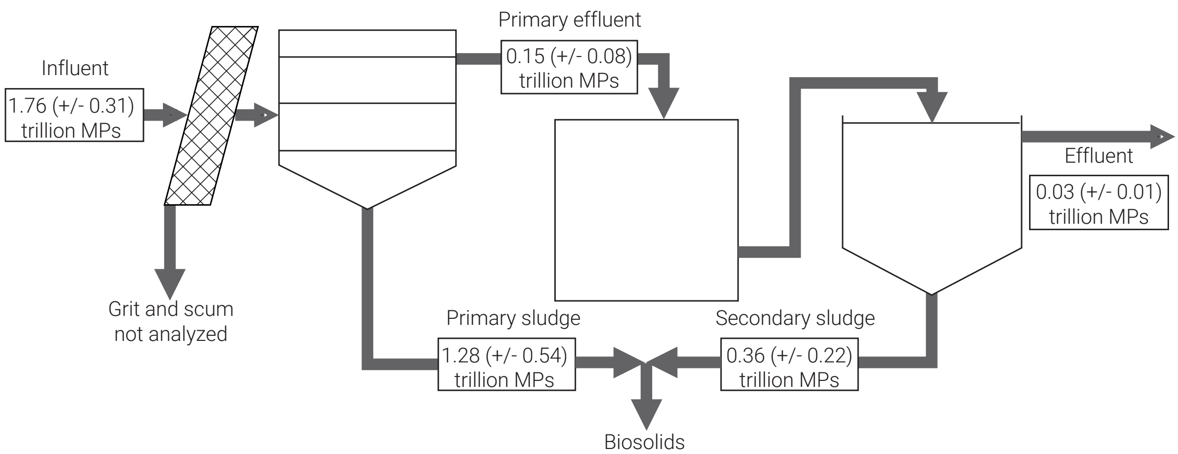
According to WHO (2019), more research is needed, for example, to understand the fate of microplastics across different wastewater and drinking water treatment processes under different operational circumstances. There could be a need to investigate further the role of microplastics breakdown and abrasion in wastewater treatment systems. In addition, the microplastics contribution from the processes themselves should be considered.

While membrane bioreactors are effective, they can be negatively affected through fouling by microbeads. Other technologies are also being investigated to improve the removal efficiency of microplastics during the treatment process. Electrocoagulation, enhanced flocculation/coagulation, dynamic membranes, combined filtration and photodegradation are examples of new treatment systems being explored and further optimized for enhanced removal of microplastics (Booth *et al*. 2020).

such as up-flow anaerobic sludge blanket (UASB) is also uncertain. This is therefore an important gap which needs to be addressed through appropriate research.

1. Example 1 - Fate of microplastics (in numbers) as they pass through a typical WWTP in Canada

Figure 23. Fate of microplastics (in numbers) as they pass through a typical WWTP



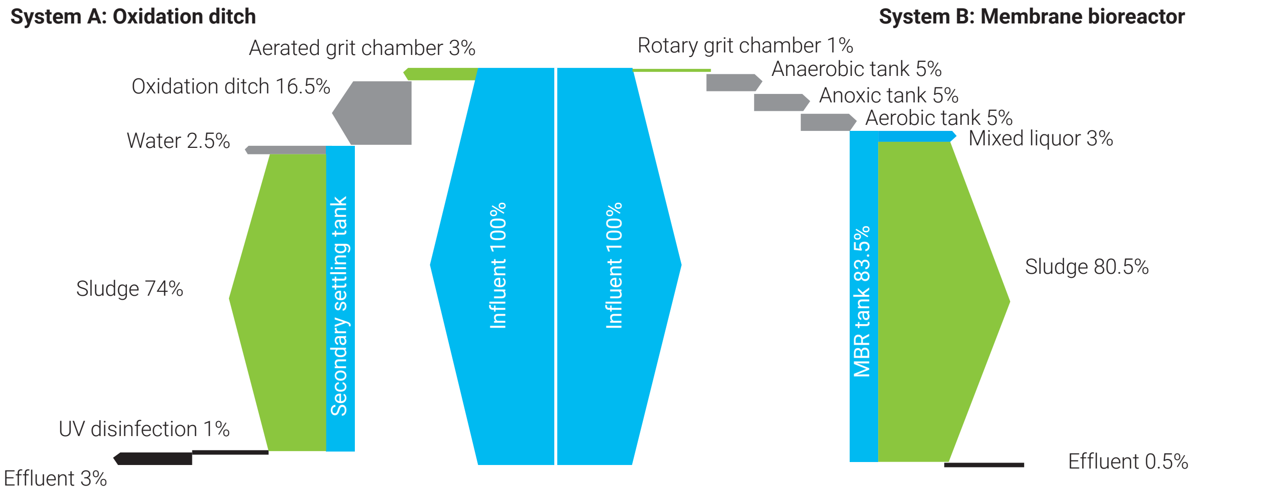
Reference: Gies *et al.* (2018)

Key findings

- Preliminary and primary treatment: 91.5 per cent removal efficiency for microplastics
- After secondary treatment: 98.3 per cent removal efficiency
- 72.7 per cent of the microplastics end up in the primary sludge and 20.5 per cent in the secondary sludge. This means 93 per cent accumulate in the biosolids. It is likely that 5 per cent are removed with the grit/scum.

2. Example 2 - Fate of microplastics (in numbers) as they pass through two WWTPs in China

Figure 24. Fate of microplastics (in numbers) as they pass through two WWTPs



Note: It is considered that the influent wastewater contains 100 per cent microplastics. The percentages for each process represent the microplastics reduced or contained at this stage.

Reference: Lv *et al.* (2019)

Key findings for System A	Key findings for System B
• 74 per cent of the microplastics leave the plant through the secondary sludge • 3 per cent of the microplastics remain in the treated effluent after passing through the WWTP • Other losses in the WWTP represent 23 per cent of the inflow and are lost in the treatment system. This could be due to accumulation in oxidation ditch.	• 80.5 per cent of the microplastics leave the plant through the secondary sludge • 0.5 per cent of the microplastics remain in the treated effluent after passing through the WWTP • Other losses in the WWTP represent 19 per cent of the inflow and are lost in the treatment system. This could be due to accumulation in other tanks.

3. Other potential solutions to improve WWTP performance in microplastics removal

So far, most studies on microplastics have been limited in their scope (e.g. they have targeted few facilities within limited geographic ranges). The authors of this report were unable to find regional studies indicating possible microplastics discharge levels for several countries. In addition to the current technologies used to remove microplastics from wastewater, Box 6 presents approaches that should be explored to reduce the impacts of microplastics in the environment.

Box 6. Approaches to be explored to reduce the impacts of microplastics in the environment

Technological solutions	These preliminary findings require further investigation: 1. Some plastic-degrading microbial species have been identified recently and seem to be able to remove microplastics from wastewater or sludge (Gatidou *et al.* 2018; Prata 2018). 2. Recent laboratory experiments have demonstrated the potential for pH-induced agglomeration to facilitate microplastics removal during some stages of the process (Herbort *et al.* 2018a). 3. There is a need to better understand the fragmentation behaviour of plastics, as well as microplastics removal during wastewater treatment
Analytical solutions	There is a need for globally accepted standardized analytical methods for microplastics analysis.
Health impacts	Beyond cost of technology, it is also important to consider human health costs in the decision to treat or not wastewater. This might help increase the uptake of more sustainable technologies as opposed to the cheapest options. However, there is an important knowledge gap on the amounts these costs.
Policies and awareness	There are currently no standards for permitted microplastics concentrations in treated WWTP effluents. Such measures would force low-performing plants to enhance their treatment performance (Prata 2018). In addition, the investigation of long-term behaviour and impacts is a prerequisite for establishing permissible levels for treated wastewater effluents.

D. Microplastics removal at industrial wastewater treatment plants

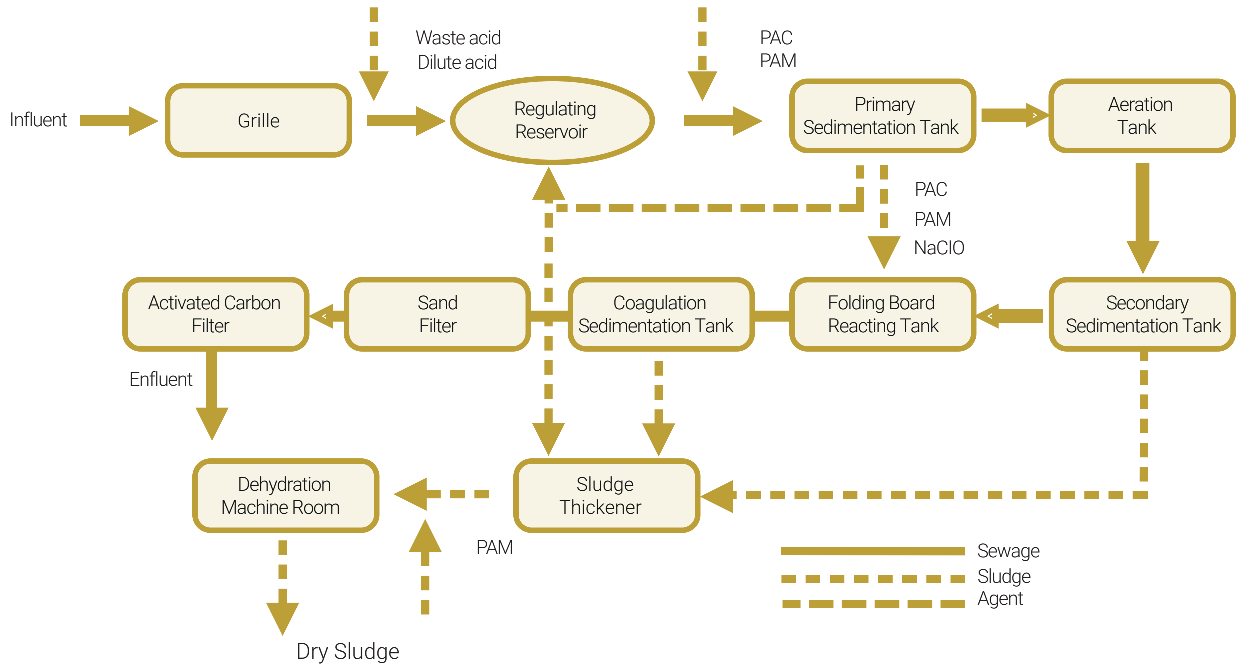
The impacts of types of industrial wastewater on the types and concentrations of microplastics in wastewater are still not fully understood (Gatidou *et al.* 2018).

1. Textile dyeing WWTP - a typical case in China

The water requirement for textile manufacturing is 0.1-0.2 m^3 of water per kg of textile product (Xu *et al.* 2018). In this particular case, the objective of wastewater treatment is to remove organic pollution and chromaticity. The WWTP treats 30,000 metric tons/day of wastewater, with 95 per cent in volume coming from 33 printing and dyeing enterprises while the remaining 5 per cent is domestic wastewater from residential areas. The influent wastewater quality is shown in Table 18.

Details on the treatment process implemented are shown in Figure 25. Treatment performance is presented in Table 18.

Figure 25. Wastewater treatment process within a facility



Note: Poly-aluminium chloride (PAC) and Polyacrylamide (PAM)

Reference: Xu *et al.* (2018)

Table 18. Influent quality and treatment performance of various elemental processes in removal of microfibres

	Influent quality	Removal efficiency	Step 1: Screening + grit separation + primary sedimentation (per cent)	Step 2: Aeration + secondary sedimentation (per cent)	Step 3: Coagulation + sand filter + activated carbon filter (per cent)
MFs	10.0 10^9 per day [a]	Cumulative	76[n]	84[n]	95[n]
		Individual process	76[n]	32[n]	70[n]
Chroma	342.0	Cumulative	-82	46	85
		Individual process	-82	70	72
COD	283.4 mg/litre	Cumulative	36	73	91
		Individual process	36	58	68
NH_3-N	3.9 mg/litre	Cumulative	28	43	68
		Individual process	28	20	44
SS	207.8 mg/litre	Cumulative	74	93	99
		Individual process	74	73	84
TP	0.3 mg/litre	Cumulative	24	49	77
		Individual process	24	33	56

Reference: Xu *et al.* (2018)

a More than 80 per cent of microfibres were larger than 0.03 mm in diameter, with the majority between 0.1 and 1mm, 60 per cent of the microfibres were microplastics, while the remaining was remainder were composed of natural fibres.

During industrial processes, removal of microfibres is mostly achieved during primary treatment and sedimentation and in membrane-based processes such as membrane bioreactor or reverse osmosis. The pigments found in the influent wastewater responded differently to the wastewater treatment process. While the percentage of pink and red microfibres was reduced in the final effluent (which is indicative of high removal efficiency), black and transparent microfibres increased to 16 and 24 per cent, respectively. For this plant the impacts of the treatment on the removal of pigmented microfibres remain uncertain. Moreover, the sludge from the WWTP was not analysed.

2. Landfill leachate

Typical compositions of landfill leachate effluents in China are presented in Table 19. Concentrations of microplastics in landfill leachate were 0.42-24.58 per litre. Landfill leachate generation in China is estimated at 1.3-3.2 m^3 per metric ton of waste over a 100-year period. For comparison, in Finland landfill leachate generation is 1.4 m^3 per metric ton of waste. In the Chinese leachate 17 different types of plastic materials were identified, with PE and PP representing 99 per cent of the total. In terms of shapes, the authors identified pellets (59 per cent), fragments (23 per cent) and fibres (15 per cent). In terms of sizes, 77.5 per cent of microplastics were 0.1-1 mm in diameter. Microplastics in landfill leachate originate from the fragmentation of plastics buried in landfills, as described in Annex C.

Conclusions:

- Concentrations of microplastics in landfill leachate seem to be lower than in many WWTPs. However, concentrations could reach 24.6 per litre.
- Even closed landfills could continue to generate microplastics in leachate.
- Where plastic concentrations in landfill solid waste are higher than reported in China (e.g. this is the case in many western countries (He *et al.* 2018a) and is likely in Africa, where recycling is limited), higher microplastic concentrations in effluents could be expected.

In India the typical investment costs for a landfill leachate treatment plant are USD 10,000-73,000 per m^3/day of treatment capacity. Annual O&M costs, which represent 2-7 per cent of the capital costs, are typically USD 1,460 per m^3/day (i.e. USD 4.0 per m^3 of leachate treated) (Gupta and Singh 2007). In the United States, O&M costs are reported to be USD 9.3 per m^3 of leachate or USD 3,240 per m^3/day (Evoqua Water Technologies 2020).

Table 19. Composition of landfill leachate in China

Six landfills in China	Operation time	Storage capacity in million metric tons (Mt)	pH	COD (mg/litre)	BOD5 mg/litre	Dissolved nitrogen (mg/litre)	Average MPs concentration (items per litre
Shanghai 1	2013 to date	6.9	7.8	3,052	132	1,760	11.8
Shanghai 2	2010-2016	3.8	8.0	1,905	295	1,757	1.3
Shanghai 3	1989-2014	0.23	7.7	880	36	1,217	1.0
Wuxi	2008 to date	4.23	7.9	12,220	2,371	3,711	0.7
Suzhou	1993 to date	13	7.9	3,960	1,520	2,199	3.0
Changzhou	2003 to date	3	8.0	9,815	2,493	4,106	2.9

Reference: He *et al.* (2018)

Section V

Technologies to Treat Contaminated Sewage Sludge

A. Macroplastics removal

Sewage sludge is an inevitable by-product of wastewater treatment. It is a mixture of solids and water produced during treatment. Since preliminary treatment of wastewater removes macroplastics and other debris, fewer macroplastics are found at the sewage sludge stage. The conventional sludge treatment technology is detailed in Section IV.B.

B. Microplastics removal

1. Composition of sludge

Table 20 shows microplastic concentrations in sludge produced at different stages of wastewater treatment. Different biological wastewater treatment processes result in different concentrations of microplastics in the generated sludge (Li *et al.* 2018). For example, anaerobic/aerobic (A/O) processes and their variants yield higher microplastic concentrations in sludge than an oxidation ditch and sequencing batch reactor due to retention time or settling efficiencies.

Table 20. Composition of sludge based on its origin

Origin of sludge within a WWTP	Typical concentration of microplastics (MP counts/g or /g dry weight [DW])	Description
Primary sludge (from primary clarifier)	14.9 MP counts/gram [b]	• Includes 65 per cent MFs. • MP fragments represent 34 per cent of the MPs. • Foam and pellets are also present in negligible proportion.
Activated sludge (from secondary clarifier)	23.0 MP counts/gram DW [e]	-
	113 MP counts/gram DW [c]	• 30 metric tons of sludge are produced daily (i.e. sludge generation is 0.075 g DW/litre of wastewater). • 47 per cent as MFs and 53 per cent as other MP shapes.
	4.4 MP counts/gram [b]	• Includes 82 per cent MFs. • MP fragments represent 58 per cent of the remaining MPs. • Foam and pellets are also present in negligible proportion.
A2O sludge (from secondary clarifier)	14.9 MP counts/gram [d]	• Contains 8 per cent MFs and fragments with size > 300µm • The ratio of sludge to influent is about 0.99 kg/m^3.
	240.3 MP counts/gram DW	• Average MP size in sludge is 223µm. • MFs (33-57 per cent) and fragments (30-46 per cent) dominated in the sludge.
Sequential batch reactor sludge	9.7 MP counts/gram [d]	• Contains 24 per cent fibres and fragments with size > 300µm. • The ratio of sludge to influent is about 0.76 kg/m^3.
Media-based process	13.2 MP counts/gram [d]	• Contains 20 per cent fibres and fragments with > 300 µm. • The ratio of sludge to influent is about 0.51 kg/m^3.
Digested sludge	170.9 MP counts/gram DW [a]	-
Membrane bioreactor (MBR) sludge	27.3 MP counts/gram DW [a]	-
Lagoon sediments	3.4-18.0 (average: 8.0±6.8) MP counts/gram DW [f]	• MFs represent 82 per cent, 89 per cent and 91 per cent of MPs in the sludge at sites processing. • Surrounding cities water discharges, fishing activity and industrial production sites are the most likely sources of MPs.
Various sewage sludges in France	Up to 5,000 MPs/gram DW About 0.5 per cent DW	• This applies to sludge from different process points.

References: [a]Lares *et al.* (2018); [b]Gies *et al.* (2018); [c]Magni *et al.* (2019); [d]Lee and Kim (2018); [e]Lv *et al.* (2019); [f]Abidli *et al.* 2017; [g]Saur 2020

2. Impact of sludge treatment on microplastic concentrations within WWTPs

As mentioned before, typically 69-99 per cent of the microplastics initially in the influent wastewater are transferred to the sludge fractions produced at different stages of the treatment process. The average size of the microplastics in the sludge is higher than in the initial wastewater, demonstrating that the sludge mainly concentrates large microplastics. On the other hand, microfibres typically represent 63-80 per cent of microplastics in sludge.

Traditionally, the main aim of sludge treatment is to stabilize the sludge to enable its disposal. A key step in sludge treatment is dewatering to reduce the water level in the sludge. The dewatering method selected affects the concentration of microplastics found in the final sludge. Processes often considered in sludge dewatering are listed in Table 21.

Table 21. Main characteristics of the sludge dewatering process

Characteristics	Drying bed	Belt press	Centrifuge
Land requirements	+++	+	+
Energy requirements	-	++	++
Implementation cost	+	++	+++
Operational complexity	+	++	+++
Maintenance requirements	+	+++	++
Complexity of installation	+	++	++
Influence of climate	+++	+	+
Sensitivity to sludge quality	+	++	+++
Sensitivity to type of sludge	++	++	+
Chemical product requirement	+	+++	+++
Dewatered sludge removal complexity	++	++	+
Level of dryness	+++	++	++
Odours and vectors	++	+	+
Noise and vibration	-	++	+++

Note: (-) None; (+) Low; (++) Moderate; (+++) High

During centrifugation, part of the low-density microplastics remains in the liquid leading to moderate concentrations of microplastics in dewatered sludge. However, filter pressure and belt-type dewatering produce dewatered sludge with high concentrations of microplastics. Mechanical erosion and sedimentation contribute to lowering the average particle size of microplastics in sludge (Liu *et al.* 2019). As a result of the dewatering, the concentrations of microplastics increase in the sludge up to 15 per grams wet weight (i.e. 1.5-170 per gram dry weight). Following sludge treatment, typically 95 per cent of the microplastics initially in raw sludge are retained in the final sludge. How drying beds affect content of microplastics in dewatered sludge is still unknown.

Key findings:

- Primary sludge typically retains 72.7 per cent of the microplastics. Secondary sludge retains an extra 20.5 per cent (i.e. about 75 per cent performance retention for this single stage) (Gies *et al.* 2018).
- Sludge from primary treatment (particularly the skimming process) has the highest microplastic concentrations, typically five times those of the sludge from the grit and biosolids. To avoid contaminating the remaining sludge produced during other parts of the WWTP process, the grease could be treated separately by means of incineration, pyrolysis or other thermal processes. This will permanently destroy the microplastics. In addition, since plastics have a calorific value close to that of normal hydrocarbon-based fuels, incineration yields high energy.
- Digested sludge is five times richer in microplastics than untreated activated sludge, mainly as a result of the dewatering.
- Lime stabilization seems to induce high concentrations of small-sized microplastics, resulting from the shearing of microplastics.
- Thermal drying of sludge causes the melting and blistering of microplastics.

The costs of sludge management are usually considered part of conventional WWTP costs.

3. Sludge post-treatment

Once the sludge is stabilized, in general it is either applied to soils as an amendment, deposited in landfills or incinerated. Table 22 gives details concerning specific usage of WWTPs sludge in selected countries.

Land application is the main post-treatment process applied to the stabilized sludge. However, it should be controlled since it contributes to increasing the microplastics content of soils. Accumulation of microplastics (mostly microfibres

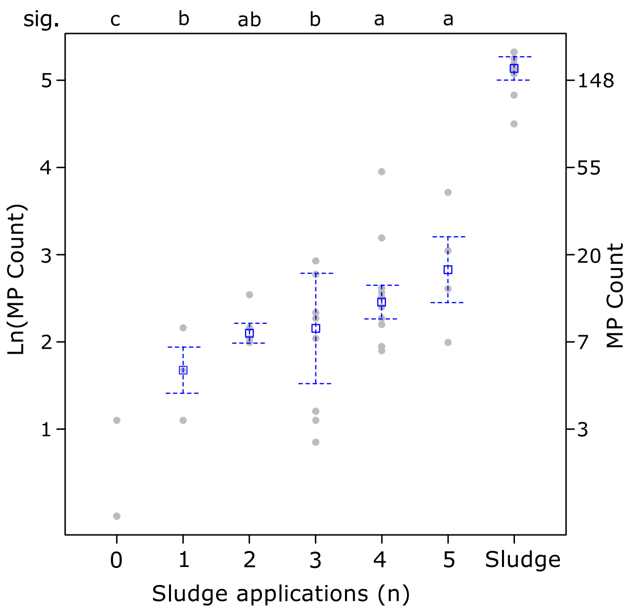
Table 22. Examples of uses of WWTP sludge in different parts of the world

Country	Available (metric tons/year)	Agriculture or soil production (per cent)	Landfill capping (per cent)	Incineration (per cent)	Amounts of MPs entering the environment (metric tons/year)
Australia	327,000	75 (direct) 8 (compost) 11 (land rehabilitation)	2	-	2,800-19,000 (land application)
China	40 million wet weight (8 million DW)[16]	45-86, depending on sources	34.5	3.5	228 18,760 MPs/kg sludge
United States	8 million	50	Not available	Not available	44,000-300,000[17] (or 125 to 850 per million inhabitants
Europe	125-800 metric tons of MPs per million inhabitants	54 (Sweden only)	25 (Sweden only)	Not available	63,000-430,000

References: Li *et al.* (2018); Raju *et al.* (2018); Da Costa *et al.* 2019

due to their abundance in the raw wastewater) in soil following sludge application has been reported by several authors, including Corradini *et al.* (2019), as shown in Figure 26. It appears that microplastics can be found in the soil even five to 15 years after the application of sludge to land (Sun *et al.* 2018).

Figure 26. Concentration of microplastics in soil following one to five consecutive applications



Total application rate over five years: 200 dry metric tons per hectare

Reference: Corradini *et al.* (2019)

This figure shows microplastic counts by number of sludge applications.

- Grey dots: average counts per field
- Blue squares: medians
- Blue dotted arrows: inter quintile range.
- Significant differences at α = 0.05 are shown in the top axis with lower case letters.

The impacts of microplastics in soils, including on soil organisms, farm productivity and food safety, are yet to be investigated in detail (Sun *et al.* 2018; Pepper *et al.* 2019) though some recent studies tend to point at the fact that microplastics could have impacts on animals and humans (Royal Society 2019). Microplastics in soil may also be carried by run-off water or wind to nearby aquatic environments and ultimately to the sea (He *et al.* 2018).

Earlier studies have shown that land sources contribute up to 80 per cent of microplastics entering water bodies (Magni *et al.* 2019). If contamination of run-off water by microplastics has been repeatedly reported, specific details on the extent of contamination remain uncertain.

a. Note

Nizzetto *et al.* (2016) estimated that, in Europe and North America, 110,000-730,000 metric tons of microplastics per year were added to agricultural soils via land application. This means the burden of microplastics in soils is greater than their current burden in oceans (Pepper *et al.* 2019). Microplastics' contribution through wastewater is lower than contamination through soil, run-off and atmospheric deposition (estimated at 118 particles per m^2 per day).

16 As of 2015. An increase in generation of 13 per cent per year is expected.
17 In North America, not only the United States.

b. Case study: China

Reference: Li *et al.* (2018)

Concentrations of microplastics in sludge are higher in China than in Europe. Microplastics in sewage sludge in China exceed those in freshwater sediments by one or two orders of magnitude. The amount of microplastics entering soils and the environment through sludge in China is about 1.56 10^{14}. This number:

- Approaches the discharge of plastic microbeads from facial scrubs in mainland China into the aquatic environment (i.e. 2.1 10^{14} or 306.9 metric tons);
- Exceeds the cumulated number of microplastics entering the global oceans from surface waters (estimated at 0.15×10^{14} to 0.51×10^{14}).

Section VI

Technologies to Treat Receiving Waters Downstream of Discharging Points

It is essential to adopt best practices and technologies to prevent freshwater pollution by macroplastics and microplastics. Nevertheless, because pollution depends on many factors (some of which cannot be completely controlled at reasonable costs), some degree of water contamination from land-based and anthropogenic activities may be impossible to avoid. In these cases, pollution prevention and treatment needs to be complemented with freshwater depollution and exposure reduction technologies.

A. Microplastics removal in wetlands

The use of wetlands as treatment systems for bioremediation, to capture and remove wide range of pollutants and nutrients, is widely practised around the world. However, they have not been much studied in regard to their capacity to reduce microplastics (Cui *et al.* 2010; Saeed 2012; Ranieri *et al.* 2014; Gorgogline 2016).

Table 23. Costs of technologies used to remove plastics in water contamination

	Investment cost	Annual O&M costs	Durability/profitability	References
Boats	A typical clean-up boat could have a trash collection capacity between 1.6 and 2.8 m^3. Investment cost not available.	Operational cost of the debris collector boat could be relatively high due to fuel consumption while O&M of the sweeper costs less. However, exact figures could not be found.	Not available	Elastec 2020
Sweepers	Variable, depending structures material. Where applicable, their costs could be a subset of bridge construction cost.	Not available	Not available	Lyn *et al.* 2007
Sea bins	A sea bin with a capacity of 20 kg trash load costs USD 4,000 in the United States.	O&M costs: USD 1,200 per year. This amount reflects an operational mode in which one bin bag is used per day. Energy consumption (500 watts) is included.	Recyclable components and structure are mobile. Sea bin can be used for five or more years.	Benioff Ocean Initiative 2019; Sea My Thoughts 2020
Wetlands	Total costs: USD 379-11,016 with an average of USD 3,441 (2017). Costs vary greatly depending upon initial site conditions. Earthworks cost is USD 2.5-15 per m^2 while planting costs USD 3-5 per transplant. These parameters typically represent 35-50 per cent and 11-17 per cent of the total construction cost, respectively.	Typically, USD 0.35-0.99 per m^2 each year. This is equivalent to up to USD 40-400 per m^3/day treated for the entire system.	Normally wetlands have indefinite lifetimes and are expected to be permanent landscapes. The opportunity cost of any land removed from agricultural production is not negligible, but could represent 50-70 per cent of total implementation costs. Other expensive components of constructed wetlands are site planning and design, excavation activities, and control structures required.	Tyndall and Bowman 2016; Hunter *et al.* 2018 Novak 2018;
Treatment of drinking water	Variable, depending on source of water used	In the United States, USD 1.5 per m^3	Usually operated on cost-recovery basis	Plappally and Lienhard 2012; Heberling *et al.* 2017

Wetlands are known for their ability to improve water quality by natural processes involving wetland vegetation, soils, and their associated microbial assemblages to filter water as it passes through the system. For conventional contaminants the removal mechanisms are primarily through transformation and uptake by microbes and plants, as well as assimilation and absorption into organic and inorganic sediments. The plants and microbes absorb nutrients and break down contaminants through biological processes (biodegradation).

Treatment wetlands (both natural and constructed) can be considered an end-of-pipe solution to reduce microplastics entering streams, rivers and the sea, while, floating wetlands provide an ongoing treatment process for freshwater systems (Wastewater Gardens [WWG] 2012; Bhomia 2015; Coalition Clean Baltic 2017). Table 23 shows the costs of this technology when used for stormwater treatment or tertiary treatment of municipal wastewater.

1. Constructed wetlands

Constructed wetlands (CWs) are engineered and managed wetland systems that are increasingly receiving worldwide attention for water treatment and reclamation. They are designed to mimic natural wetlands in overall structure while fostering processes that contribute the most to the improvement of water quality. Compared to conventional treatment plants, CWs are a cost-effective and technically feasible approach to treating polluted water. Easily operated and maintained, they have the potential to become a good alternative to conventional treatment technologies. Annex F presents some of the advantages and limitations of CWs (United States Environmental Protection Agency [US EPA] 2000a; US EPA 2004; Greenway 2017; Gorgoglione and Torretta 2018).

Several studies have reported the performance of CWs in removing different types of pollutants including nutrients, organic pollutants, suspended solids, heavy metals and pharmaceutical contaminates (Cui *et al.* 2010; Saeed 2012; Ranieri *et al.* 2014; Gorgogline 2016). However, much less is known about their ability to target microplastics. To date, we found that only one study has investigated the performance of CWs in removing microplastics. It was conducted in Sweden to determine possible microplastics reduction for two CWs located in Uppsala: Örsundsbro and Alhagen. High reduction efficiencies of up to 100 per cent for some of the identified microplastics were reported (Coalition Clean Baltic 2017). Table 24 shows removal results for different types and sizes of microplastics.

Consequently, it appears that constructed wetlands are able to remove high levels of small and rather large microplastics. Given their low costs of operation (Section 4.2), they could represent an interesting solution in developing countries. However, as in the case of all extensive processes, the land requirement is high, which could be a constraint in large cities where land is scarce.

Table 24. Microplastics removal efficiencies of two constructed wetlands (CWs) in Sweden

Facility	Örsundsbro wetland	Alhagen wetland
Area (hectares)	0.8	28
Mean flow (m^3/day)	667	5100
Theoretical residence time (per day)	3.5	86
Reduction efficiency: MPs 20-30 µm (per cent)	99.7	99.8
Reduction efficiency: MPs > 300 µm (per cent)	100	100

Reference: Coalition Clean Baltic (2017)

2. Floating wetlands

Floating wetlands (FWs) are man-made ecosystems. They are an innovative green technology which mimics natural wetlands to remove contaminants from water bodies in a passive and natural way. Small artificial platforms allow plants to grow on floating mats in open water where their roots spread through the floating mats and down into the water, creating dense columns of roots with lots of surface area. FWs are a possible suitable management practice; the unique ecosystem that develops creates the potential to capture nutrients and transform common pollutants that would otherwise plague and harm freshwater bodies into harmless by-products (Headley and Tanner 2012; Borne *et al.* 2013; Sample 2013).

FWs have primarily been used to treat wastewater and stormwater. While they have not been much studied in different freshwater bodies (rivers, ponds and lakes), they could be an invaluable alternative for reducing pollution in freshwater systems. Evaluating their effectiveness in terms of performance, cost-effectiveness, reliability and sustainability is therefore a priority area of research. There are many freshwater bodies in which floating wetlands could be deployed to improve water quality, including lakes, streams, stormwater ponds, wastewater lagoons, landfill leachate and tailings ponds, and oil spill sites (Tanner and Headley 2011).

FWs improve water quality through several mechanisms based on macrophytes, root systems, microorganisms and floating rafts. Similarly to a constructed wetland, nutrients and other pollutants are incorporated gradually into biomass and thus withdrawn from an aquatic ecosystem. While plants take up nutrients and contaminants, plant roots and the FWs' materials provide extensive surface area for microbes to grow, forming a slimy layer of biofilm. The biofilm is where the majority of nutrient uptake and contaminant degradation occurs in a FW system. The shelter provided by the floating mat also allows sediments and elements to settle by reducing turbulence and mixing by wind and waves (Hubbard 2010; Li *et al.* 2010). Like other technologies, FWs have advantages and drawbacks (Annex F).

Compared to municipal wastewater, quality of stormwater runoff is hard to monitor. This is why most studies analyse the concentration of microplastics in wetland liquid phase and compare it to that of microplastics in sediment. These analyses usually show that sediments have high microplastic concentration, which could reach 3,500 times that of the water (Olesen *et al.* 2019; Ziajahromi *et al.* 2020). This could explain why on some occasions, water may get enriched in microplastics as it transits in the pond.

B. Microplastics removal in drinking water

The possible uptake of microplastics via drinking water is of great concern with respect to public opinion. Recent studies have reported the presence of microplastics in both raw and treated drinking water, tap water and bottled water (Bouwman *et al.* 2018; Kosuth *et al.* 2018; Mason *et al.* 2018; Schymanski *et al.* 2018; Mintenig *et al.* 2019). There are a number of gendered health concerns that have to be tackled to ensure safe water access to women, youth and children who are at a higher risk. Beyond microplastics in drinking water, other plastic-based pollutants such as bisphenol A that dissolves in water are recognized mammary gland carcinogens and endocrine disrupting chemicals. Therefore, they require attention and monitoring (Lynn *et al.* 2017).

1. Bottled water

Microplastics have been detected in bottled water in several countries. It should be noted that packaging materials are often plastic, which is another possible origin of microplastics. However, significant amounts of microplastics were reported in samples from glass bottles or beverage cartons (Mason *et al.* 2018; Oßmann *et al.* 2018; Schymanski *et al.* 2018). The abundance and particle size of microplastics detected in three studies are described below and shown in Table 25.

Oßmann *et al.* (2018) studied the presence of microplastics in 32 samples of bottled water (Table 25). Microplastics

Table 25. Abundance per water volume (L-1) and size distribution of microplastics in bottled water

Type of water	1-5 (µm)	5–10 (µm)	10–20 (µm)	20–50 (µm)	50–100 (µm)	>100 (µm)	Reference
Plastic bottles, brand Aqua	NA	-	-	374	-	8	Mason *et al.* 2018
Plastic bottles, brand Aquafina	NA	-	-	200	-	13	Mason *et al.* 2018
Plastic bottles, brand Bisleri	NA	-	-	338	-	9	Mason *et al.* 2018
Plastic bottles, brand Dasani	NA	-	-	109	-	10	Mason *et al.* 2018
Plastic bottles, brand E-Pura	NA	-	-	238	-	10	Mason *et al.* 2018
Plastic bottles, brand Evian	NA	-	-	114	-	14	Mason *et al.* 2018
Plastic bottles, brand Gerolsteiner	NA	-	-	1,396	-	15	Mason *et al.* 2018
Glass bottles, brand Gerolsteiner	NA	-	-	159	-	9	Mason *et al.* 2018
Plastic bottles, brand Minalba	NA	-	-	63	-	4	Mason *et al.* 2018
Plastic bottles, brand Nestle Pure Life	NA	-	-	912	-	20	Mason *et al.* 2018
Plastic bottles, brand San Pellegrino	NA	-	-	27	-	2	Mason *et al.* 2018
Plastic bottles, brand Wahaha	NA	-	-	90	-	6	Mason *et al.* 2018
Single use PET bottles	2604	45	-	-	-	-	Oßmann *et al.* 2018
Reusable PET bottles	4664	142	-	-	83	-	Oßmann *et al.* 2018
Glass bottles	4895	969	-	-	434	-	Oßmann *et al.* 2018
Single use plastic bottles	NA	6	4	3	2	0	Schymanski *et al.* 2018
Returnable plastic bottles	NA	66	34	14	2	1	Schymanski *et al.* 2018
Glass bottles	NA	26	16	7	4	2	Schymanski *et al.* 2018
Beverage cartons	NA	4	3	2	1	1	Schymanski *et al.* 2018

Note: NA means this category is below detection limit were found in water from all bottle types, including single-use and reusable PET bottles PET as well as from glass bottles. While PET was the predominant polymer type in plastic bottles, various polymers were detected in glass bottles, including polyethylene or styrene-butadiene-copolymer. The authors therefore suggested other sources of contamination besides the packaging itself. Over 90 per cent of the detected microplastics were found to be smaller than 5 µm in diameter.

Schymanski *et al.* (2018) reported the presence of microplastics in all analysed bottled water, including from plastic bottles (reusable and single-use), beverage cartons and glass bottles. Most of the particles detected in water from reusable plastic bottles were identified as polyester (primary PET, 84 per cent) and polypropylene (PP; 7 per cent). This is not surprising since the bottles are made of PET and the caps are made of PP. In water from single-use plastic bottles only a few micro-PET particles were found. In water from beverage cartons and also from glass bottles, microplastic particles other than PET were found, for example polyethylene or polyolefins. This can be explained by the fact that beverage cartons are coated with polyethylene foils. However, high levels of microplastics were detected in some of the glass bottled waters. Most of detected microplastics particles were very small; almost 80 per cent were in the range of 5-20 µm.

Mason *et al.* (2018) reported microplastics in 93 per cent of 259 samples of globally sourced brands of bottled water purchased at 19 locations in nine different countries. The most common polymer type detected in all samples was polypropylene, which accounted for 54 per cent of total identified microplastics. A small fraction of particles (4 per cent) showed the presence of industrial lubricants. Most of the detected microplastics were small; 95 per cent ranged between 6.5 and 100 µm. The authors suggest that microplastic contamination was at least partially from the packaging and/or from the bottling process.

Overall, studies available in the current literature are limited and can with difficulty be compared with one another, given the variable context and methods used and associated quality assurance/quality control mechanisms (WHO 2019). However, the key initial results are that microplastics 1 µm or more in diameter are present in drinking water from different sources. It also appears that large particles occur less frequently than small ones, which is likely due to effective microplastics removal in sludge during wastewater treatment. So far, fragments and fibres seem to be the most abundant shapes of microplastics in drinking water, which compares well with observations for wastewater (WHO 2019).

2. Drinking water treatment

To date, there is no legislative limit for microplastics content in drinking water nor is any treatment technology targeted directly at their removal (Novotna *et al.* 2019).

As microplastics are increasingly detected in freshwater and potable water, and in view of their daily consumption, potable water could be a significant source of microplastics to humans, indicating that their behaviour during water treatment should be evaluated. Available data on the efficacy of microplastics removal during drinking water treatment are very limited. However, drinking water treatment processes (DWTPs) are highly effective in removing particles with characteristics similar to those of microplastics in terms of their sizes and concentrations. For example, conventional treatment such as coagulation, flocculation, sedimentation/flotation and filtration removes particles smaller than 1 µm in diameter. Coagulation in water treatment is aimed at associating dissolved or colloidal compounds to create aggregates suitable for subsequent separation. Then filtration, sedimentation, or flotation is applied as an intermediate separation step. Advanced treatment removes particles as small as > 0.001 µm (nanofiltration) and > 0.01 µm (ultrafiltration) (Ma *et al.* 2019a; Ma *et al.* 2019b; WHO 2019). In DWTPs, coagulation is a technology commonly applied while ultrafiltration (UF) membranes have been widely utilized (Ma *et al.* 2019b). However, little attention has been paid to microplastics' removal behaviour during these processes.

To date, only a few studies have reported the removal of microplastics during water treatment process. Pivokonsky *et al.* (2018) examined the efficiency of three DWTPs in the Czech Republic, all based on conventional coagulation/flocculation followed by separation techniques. The first DWTP involves two processes, coagulation plus one-step separation (sand filtration); the second DWTP involves coagulation plus three-stage separation (sedimentation, sand filtration, and granular activated carbon filtration); and the third uses flotation instead of sedimentation, together with the sand filtration and granular activated carbon filtration. All three DWTPs were found to remove microplastics with a removal efficiency range of 70-80 per cent (Table 26). The average removal rate was lower at

Table 26. Microplastics removal during drinking water treatment processes

Water sample taken from DWTPs	MPs abundance L−1 In raw water	MPs abundance L−1 In treated water	Size distribution (µm) <10 10-100 >100	Removal (per cent)
One-stage separation (i.e. sand filtration)	1,473	443	86, 13, 1 per cent	70
Two-stage separation (i.e. sedimentation + sand filtration) and filtration on granular activated carbon (GAC)	1,812	338	92, 8, 0 per cent	81
Two-stage separation (i.e. flotation + sand filtration) and GAC filtration	3,605	628	81, 17, 1 per cent	83

Note: Microplastics abundance is reported as an average (calculating the mean of minimum and maximum value).

Reference: Pivokonsky *et al.* (2018)

the first DWTP (70 per cent) compared with the other two (81 per cent and 83 per cent). This could be attributed to differences in the separation steps.

Mintenig *et al.* (2019) report that a very low level of microplastics (an average of 0.7 particles/m^{-3}) was detected in both raw and treated water. While it is difficult to observe any trend or draw any conclusions about the efficiency of the tested DWTPs in removing microplastics, it should be noted that the detection limit for this study was 20 µm. The authors argue that the detected microplastics were probably introduced as abrasives of plastic materials used during drinking water treatment and transport.

Uhl *et al.* 2018 conducted an assessment of treatment plants in Norway that use coagulation and filtration processes. They reported very low concentrations of microplastics in treated water and concluded that coagulation and filtration were effective in removing them.

In another study by Mintenig *et al.* (2019) the number and size of microplastics was determined at five different DWTPs in Germany which were supplied by groundwater. The study also investigated the presence of microplastics in five conventional household taps supplied by the same DWTPs. While the first study revealed high drinking water contamination (1,473-3,605 P L^{-1}), very low microplastic concentrations were detected in the second study (0-7 P m^{-3}). The dissimilarity in the results of the two studies may be related to several factors: the source of raw water (groundwater versus surface water), location, applied treatment technologies, and lower size detection limit of particles. For example, the majority of microplastics detected by Pivokonsky *et al.* (2018) were below 10 µm in diameter, while the detection limit in Mintenig *et al.* (2019) was 20 µm. There is a definite need for more research aimed at microplastics in public drinking water systems, with a special focus on those of small size.

Drinking water distribution and final consumption may also be of concern, as pipes and containers in households are often made of plastic and may contribute microplastics to delivered water (WHO 2019). More data are therefore needed to determine importance of each contamination sources (Mintenig *et al.* 2019).

3. Future trends

Regarding investigations carried out under laboratory conditions, two studies by (Ma *et al.* 2019a; Ma *et al.* 2019b) investigated the removal of microplastics. However, these studies followed the same treatment process employed by most DWTPs, that is, coagulation and subsequent separation (in this case ultrafiltration). The removal behaviour of polyethylene (PE) particles of different sizes (the main component of microplastics) was examined using commonly employed coagulants (iron [Fe]-based salt and aluminium [Al]-based salt). The Al-based coagulant showed better performance than Fe-based salt, but the required dose for both coagulants was extremely high (405 mg L^{-1} Al, and 112 mg L^{-1} Fe) compared to doses commonly applied for coagulation in drinking water treatment (13.5 mg L^{-1} Al and < 20 mg L^{-1} Fe) (Novotna *et al.* 2019). When using doses relevant to real conditions at DWTPs, PE removal was as low as 8 per cent. The authors then studied removal behaviour using a combination of coagulants, polyacrylamide (PAM) with Fe-based salt or Al-based salt. PAM is widely used to enhance coagulation during water treatment. High removal efficiency of up to 90 per cent was reported, yet the applied PAM concentration (3-15 mg L^{-1}) greatly exceeded the maximum authorized dose recommended by WHO (1 mg L^{-1}) (WHO 2011).

Based on these studies, coagulation and membrane filtration seem to be promising technologies for microplastics removal and deserve further investigation. However, the experimental set-up and conditions should better reflect conditions applicable in water treatment

Box 7. Microplastics' behaviour during water treatment: knowledge gaps

Possible interaction with other pollutants

As an example of this, microplastics may adsorb organic compounds owing to their hydrophobic nature (Napper *et al.* 2015). In that case they are likely to adopt the characteristics of a background organic pollutant, which in turn influences their removal profile. For instance, humic acids, a common organic pollutant, can stabilize particles in water and prevent aggregation (Jarvis *et al.* 2005).

Possible interaction with the chemicals used for treatment

A variety of chemicals are used throughout treatment processes that might interact with microplastics. For instance, a very recent study by Kelkar *et al.* (2019) investigated microplastics' behaviour during chlorination, a process commonly used during wastewater and water treatment to remove residual disinfectant. The study revealed physical and chemical alteration in the studied polymers, including polypropylene (PP), high density polyethylene (HDPE) and polystyrene (PS). Whereas the susceptibility to chemical degradation varied by polymer type, all three polymers showed some degree of alteration, which were detectable as changes in Raman intensity. The disappearance of some Raman peaks and the emergence of new peaks after sterilization with chlorine indicate both a breakdown of some chemical bonds and formation of new ones. The interaction of microplastics with other pollutants, and with chemicals used during treatment, might affect their removal and could have unknown health consequences.

Possible contribution of treatment processes and distribution system to microplastics contamination

As an example, many membranes are composed of polymeric materials and some processes such as ion-exchange use polymeric plastic materials, high shear-rate processes (e.g. in mixing systems) may degrade plastic particles into smaller particles, water pipes composed of plastic materials. These processes are exposed to abrasion and wear over time, which might release low quantities of microplastics into water.

Knowledge gaps in terms of gendered health impacts / risks

Gendered health effect of plastic constituents or associated with ingestion of microplastics or exposure to contaminants released following manufacture, treatment or recycling of plastics remain largely unavailable, especially in developing countries.

practices. Special consideration should also be given to the effect of microplastics' shape and size in regard to their removal.

C. Macroplastics removal in freshwater or the sea

Plastics from freshwater compartments all originate from land-based sources, which contribute approximately 80 per cent of plastics in marine environments (Bauer-Civiello *et al.* 2019). Many countries and inter-governmental organizations have been working to understand this issue and develop management interventions to mitigate the growing problem of plastics in marine environments.

Freshwater systems are a common pathway by which land-based plastic waste reaches the marine environment, as they connect coastal and inland urban communities to the oceans. Plastic items enter freshwater systems through various sources, including sewage effluents, storm drains, and recreational and commercial activities such as water-sports and fishing (Benioff Ocean Initiative 2019). Once these items are in a freshwater system, they can accumulate over time and the flow of water ultimately flushes the wastes into coastal and sea environments. Plastics originating from sidewalks, streets, highways, parklands and car parks, for example, are collected by water flowing throughout the catchments and washed into storm drains leading to river systems and eventually the sea. Other environmental factors, such as wind, also play an important role in the transport of plastics (Tramoy *et al.* 2019) such as by moving items from adjacent parks and recreational areas into aquatic systems.

1. Boats

Several types of boat are precisely designed to collect plastic pollution from river surfaces. They are positioned in locations ranging from nuclear waste facilities to major municipalities. Skimmers or conveyor belts skim plastics as the boats move on the water surface, as seen in Figure 27. Clean-up boats for removal of plastics are a simple and flexible technology to operate and maintain. This is a very practical option for river plastic clean-up. Clean-up boats have been deployed successfully in several rivers in the United States (Bauer-Civiello *et al.* 2019). An example is a skimmer baskets boat that cruises the Chicago River collecting plastics from the city's municipal sewer system. Additional details for this technology can be found in Table 23.

Figure 27. Garbage collection boat on the Pearl River in Guangzhou, China

Source: Elastec (2020)

2. Debris sweepers

Rather than attempting to control upstream of a structure such as bridge, sweepers are intended to buffer structures themselves from impacts and to steer plastics around a downstream structure. Sweepers are free to rotate on their vertical axis. Because sweepers rotate freely, they shed plastics, greatly reducing the likelihood of accumulation. The efficacy of installed sweepers in a number of locations, varies widely. American Association of State Highway and Transportation Officials expressed disparate opinions on the merits of sweepers (Lyn *et al.* 2007). Sweepers may be subject to failures due to clogging. They could be crushed by large plastics, or be dislodged from their mounts. A possible factor that may contribute to clogging failure of the sweeper devices is water flow speed; it has been observed that sweepers are not generally effective when flow speeds are low. For recently designed systems, some manufacturers claim that their sweepers (Lyn *et al.* 2007) could address this problem. These are vertically aligned cylinders that are attached to the upstream side of a structure (Figure 28). Such sweepers could rise and fall with the water's surface to mitigate the water flow speed issue (Bradley *et al.* 2005). Table 23 discusses the costs of this technology.

3. Sea bins

Sea bins look like floating trashcans, but are powered by pumps that pull water from their open tops through a filter bag at the bottom to collect plastic particles (Figure 29). They are designed to be placed in calm waters near a power source (a dock or a marina, for example). A typical sea bin is estimated to collect up to 1.4 tons per year of floating plastics, from large to small plastic particles (Riggs and Naito 2012). Table 22 discusses the costs of this technology. A larger version of this concept is the Marina Trash Skimmer which operates essentially as a large, industrial-sized water filter, capturing floating debris and absorbing surface oil and other contaminant. This is a dumpster-sized pump-and-filter tool also designed to attach to docks; It has been piloted successfully in e.g. California, Oregon, Hawaii and Texas, demonstrating its potential for deployment along rivers (Benioff Ocean Initiative 2019).

It is importance to acknowledge the significant barrier of cost available to sweeper technologies, a successful strategic solution will eventually comprise of the combination of methods and tools that is logistically and financially feasible in a given location.

Figure 28. Debris sweepers

Source: Tyler (2011)

Figure 29. Seabin placed in a river

Seabin before its installation

Sea bin in the water after installation

Source: The Seabin Project (2020)

Box 8. The Seabin V5

Seabin Group, a clean tech start-up with offices in Australia and Europe, initiated the Seabin Project. The project's ambitious mission is to develop innovative upstream solutions to help solve the global problem of ocean plastic pollution. Seabin technologies have the prospective to intercept mismanaged waste such as macroplastic debris in fresh water bodies before they have the chance to reach the ocean. They have developed a floating debris bin device called the Seabin V5, which acts as a trash skimmer and debris interceptor, and is used in tackling plastics pollution that are located in the water at marinas, ports and yacht clubs.

Each Seabin V5 device cleans projected floating debris of about 1.4 tons of per year (depending on weather and debris volumes). The device has a capacity of 20kg and can be replaced numerous times per day, if required. The Seabin V5 effectiveness relies on its strategic positioning, to allow wind and current bring the debris to its location. Seabin V5 consumes 500 watts.

The Seabin V5 has been piloted and tested in the Tutukaka Marina in New Zealand for a 11-month period. It gradually removed human-generated debris that finds its way into the marina. Data sheets were used to document the amount of debris collected in a range of different categories such as cigarette butts, plastic food wrappers, clear plastic packaging, pieces of foam, fishing gear and plastic bottles. Based on the result analysis, the most notable items removed during the period were 1,468 pieces of plastic and 517 cigarette butts.

Reference: Sea My Thoughts (2020)

Section VII

Selecting and Combining Solutions

Water pollution by plastic debris and microplastics is complex and multidimensional. Managing it effectively requires a range of responses. Solutions need to act on the design, production, consumption and disposal of the plastics that we will still use in the decades to come. This can reduce plastic and microplastic pollution at the source. Other responses need to limit the export of microplastics from cities and the landscape through the treatment of wastewater and run-off, protect water bodies from pollution loads, restore affected water ecosystems and minimize exposure to populations at risk. All these efforts must be supported by legislation, economic instruments, education and awareness that force real change on the ground. As shown in previous sections of the report, there is a large number of available solutions and policy makers and practitioners need to set priorities and select those that are more cost-effective and suitable for their local context.

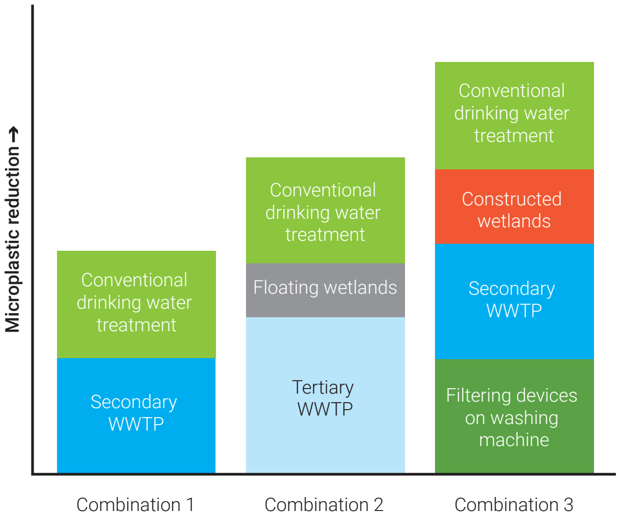
When planning to cope with water pollution, typically the first step is to set water quality objectives. Setting such objectives for microplastics is particularly challenging because their actual environmental or health risks are still being debated by the scientific community (Cole *et al.* 2011; Wang *et al.* 2019; WHO 2019). There is no consensus, for example, on acceptable daily intake values. One consequence is that there are still no international standards for microplastics that can help to set water quality objectives for drinking water, freshwater, or effluents of treatment plants (e.g. number of microplastics in a given size range per litre of water). Nevertheless, for planning purposes local decision makers and experts would need to tentatively agree on the desired water quality for microplastics and plan accordingly.

The decision on the type of water to be improved (e.g. WWTP effluents, freshwater or drinking water) will determine the types of solutions to consider. For example, while solutions such as wastewater treatment and reduction of microplastics at source in cities are key to achieve wastewater treatment effluents with low levels of microplastics, this will probably not be enough to achieve drinking water and freshwater quality objectives. Meeting these other objectives will require that pollution from land and road run-off is addressed or the adoption of further drinking water treatment.

Once water quality objectives are set one needs to assess what are the most relevant sources of microplastics and pathways to water in the local context. In a given watershed, microfibres from synthetic textiles could be the most relevant source and urban wastewater the main pathway, while in another watershed the main source could be microplastics from tyre abrasion with road run-off as the key pathway. In any case, it seems apparent that the challenge of secondary microplastics from plastic litter needs to be addressed with priority. First and foremost, single-use of plastics need to be reduced to a minimum and, more broadly, plastic waste management needs to improve, including reuse and recycling. This will prevent the occurrence of plastic litter in the environment which, with time, would tend to become microplastic.

Finally, for the priority sources and pathways, decision makers, in consultation with local stakeholders, need to select the most cost-effective and sustainable combination of solutions, as opposed to selecting single solutions at whatever cost. For example, to achieve a desired maximum number of microplastics in drinking water a secondary wastewater treatment (upstream) and a conventional drinking water treatment (downstream) could be combined. If the water quality target needs to be more stringent, a tertiary wastewater treatment (e.g. sand filter) could be added and floating wetlands could be installed along the water body for further effectiveness. However, the combination of solutions could be totally different and address mainly microplastic pollution at the source (Figure 30). The final selection of solutions will mainly depend on what combination of solutions can be achieved with the minimum cost. Still, costs and effectiveness will not be the only criteria. The capacities and perceptions of local stakeholders (as well as other practical challenges to adopt certain solutions in a given local context) will all influence the final selection.

Figure 30. Examples of combinations of solutions to water pollution by microplastics from source to tap



Section VIII

Annexes

A. Types of plastics and their use

Plastic type	Plastic use (common applications)	Abbreviation	Market share (per cent)	Specific gravity	Abundance in wastewater[18]
Polymethyl methacrylate or acrylic	Paints, packaging	PMMA		1.09-1.20	++
Acrylonitrile butadiene styrene	Electronics and electrics, car interiors	ABS		1.03-1.11	NR (not reported)
Alkyd	Paints and moulds for casting			1.24-2.01	++
Cellulose acetate	Cigarette filters			1.22-1.24	NR
Ethylene vinyl acetate	Orthotics (medical devices), cigarettes, surfboard and skimboard traction pads, flowers	EVA		0.92-0.95	+
Plexar resin	Bonding materials			0.92	+
Polylactic acid or polylactide	Packaging, cups, mulch film	PLA		1.21-1.43	++
Polyamide (nylon)	Fishing gear, fish farming nets, rope	PA		1.13-1.16	+++
Polyaryl ether	Medical implants, sealing rings, piston parts, pumps, cable insulation	PAE		1.14	+
Polycarbonate	Impact-resistant "glass-like" surfaces	PC		1.2-1.22	+
Polyester	Textile	PES/PEST		1.24-2.30	+++
Polyethylene (or polythene)	Plastic bags, bottles, six-pack rings, gear, cages and pipes for fish farming	PE[19]		0.89-0.98	
Polyethylene terephthalate	Bottles, strapping, gear	PET or PETE	6.5	0.96-1.45	+++
Poly (oxymethylene)	Mechanical gears	POM		1.41	+
Polypropylene	Rope, bottle caps, gear, strapping	PP	18.8	0.83-0.92	++
Polystyrene	Utensils, containers, packaging	PS	7.4	1.04-1.10	++
Polystyrene (expanded)	Bait boxes, floats, cups, expanded packaging	EPS		0.01-1.05	
Polysulfone	Membranes or as co-polymer	PSU		1.24	+
Polytetrafluorethylene (or teflon)	Personal care products	PTFE		2.1-2.3	+
Polyurethane	Insulation	PU/PUR	7.3	1.2	++

18 Refers to relative abundance that is (+) Low, (++) medium and (+++) high; NR means not reported.
19 LDPE and HDPE are Low density PE and High density PE.

Plastic type	Plastic use (common applications)	Abbreviation	Market share (per cent)	Specific gravity	Abundance in wastewater[18]
Polyvinyl acetate	Water-based (latex) paints, adhesives	PVAC		1.19	+
Polyvinyl acrylate	Papermaking, textiles, coatings	PV acrylate			+
Polyvinyl alcohol	Films, adhesives	PVAL		1.19-1.31	++
Polyvinyl chloride	Film, pipe, containers	PVC	10.7	1.16-1.58	+
Polyvinyl ethers	Lubricants	PVE			+
Polyvinyl fluoride	Film	PVF		1.7	+
Silicone	Personal care products, electronics, solar panels, construction, kitchenware			1.1-1.2	+
Styrene butadene rubber	Roofing felt and vehicle tyres	SBR		0.94	NR
Terpene resin	Adhesives and in the preparation of adhesive tapes includes their use in rubber cement and friction tapes			0.98	+
Diverse co-polymers					+

Note: The specific gravity of sea water is approximately 1.02.

References: Sundt *et al.* 2014, Sun *et al.* 2018

B. Plastic breakdown pathways in the environment.

In the environment, when they are in contact with water, light or wind, the degradation of plastics to form microplastics is occurring, contributing to increasing concentrations of microplastics in the aquatic system (Gatidou *et al.* 2018; Klein *et al.* 2018). In principle, synthetic polymers are resistant to environmental influences. However, some mechanisms can aid their degradation and disintegration. These mechanisms include photodegradation, thermooxidative degradation, mechanical degradation, hydrolysis, and biodegradation by microbes (Rhodes 2018). Another mechanism, which will not be discussed, is defragmenting of macroplastics to microplastics directly by animal activity.

Following the degradation process, the plastic polymer is converted to smaller molecular units (e.g. oligomers, monomers or different chemicals) and could eventually incorporate the carbon atoms from the polymer chains into biomolecules or be completely mineralized to CO_2 (Eubeler *et al.* 2009; Rhodes 2018). Different plastics, and different plastic shapes, defragment at different rates, with thinner pieces expected to degrade relatively faster (Sundt *et al.* 2014).

Photodegradation: This occurs as a result of light, usually sunlight in outdoor exposure. Degradation initiated by solar UV radiation is a very efficient mechanism in plastics exposed in air or lying on a beach surface (Andrady 2011). It is essential to note that photodegradation by sunlight is generally the initial event, which primes the material for subsequent thermooxidative degradation (Rhodes 2018).

Mechanical degradation: This can occur through action of abrasive forces, heating/cooling, freezing/thawing and wetting/drying (Duwez and Nysten 2001). Mechanical factors are not predominant during biodegradation process, but mechanical damage can activate it or speed it up (Briassoulis 2005). Mechanical stress acts under field conditions and synergizes with other non-biological parameters (temperature, solar energy radiation and chemicals). Photodegradation and mechanical degradation are the main causes of releases of secondary microplastics to the environment (Sundt *et al.* 2014).

Chemical degradation: Chemical transformation is the other most important parameter in abiotic degradation. Atmospheric pollutants and agrochemicals may interact with polymers changing the macromolecule properties. Chemical degradation process can be divided into two steps:

- Oxidative degradation can be accompanied or induced by photodegradation. It produces free radicals which can cause cross-linking reactions and/or chain scission.

- Hydrolysis - To be separated by H_2O, the polymer must contain a hydrolysable covalent bond, such as esters, ethers, anhydrides, amide, urea (urea), ester amide (urethane), and the like. Hydrolysis depends on parameters such as water activity, temperature, pH and time.

Biodegradation: Biodegradation occurs through the action of living organisms, usually microbes. Generally the polymer biodegradation process can be divided into four steps (Lucas *et al.* 2008):

- Biodeterioration: In the first step, the formation of a microbial biofilm results in surface degradation in which the polymeric material is broken into smaller particles.

- Depolymerisation: Biofilm microorganisms secrete extracellular enzymes, which in turn catalyse the depolymerisation of polymer chains into oligomers, dimers or monomers.

- Bioassimilation: The absorption of microbial cells by small molecules produced in this way and the subsequent production of primary and secondary metabolites.

- Mineralization. In the final step, these metabolites are mineralized and form final products such as CO_2, CH_4, H_2O and N_2 and released to the environment

In general, the defragmentation rate is below 1-3 per cent/year in sediments or water.

C. Plastic breakdown pathways in landfills

It has been estimated that 79 per cent of the plastic wastes ever produced have been stored in landfills (UNEP 2018). The decomposition rate of plastics in landfills is unknown. However, it is understood that these plastics are affected by severe environmental conditions, which are foreseen to influence their behaviour and fragmentation.

- PH from 4.5 to 9;

- High salinity;

- Wide variations in temperatures;

- Biogas generation;

- Physical stress;

- Microbial activity;

According to He *et al.* (2018), most microplastics will remain trapped in the landfill under normal conditions. However, they mention that practices such as landfill mining would make it possible for the microplastics to be reintroduced into the environment.

D. Characteristics of microplastics found in wastewater

Some authors have categorized microplastics into five groups (He *et al.* 2018):

1. Lines: elongated particles with one dimension greater than the other two.

Microfibres are the main contaminants in wastewater. They represent 53 per cent (Sun *et al.* 2018) or 66 per cent (Gies *et al.* 2018) of microplastics in wastewater. However, some of the fibres found there are natural fibres such as cotton and not synthetic. Such natural fibres could exceed 50 per cent of total fibres in a wastewater sample.

2. Fragments: pieces of irregular thick plastic with all three sizes being comparable.

These fragments typically represent 28-29 per cent of the microplastics in wastewater (Sun *et al.* 2018; Gies *et al.* 2018). Fragments could be remnants of microbeads (Raju *et al.* 2018).

3. Pellets: spherical particles. They dominate near industrial areas and typically represent 5 per cent of microplastics in municipal wastewater.

4. Flakes (or films): sheets with their thickness significantly lower than other two dimensions.

5. Foams: particles with a spongy texture. Both fragments and foams are abundant in fishing ports.

The last two groups represent 5 per cent or less of the microplastics in wastewater.

These ratios may vary significantly with local contexts. For example, Magni *et al.* (2019) described a municipal WWTP in Italy where 73 per cent of microplastics were films, 21 per cent were fragments, and only 6 per cent were fibres.

In domestic WWTPs plastic beads found in personal care products and fibres from the washing of polyester or other synthetic textiles are two of the main microplastic inputs (He *et al.* 2018).

So far, 30 kinds of microplastic polymers have been identified in WWTP influent and effluent (Sun *et al.* 2018).

E. Removal of microplastics by wastewater treatment plants – compilation of data

Table 27. Costs of wastewater treatment in developing countries

Treatment stage	Process	Biochemical oxygen demand (BOD) removal (per cent)	Total suspended solids (TSS) removal (per cent)	Investment cost		O&M cost	
				USD per capita	per cent of activated sludge (AS) cost	USD per capita	per cent of activated sludge (AS) cost
Primary	Rotating microscreens	0-30	0-30	3-10	4-10	0.1-0.15	1.9-2.5
Primary	Chemically enhanced primary treatment	70-75	80-90	20-40	20-40	1.5-2.0	25-38
Secondary	Activated sludge (AS)	80-90	80-90	100-150	100	4-8	100
Secondary	Lagoons (waste stabilization ponds)	70-90	70-90	20-40	25-40	0.2-0.4	5-8
Secondary	Mixer aided lagoons (incl. covered anaerobic)	70-95	80-90	20-40	25-40	0.2-0.4	5
Secondary	covered anaerobic + mixer aided lagoons	80-95	80-90	20-50	25-50	0.2-0.4	5
Secondary	Up-flow anaerobic sludge blanket (UASB) reactors	60-75	60-70	20-40	25-50	1.0-1.5	19-25
Secondary	Anaerobic filters	70-80	70-80	10-25	10-25	0.8-1.0	13-20
Secondary	Constructed wetlands	80-90	80-90	20-30	20-30	1.0-1.5	19-25
Secondary	Stabilization reservoir systems	75-95	75-90	30-50	30-50	0.2-0.4	5
Secondary	UASB-anaerobic filter combination	80-90	80-90	20-40	20-40	1.0-1.5	19-25
Secondary	UASB-lagoon combination	80-90	70-80	30-50	30-50	1.0-1.5	19-25
Secondary	Chemically enhanced primary treatment – sand filtration combination	80-90	80-90	40-50	40-50	1.5-2.0	25-38
Secondary	UASB-sand filtration combination	80-90	80-90	30-50	30-50	1.0-1.5	19-25
Secondary	UASB-dissolved air flotation combination	80-90	80-90	30-40	30-40	1.0-1.5	19-25

Reference: Drechsel *et al.* (2015)

Table 28. Costs in USD of water treatment in the United States for different WWTP capacities (in m^3 per day)

Technology	Treatment stage	Capital cost (2012 USD)	Annual O&M (2012 USD): typically 8 per cent of capital cost
Coagulation and flocculation	Primary	log (cost) = 0.222 x $[\log(\text{capacity})]^{1.516}$ + 3.071	log (cost) = 0.347 x $[\log(\text{Capacity})]^{1.448}$ + 2.726
Activated sludge	Secondary	log (cost) = 0.256 x $[\log(\text{capacity})]^{1.556}$ + 4.545	
Trickling filter (US EPA 2020)	Secondary	log (cost) = 96849 x [log(capacity)]0.2801	log (cost) = 494.9 x $[\log(\text{capacity})]^{0.9606}$
Membrane bioreactor	Secondary	log (cost) = 0.569 x $[\log(\text{capacity})]^{1.135}$ + 4.605	log (cost) = 0.639 x $[\log(\text{capacity})]^{1.143}$ + 2.633
Reverse osmosis	Tertiary	log (cost) = 0.966 x $[\log(\text{capacity})]^{0.929}$ + 3.082	log (cost) = 0.534 x $[\log(\text{capacity})]^{1.253}$ + 2.786
Ultrafiltration (membrane-based)	Tertiary	log (cost) = 1.003 x $[\log(\text{capacity})]^{0.830}$ + 3.832	log (cost) = 1.828 x $[\log(\text{capacity})]^{0.598}$ + 1.876
Peroxone	Tertiary	log (cost) = 0.405 x $[\log(\text{capacity})]^{1.428}$ + 4.528	log (cost) = 0.845 x $[\log(\text{capacity})]^{1.057}$ + 2.606
Granular activated carbon	Tertiary	log (cost) = 0.722 x $[\log(\text{capacity})]^{1.023}$ + 3.443	log (cost) = 1.669 x $[\log(\text{capacity})]^{0.559}$ + 2.371

Reference: Guo *et al.* (2014)

Table 29. Construction and operating and maintenance costs for secondary treatment upgrades or new construction in the United States

Location	USD/ year	Type of secondary treatment	Capacity (m^3 per day)	Construction Costs [USD/ (m^3/day)]	Operation & maintenance Costs [USD/ (m^3/day)]	Total costs [USD / (m^3/day)]	Total Costs [USD 2017/ (m^3/day)]
Florida	1998	Conventional	38	5,891	700	6,591	9,896
Florida	1998	Conventional	189	2,436	275	2,710	4,068
Madisonville, Louisiana	2008	Rehabilitated treatment plant	303			3,424	3,891
United States	1998	Conventional	379	1,773	206	1,979	2,972
Cape Cod, Massachusetts	2010	No description given	379	9,246	1,321	10,567	11,859
Livonia, Louisiana	2008	New sewer and treatment plant	568			5,928	6,739
Rosepine	2008	Treatment capacity increase	1,136			1,305	1,485
Pearl River	2008	Rehabilitated treatment plant	1,211			296	335
Calhoun County, Texas	2012	No description given	1,893	2,378	594	2,972	3,167
Brodhead, Kentucky	2015	Extended aeration	2,271	2,087	95	2,182	2,293

Location	USD/ year	Type of secondary treatment	Capacity (m^3 per day)	Construction Costs [USD/ (m^3/day)]	Operation & maintenance Costs [USD/ (m^3/day)]	Total costs [USD / (m^3/day)]	Total Costs [USD 2017/ (m^3/day)]
Brodhead, Kentucky	2015	Oxidation ditch	2,271	2,103	92	2,195	2,306
Brodhead, Kentucky	2015	Sequencing batch reactor	2,271	2,227	92	2,319	2,436
Calhoun County, Texas	2012	No description given	3,785	2,113	528	2,642	2,816
Cape Cod, Massachusetts	2010	No description given	3,785	4,491	528	5,019	5,632
Calhoun County, Texas	2012	No description given	5,678	1,915	423	2,338	2,491
Calhoun County, Texas	2012	No description given	7,571	1,717	264	1,981	2,111
Crowley, Louisiana	2008	Rehabilitated treatment plant	9,350			320	365
Lafayette, Louisiana	2008	Rehabilitated sewer and TP	11,356			2,275	2,586
Matanuska-Susitna Borough, Alaska	2013	Aerated lagoon	14,536	399	29	428	486
Matanuska-Susitna Borough, Alaska	2013	Secondary batch reactor	14,536	444	34	478	502
Mandeville, Louisiana	2008	Treatment capacity increase	30,283			460	523
Average			5,419	2,801	370	2,781	3,284
Standard deviation			7,349	2,327	353	2,488	3,040
Minimum			38	399	29	296	335
Maximum			30,283	9,246	1,321	10,567	11,859

Reference: Modified from Hunter *et al.* (2018)

Table 30. Construction and operating and maintenance costs for tertiary treatment upgrades or new construction

Location	USD/ year	Type of tertiary treatment[20]	Treated wastewater total nitrogen (TN)[20] (mg/litre)	Treated wastewater total phosphorous (TP)[20] (mg/litre)	Capacity (m^3/ day)	Construction costs [USD/(m^3/ day)]	O&M costs [USD/(m^3/ day)]	Total costs [USD/ (m^3/day)]	Total costs [USD 2017/ (m^3/d)]
Florida	1998	Submerged biofilter	2	2	38	112,048	9,236	121,285	182,116
Florida	1998	MLE(2-stage) continuous-flow suspended growth	10	2	38	117,726	13,438	131,164	196,917
Florida	1998	3-stage continuous-flow suspended growth	6	2	38	126,054	15,861	141,915	213,081
Florida	1998	4-stage SBR suspended growth	8	2	38	144,224	12,908	157,132	235,907
Florida	1998	4-stage continuous-flow suspended growth	6	2	38	139,303	21,804	161,107	258,960
Florida	1998	MLE(2-stage) continuous-flow suspended growth	10	2	189	45,501	4,997	50,497	75,822
Florida	1998	3-stage continuous-flow suspended growth	6	2	189	47,469	5,754	53,223	79,910
Florida	1998	4-stage continuous-flow suspended growth	6	2	189	50,422	7,268	57,690	86,610
Florida	1998	4-stage SBR[20] suspended growth	8	2	189	52,769	5,110	57,879	86,913
Texas	2001	Retrofit to existing 2nd Trt[20] - chemical precipitation		2.96	189	59,961	568	60,529	83,695
Florida	1998	Submerged biofilter	2	2	189	64,125	4,580	68,705	103,152

20 TN: total nitrogen; TP: total phosphorus; MLE: modified Ludzack-Ettinger activated sludge process; SBR: sequencing batch reactor; 2nd Trt: Secondary treatment.

Location	USD/ year	Type of tertiary treatment[20]	Treated wastewater total nitrogen (TN)[20] (mg/litre)	Treated wastewater total phosphorous (TP)[20] (mg/litre)	Capacity (m^3/ day)	Construction costs [USD/(m^3/ day)]	O&M costs [USD/(m^3/ day)]	Total costs [USD/ (m^3/day)]	Total costs [USD 2017/ (m^3/d)]
Chesapeake Bay, Maryland	2000	-	<5		379	9,123	265	9,388	13,325
Florida	1998	MLE(2-stage) continuous-flow suspended growth	10	2	379	33,084	3,785	36,870	55,343
Florida	1998	3-stage continuous-flow suspended growth	6	2	379	34,561	3,785	38,346	57,576
Florida	1998	4-stage SBR suspended growth	8	2	379	36,567	3,785	40,352	60,604
Florida	1998	4-stage continuous-flow suspended growth	6	2	379	36,643	4,997	41,640	62,535
Texas	2001	Retrofit to existing 2^{nd} Trt[37] - chemical precipitation		3.52	757	15,634	189	15,823	21,880
Texas	2001	Retrofit to existing 2^{nd} Trt[37] - chemical precipitation		3.14	1,363	10,069	227	10,296	14,233
Texas	2001	Retrofit to existing 2^{nd} Trt[37] - chemical precipitation		3.36	1,703	19,268	265	19,533	26,990
Guste Island, Louisiana	2008	Treatment capacity increase			2,271			33,425	38,006
Texas	2001	Retrofit to existing 2^{nd} Trt[37] - chemical precipitation		2.4	2,461	5,716	76	5,792	8,025
Chesapeake Bay, Maryland	2000		<5		3,785	4,202	1,136	5,337	7,571
United States	2004	1-stage activated sludge		<6	3,785	27,861	3,520	31,381	40,655
United States	2004	2-stage activated sludge		<4	3,785	33,842	4,126	37,968	49,172

Location	USD/ year	Type of tertiary treatment[20]	Treated wastewater total nitrogen (TN)[20] (mg/litre)	Treated wastewater total phosphorous (TP)[20] (mg/litre)	Capacity (m^3/ day)	Construction costs [USD/(m^3/ day)]	O&M costs [USD/(m^3/ day)]	Total costs [USD/ (m^3/day)]	Total costs [USD 2017/ (m^3/d)]
United States	2004	3-stage activated sludge		<3	3,785	36,681	4,429	41,110	53,261
United States	2004	3-stage activated sludge + metal addition		<1	3,785	36,946	5,110	42,056	54,472
United States	2004	3-stage activated sludge + metal addition + tertiary clarifier		<0.35	3,785	38,308	5,337	43,646	56,554
United States	2004	3-stage activated sludge + metal addition + tertiary clarifier + filtration		<0.15	3,785	41,034	5,640	46,674	60,453
United States	2004	3-stage activated sludge + clarifier + Al absorption		<0.10	3,785	42,056	6,057	48,113	62,346
Texas	2001	Retrofit to existing 2nd trt[37] - Chemical precipitation		2.69	11,356	984	76	1,060	1,476
Ansonia, Connecticut	2008	Rehabilitate sewer system, increase level of trt[37]			13,249			53,753	61,097
Matanuska-Susitna Borough, Alaska	2013	Lagoon retrofit			14,536	15,747	416	16,164	16,996
Matanuska-Susitna Borough, Alaska	2013	Secondary batch reactor retrofit			14,536	16,694	492	17,186	18,056
Matanuska-Susitna Borough, Alaska	2013	New construction			14,536	20,063	644	20,706	21,766
St. John the Baptist, Louisiana	2008	New mechanical treatment plant			17,034			12,681	14,422

Location	USD/ year	Type of tertiary treatment[20]	Treated wastewater total nitrogen (TN)[20] (mg/litre)	Treated wastewater total phosphorous (TP)[20] (mg/litre)	Capacity (m^3/ day)	Construction costs [USD/(m^3/ day)]	O&M costs [USD/(m^3/ day)]	Total costs [USD/ (m^3/day)]	Total costs [USD 2017/ (m^3/d)]
Chesapeake Bay, Maryland	2000		<5		37,854	1,855	606	2,461	3,483
United States	2004	1-stage activated sludge		<6	37,854	14,952	1,552	16,504	21,388
United States	2004	2-stage activated sludge		<4	37,854	17,754	1,855	19,608	25,400
United States	2004	3-stage activated sludge		<3	37,854	20,630	2,044	22,675	29,375
United States	2004	3-stage activated sludge + metal addition		<1	37,854	20,706	2,650	23,356	30,245
United States	2004	3-stage activated sludge + metal addition + tertiary clarifier		<0.35	37,854	21,198	2,839	24,037	31,154
United States	2004	3-stage activated sludge + metal addition + tertiary clarifier + filtration		<0.15	37,854	22,220	2,953	25,173	32,630
United States	2004	3-stage activated sludge + clarifier + Al absorption		<0.10	37,854	24,946	3,180	28,126	36,453
Average					10,003	40,474	4,339	44,008	62,559
Standard deviation					14,244	37,075	4,673	40,269	62,797
Minimum					38	984	76	1,060	1,476
Maximum					37,854	144,224	21,804	161,107	258,960

Reference: Modified from Hunter *et al.* (2018)

Table 31. Construction and O&M costs for existing and planned assimilation wetlands in coastal Louisiana (United States)

Location	Capacity (m^3/day)	Construction Costs (USD 2017)	O&M costs (USD 2017)[21]	Total costs (2017) in USD	Total costs [2017/(m^3/day)] in USD
Breaux Bridge	3,785	860,258	62,187	922,445	3,483
Broussard	3,785	387,200	62,187	449,387	1,703
Mandeville	15,142	4,002,859[22]	62,187	4,065,046	4,126
St. Martinville	5,678	1,428,000	62,187	1,490,187	3,748
Luling	13,249	302,000	62,187	364,187	379
Amelia	3,407	186,000	62,187	248,187	1,060
Thibodaux	15,142	2,006,008	62,187	2,068,195	1,968
Riverbend	1,893	400,000	62,187	462,187	3,483
South Vacherie	606	403,000	62,187	465,187	11,016
Average	6,965	746,558	62,187	1,170,556	3,441
Standard deviation	5,851	648,364	-	1,243,381	3,126
Minimum	606	186,000	62,187	248,187	379
Maximum	15,142	2,006,008	62,187	4,065,046	11,016

Reference: Modified from Hunter *et al.* (2018)

F. Wetlands

Table 32. Advantages and limitations of constructed wetlands

Advantages	Limitations
Cost-effective: including installation cost, operation and maintenance cost.	They require larger land areas and are economical relative to other options only where land is affordable.
Technically feasible: no need for continuous operation and maintenance, which are carried out periodically.	Wetland treatment efficiencies may vary seasonally in response to changing environmental conditions, including rainfall and drought.
Resilient: they are able to tolerate fluctuations in flow.	While wetlands can tolerate temporary drawdowns, they cannot withstand compete drying, they require a minimum amount of water.
Environmentally friendly: they do not produce residual biosolids requiring subsequent treatment and disposal.	There is yet no consensus on the optimal design of wetland systems nor is there much information on their long-term performance.
In addition to water quality improvements, they provide other benefits, e.g. provide habitat for many wetland organisms; provide valuable addition to the “green space” and aesthetic enhancement of open spaces and can be built to fit harmoniously into the landscape.	Performance may be less consistent and vary seasonally in response to changing conditions while average performance over the year may be acceptable, wetland treatment cannot be relied upon if effluent quality must meet stringent discharge standards at all times.

References: US EPA (2000a); WWG (2012)

21 O&M costs are the same for all wetlands based on annual wetland monitoring costs and sample analyses.
22 Includes USD 1 million for purchase of land used for assimilation in 2004 adjusted to 2017$.

Table 33. Advantages and limitations of floating wetlands (FWs)

Advantages	Limitations
Design flexibility and ease of management: different sizes and shapes to fit any water body; lightweight, durable and easy to install components. In addition, environmentally friendly design with minimal use of raw materials for reduced carbon footprint, low energy expenses compared to other treatment methods.	Chemical properties of water bodies with FWs can be affected by, for example, the presence of floating mats together with insufficient algal photosynthesis activity because the shading effects of rafts may reduce diffusion of oxygen from the atmosphere, resulting in lowering dissolved oxygen.
Anchoring system incorporated into each module for increased strength. Quick-attach modules can also be reconfigured and/or relocated with minimal effort.	The physical limitations of FWs include raft structure and buoyancy. Like other floating objects, floating rafts are vulnerable to strong waves which could seriously damage them.
Ease of maintenance: FWs are very easy to maintain compared to constructed wetlands (CWs), they can be harvested annually to enhance removal efficiency and avoid releasing nutrients back into the water body; sediments can be easily dredged without excessive damage to the FW plants; floating rafts can be repaired off-site, whereas a CW must be built in place.	Plant species selection is critical, not only to pollutant removal but also to ecosystem integrity. While some invasive species have high nutrient and pollutant uptake rate and grow rapidly, thus enhancing the removal process, they may have adverse effects on the ecosystem and biodiversity. Thus, non-native species and invaders should not be planted on FWs and may need to be weeded out of the FWs to avoid adverse effects to local ecosystems.
Cost-effectiveness: the cost of installing a FW is lower than that of land acquisition and construction of a CW. Further, the cost can be reduced by using recycled materials to construct the floating mats. This is a significant advantage in areas with high land values and limited space.	Biomass accumulation may exceed the buoyancy provided by floating rafts, however, this could be avoided by prediction of the maximum biomass when designing the rafts.
Weather resistant: FWs reported to be resilient to environmental extremes, adjust to external stresses, and recover quickly from ranges of drying and burning regimes. Also, they are minimally affected by the change of water depth, low water transparency, and high sediment concentrations, and these are the typical attributes of degraded water bodies.	FWs occupy open water surface and may block access or reduce available area for lake/pond recreational use

References: Kerr-Upal *et al.* (2000); Billore *et al.* (2009); Kato *et al.* (2009); American Water Works Association (2011)

References and Further Information

Abbott Chalew, T.E., Ajmani, G.S., Huang, H. and Schwab, K.J. (2013). Evaluating nanoparticle breakthrough during drinking water treatment. *Environmental Health Perspectives* 121, 1161-1166. http://doi.org/10.1289/ehp.1306574. Accessed 5 September 2020.

Abidli, S., Toumi, H., Lahbib, Y. and Trigui El Menif, N. (2017). The first evaluation of microplastics in sediments from the complex lagoon-channel of Bizerte (Northern Tunisia). *Water, Air and Soil Pollution* 228(7), 1-10. https://doi.org/10.1007/s11270-017-3439-9. Accessed 5 September 2020.

After Wildlife: A Guide for New Mexico Communities (n.d.). *Debris rack and defectors*. https://afterwildfirenm.org/post-fire-treatments/treatment-descriptions/road-and-trail-treatments/debris-rack-and-deflectors. Accessed 5 September 2020.

Ahmad, J. and EL-Dessouky, H. (2008). Design of a modified low cost treatment system for the recycling and reuse of laundry waste water. Resources, *Conservation and Recycling* 52(7), 973-978. https://doi.org/10.1016/j.resconrec.2008.03.001. Accessed 12 September 2020.

Akarsu, C., Kıdeyş, A.E. and Kumbur, H. (2017). Microplastic threat to aquatic ecosystems of the municipal wastewater treatment plant. *Türk Hijyen ve Deneysel Biyoloji Dergisi* 74(1), 73-78. http://doi.org/10.5505/TURKHIJYEN.2017.36845. Accessed 12 September 2020.

Alimi, O., Farner, J., Hernandez, L. and Tufenkji, N. (2017). Microplastics and nanoplastics in aquatic environments: Aggregation, deposition, and enhanced contaminant transport. *Environmental Science & Technology* 52(4), 1704-1724. http://doi.org/10.1021/acs.est.7b05559. Accessed 12 September 2020.

American Water Works Association (2011). Industry news: Floating islands to help impaired Minneapolis lake. *Journal (American Water Works Association)* 103 (11), 34-37. https://www.jstor.org/stable/23072396. Accessed 12 September 2020.

Andrady, A.L. (2007a). Biodegradability of polymers. In *Physical Properties of Polymers Handbook*. Mark, J.E. (ed.). New York: Springer. 951-964. https://link.springer.com/book/10.1007/978-0-387-69002-5. Accessed 12 September 2020.

Andrady, A.L. (2007b). Ultraviolet radiation and polymers. In *Physical Properties of Polymers Handbook*. Mark, J.E. (ed.). New York: Springer. 857-866. https://link.springer.com/book/10.1007/978-0-387-69002-5. Accessed 12 September 2020.

AZU Water (2015). Laundry wastewater treatment plant. https://wwastewater.azuwater.com/solution/laundry-wastewater-treatment-plant/. Accessed 12 September 2020.

Bauer-Civiello, A., Critchell, K., Hoogenboom, M. and Hamann, M. (2019). Input of plastic debris in an urban tropical river system. *Marine Pollution Bulletin* 144, 235-242. https://doi.org/10.1016/j.marpolbul.2019.04.070. Accessed 12 September 2020.

BBC News (2019). Has Kenya's plastic bag ban worked? 28 August. https://wwastewater.bbc.com/news/world-africa-49421885. Accessed 12 September 2020.

Beims, R. F., Hu, Y., Shui, H. and Xu, C. (Charles) (2020). Hydrothermal liquefaction of biomass to fuels and value-added chemicals: Products applications and challenges to develop large-scale operations. *Biomass and Bioenergy* 135(February), 105510. https://doi.org/10.1016/j.biombioe.2020.105510. Accessed 30 November 2020.

Benioff Ocean Initiative (2019). *River Plastic Pollution: Considerations for Addressing the Leading Source of Marine Debris*. University of California, Santa Barbara. https://boi.ucsb.edu/wp-content/uploads/2019/05/River_Plastic_Pollution_BOI_white_paper.pdf. Accessed 12 September 2020.

Bhomia, R.K., Inglett, P.W. and Reddy, K.R. (2015). Soil and phosphorous accretion rates in sub-tropical wetlands: Everglades Storm Water Treatment Areas as a case example. *Science of The Total Environment* 533, 297-306. https://doi.org/10.1016/j.scitotenv.2015.06.115. Accessed 12 September 2020.

Billore, S.K., Prashant and Sharma, J.K. (2009). Treatment performance of artificial floating reed beds in an experimental mesocosm to improve the water quality of river Kshipra. *Water Science and Technology* 60(11), 2851-2859. https://doi.org/10.2166/wst.2009.731. Accessed 12 September 2020.

Biswas, A.K. and Hartley K. (2017). *Real Solutions for Plastic Problems: Tackling Microplastics Requires Big Policy Proposals*. Asia and the Pacific Policy Society, Policy Forum. 28 September. https://www.policyforum.net/real-solutions-plastic-problems/. Accessed 12 September 2020.

Booth, A.M., Sabbah, I., Angel, D.L., David, E.B. and Javidpour, J. (2020). Solutions for removing microplastic from wastewater treatment plant effluents. *OECD Workshop on Microplastics from Synthetic Textiles in the Environment: Knowledge, Mitigation and Policy*. 11 February. https://www.oecd.org/water/Draft_Agenda_Public_OECD_Workshop_MP_Textile.pdf. Accessed 12 September 2020.

Borne, K.E., Fassman, E.A. and Tanner, C.C. (2013). Floating treatment wetland retrofit to improve stormwater pond performance for suspended solids, copper and zinc. *Ecological Engineering* 54, 173182. https://doi.org/10.1016/j.ecoleng.2013.01.031. Accessed 12 September 2020.

Borne, K.E., Fassman-Beck, E.A., Winston, R.J., Hunt, W.F. and Tanner, C.C. (2015). Implementation and maintenance of floating treatment wetlands for urban storm water management. *Journal of Environmental Engineering* 141(11), 04015030. https://ascelibrary.org/doi/pdf/10.1061/%28ASCE%29EE.1943-7870.0000959. Accessed 12 September 2020. Accessed 12 September 2020.

Boucher, J. and Friot, D. (2017). *Primary Microplastics in the Oceans: A Global Evaluation of Sources*. Gland, Switzerland: International Union for Conservation of Nature (IUCN). https://www.iucn.org/content/primary-microplastics-oceans. Accessed 12 September 2020.

Bouwman, H., Minnaar, K., Bezuidenhout, C. and Verster, C. (2018). *Microplastics in Fresh Water Environments: A Scoping Study.* Report to the Water Research Commission, South Africa. https://www.researchgate.net/publication/327230974_Microplastics_in_freshwater_environments. Accessed 12 September 2020.

Bradley, J.B., Richards, D.L. and Bahner, C.D. (2005). *Debris Control Structures Evaluation and Countermeasures.* Third edition. United States Department of Transportation, Federal Highway Administration. https://www.fhwa.dot.gov/engineering/hydraulics/pubs/04016/hec09.pdf. Accessed 12 September 2020.

Briassoulis, D. (2005). The effects of tensile stress and the agrochemical Vapam on the ageing of low density polyethylene (LDPE) agricultural films. Part I. Mechanical behaviour. *Polymer Degradation and Stability* 88(3), 489-503. https://doi.org/10.1016/j.polymdegradstab.2004.11.021. Accessed 12 September 2020.

Brodin, M., Norin, H., Hanning A.-C., Persson, C. and Okcabol, S. (2018). *Microplastics from Industrial Laundries – A Laboratory Study of Laundry Effluents.* Swedish Environmental Protection Agency. http://www.swedishepa.se/upload/miljoarbete-i-samhallet/miljoarbete-i-sverige/plast/1003-10-report-microplastics-from-industrial-laundries.pdf. Accessed 12 September 2020.

Brophy, J.T., Keith, M.M., Watterson, A., Park, R., Gilbertson, M Maticka-Tyndale, E. *et al.* (2012). Breast cancer risk in relation to occupations with exposure to carcinogens and exposure to carcinogens and endocrine disruptors: A Canadian case-control study. *Environmental Health* 11(1), 87. http:// www.pubmedcentral.nih.gov/articlerender.fcgi?artid=3533941&tool=pmcentrez&rendertype=abstr act. Accessed 11 December 2020.

Browne, M.A., Crump, P., Niven, S.J., Teuten, E., Tonkin, A., Galloway, T. and Thompson, R. (2011). Accumulation of microplastic on shorelines worldwide: Sources and sinks. *Environmental Science & Technology* 45(21), 9175-9179.

CARBIOS (2020). Biorecycling – Principle of Carbios enzymatic biorecycling process. https://carbios.fr/en/technology/biorecycling/. Accessed 13 September 2020.

Carr, S.A. (2017). Sources and dispersive modes of micro-fibers in the environment. *Integrated Environmental Assessment and Management* 13(3), 466-469. https://doi.org/10.1002/ieam.1916. Accessed 13 September 2020.

Carr, S.A., Liu, J. and Tesoro, A.G. (2016). Transport and fate of microplastic particles in wastewater treatment plants. *Water Research* 91, 174-182. https://doi.org/10.1016/j.watres.2016.01.002. Accessed 14 September 2020.

Chang, M. (2015). Reducing microplastics from facial exfoliating cleansers in wastewater through treatment versus consumer product decisions. *Marine Pollution Bulletin* 101(1), 330-333. https://doi.org/10.1016/j.marpolbul.2015.10.074. Accessed 14 September 2020.

Cherelus, G. (2018). New Orleans pulls 46 tons of Mardi Gras beads from storm drains. Reuters, 27 January. https://uk.reuters.com/article/us-usa-mardigras-new-orleans/new-orleans-pulls-46-tons-of-mardi-gras-beads-from-storm-drains-idUKKBN1FF2KJ. Accessed 14 September 2020.

Cheung, P.K. and Fok, L. (2017). Characterization of plastic microbeads in facial scrubs and their estimated emissions in Mainland China. *Water Research* 122, 53-61. https://doi.org/10.1016/j.watres.2017.05.053. Accessed 14 September 2020.

Christopher, J. (2018). Plastic pollution and potential solutions. *Science Progress* 101(3). 207-260. https://doi.org/10.3184/003685018X15294876706211. Accessed 14 September 2020.

Coalition Clean Baltic (2017). *Guidance on Concrete Ways to Reduce Microplastic Inputs from Municipal Storm Water and Wastewater Discharges.* Uppsala, Sweden. https://www.ccb.se/documents/Postkod2017/CCB%20-%20Guidance%20on%20concrete%20ways%20to%20reduce%20microplastics%20in%20stormwater%20and%20sewage.pdf. Accessed 14 September 2020.

Cofie, O., Nikiema, J., Impraim, R., Adamtey, N., Paul, J. and Koné, D. (2016). *Co-composting of Solid Waste and Fecal Sludge for Nutrient and Organic Matter Recovery.* Colombo, Sri Lanka: International Water Management Institute (IWMI). Accessed 30 November 2020.

Cole, M., Lindeque, P., Fileman, E., Halsband, C., Goodhead, R., Moger, J. and Galloway, T.S. (2013). Microplastic ingestion by zooplankton. *Environmental Science & Technology* 47(12), 6646-6655. https://doi.org/10.1021/es400663f. Accessed 14 September 2020.

Cole, M., Lindeque, P., Halsband, C. and Galloway, T.S. (2011). Microplastics as contaminants in the marine environment: A review. *Marine Pollution Bulletin* 62 (12), 2588-2597. https://doi.org/10.1016/j.marpolbul.2011.09.025. Accessed 30 November 2020.

Conley, K., Clum, A., Deepe, J., Lane, H. and Beckingham, B. (2019). Wastewater treatment plants as a source of microplastics to an urban estuary: Removal efficiencies and loading per capita over one year. *Water Research* X, 3, 100030. https://doi.org/10.1016/j.wroa.2019.100030. Accessed 13 September 2020.

Corradini, F., Meza, P., Eguiluz, R., Casado, F., Huerta-Lwanga, E. and Geissen, V. (2019). Evidence of microplastic accumulation in agricultural soils from sewage sludge disposal. *Science of The Total Environment* 671, 411-420. https://doi.org/10.1016/j.scitotenv.2019.03.368. Accessed 13 September 2020.

Cox, K.D., Covernton, G.A., Davies, H.L., Dower, J.F., Juanes, F. and Dudas, S. (2019). Human consumption of microplastics. *Environmental Science & Technology* 53(12), 7068-7074. https://doi.org/10.1021/acs.est.9b01517. Accessed 13 September 2020.

Cui, L., Ouyang, Y., Lou, Q., Yang, F., Chen, Y., Zhu, W. and Luo, S. (2010). Removal of nutrients from wastewater with *Canna indica* L. under different vertical-flow constructed wetland conditions. *Ecological Engineering* 36(8), 1083-1088. https://doi.org/10.1016/j.ecoleng.2010.04.026. Accessed 13 September 2020.

Da Costa, J.P., Paço, A., Santos, P.S., Duarte, A.C. and Rocha-Santos, T. (2018). Microplastics in soils: Assessment, analytics and risks. *Environmental Chemistry* 16(1), 18-30. https://doi.org/10.1071/EN18150. Accessed 13 September 2020.

De Falco, F., Cocca, M., Avella, M. and Thompson, R. (2020). Microfiber release to water, via laundering, and to air via everyday use: A comparison between polyester clothing with differing textile parameters. *Environmental Science & Technology* 54, 3288-3296. Accessed 10 December 2020.

De Falco, F., Gullo, M.P., Gentile, G., Pace, E. di, Cocca, M., Gelabert, L. *et al.* (2018). Evaluation of microplastic release caused by textile washing processes of synthetic fabrics. *Environmental Pollution* 236, 916-925. https://doi.org/10.1016/j.envpol.2017.10.057. Accessed 14 September 2020.

De Gueldre, G. (2020). Perspective of Flanders and EurEau on mitigation of microplastics from textiles through wastewater treatment. *OECD Workshop on Microplastics from Synthetic Textiles in the Environment: Knowledge, Mitigation and Policy..* 11 February. https://www.oecd.org/water/Draft_Agenda_Public_OECD_Workshop_MP_Textile.pdf. Accessed 13 September 2020.

Delva, L., Van Kets, K., Kuzmanovic, M., Demets, R., Hubo, S., Mys, N. et al. (2019). An introductory review: Mechanical recycling of polymers for dummies. *Capture, Plastics Research,* 1-25. https://www.researchgate.net/publication/333390524_AN_INTRODUCTORY_REVIEW_MECHANICAL_RECYCLING_OF_POLYMERS_FOR_DUMMIES. Accessed 30 November 2020.

de Villiers, S. (2019). Microfiber pollution hotspots in river sediments adjacent to South Africa's coastline. *Water SA* 45(1), 97-102. https://doi.org/10.4314/wsa.v45i1.11. Accessed 13 September 2020.

Drechsel, P., Qadir, M. and Wichelns, D. (eds.) (2015). *Wastewater: Economic Asset in an Urbanizing World.* Dordrecht: Springer. https://doi.org/10.1007/978-94-017-9545-6_1. Accessed 14 September 2020.

Driedger, A.G., Dürr, H.H., Mitchell, K. and Van Cappellen, P. (2015). Plastic debris in the Laurentian Great Lakes: A review. *Journal of Great Lakes Research* 41(1), 9-19. https://doi.org/10.1016/j.jglr.2014.12.020. Accessed 14 September 2020.

Dris, R., Gasperi, J., Rocher, V., Saad, M., Renault, N. and Tassin, B. (2015). Microplastic contamination in an urban area: A case study in Greater Paris. *Environmental Chemistry* 12(5), 592-599. https://doi.org/10.1071/EN14167. Accessed 14 September 2020.

Dris, R., Gasperi, J., Saad, M., Mirande, C. and Tassin, B. (2016). Synthetic fibers in atmospheric fallout: A source of microplastics in the environment? *Marine Pollution Bulletin* 104(1-2), 290-293. https://doi.org/10.1016/j.marpolbul.2016.01.006. Accessed 14 September 2020.

Dubaish, F. and Liebezeit, G. (2013). Suspended microplastics and black carbon particles in the Jade System, Southern North Sea. *Water, Air and Soil Pollution* 224(2), 1352. https://doi.org/10.1007/s11270-012-1352-9. Accessed 13 September 2020.

Duwez, A.S. and Nysten, B. (2001). Mapping aging effects on polymer surfaces: Specific detection of additives by chemical force microscopy. *Langmuir* 17(26), 8287-8292. https://doi.org/10.1021/la0113623. Accessed 15 September 2020.

Dyachenko, A., Mitchell, J. and Arsem, N. (2017). Extraction and identification of microplastic particles from secondary wastewater treatment plant (WWTP) effluent. *Analytical Methods* 9(9), 1412-1418. https://doi.org/10.1039/C6AY02397E. Accessed 15 September 2020.

Elastec (2020). Trash and debris boom. https://www.elastec.com/products/floating-boom-barriers/trash-debris-boom/. Accessed 5 September 2020.

Environmental Audit Committee (2019). *Fixing Fashion: Clothing Consumption and Sustainability.* House of Commons Environmental Audit Committee, United Kingdom. https://publications.parliament.uk/pa/cm201719/cmselect/cmenvaud/1952/1952.pdf. Accessed 5 September 2020.

Eriksen, M., Thiel M., Prindiville M. and Kiessling T. (2018). Microplastic: What are the solutions? In *The Handbook of Environmental Chemistry.* Wagner, M. and Lambert, S. (eds.). 58, 273-298. Cham: Springer. https://doi.org/10.1007/978-3-319-61615-5_13. Accessed 5 September 2020.

Essel, R., Engel, L., Carus, M. and Ahrens, R.H. (2015). *Sources of Microplastics Relevant to Marine Protection in Germany.* Umweltbundesamt (Federal Environment Agency). https://d3n8a8pro7vhmx.cloudfront.net/boomerangalliance/pages/509/attachments/original/1481157150/Sources_of_Microplastics_Relevant_to_Marine_Protection_in_German.pdf?1481157150. Accessed 14 September 2020.

Estahbanati, S. and Fahrenfeld, N.L. (2016). Influence of wastewater treatment plant discharges on microplastic concentrations in surface water. *Chemosphere* 162, 277-284. https://doi.org/10.1016/j.chemosphere.2016.07.083. Accessed 14 September 2020.

Eubeler, J.P., Zok, S., Bernhard, M. and Knepper, T.P. (2009). Environmental biodegradation of synthetic polymers I. Test methodologies and procedures. *TrAC Trends in Analytical Chemistry* 28(9), 1057-1072. https://doi.org/10.1016/j.trac.2009.06.007. Accessed 14 September 2020.

European Food Safety Authority (EFSA) (2016). Presence of microplastics and nanoplastics in food, with particular focus on seafood. EFSA Panel on Contaminants in the Food Chain (CONTAM). *EFSA Journal* 14(6), e04501. https://doi.org/10.2903/j.efsa.2016.4501. Accessed 14 September 2020.

Evangeliou, N., Grythe, H., Klimont, Z. Heyes, C., Eckhardt, S., Lopez-Aparicio, S. and Stohl, A. (2020). Atmospheric transport is a major pathway of microplastics to remote regions. *Nature Communications* 11, 3381 (2020). https://doi.org/10.1038/s41467-020-17201-9. Accessed 30 November 2020.

Evoqua Water Technologies (2020). Landfill leachate treatment system significantly reduces operations cost. https://www.evoqua.com/en/case-studies/solid-waste-agency-municipal-anaerobic-bioreactor/. Accessed 30 November 2020.

Garforth, A., Akah, A. and Hernández-Martínez, J. (2013). Hydrocracking of mixed polymer waste, NovaCrack. In 23rd Annual Saudi-Japan Symposium on Catalysts in Petroleum Refining and Petrochemicals, 2-3 December 2013. https://www.escholar.manchester.ac.uk/uk-ac-man-scw:258106. Accessed 14 September 2020.

Gasperi, J., Wright, S.L., Dris, R., Collard, F., Mandin, C., Guerrouache, M. *et al.* (2018). Microplastics in air: Are we breathing it in? *Current Opinion in Environmental Science and Health* 1, 1-5. https://doi.org/10.1016/j.coesh.2017.10.002. Accessed 14 September 2020.

Gatidou, G., Arvaniti, O.S. and Stasinakis, A.S. (2019). Review on the occurrence and fate of microplastics in Sewage Treatment Plants. *Journal of Hazardous Materials* 367: 504-512. https://doi.org/10.1016/j.jhazmat.2018.12.081. Accessed 13 September 2020.

Geyer, R., Jambeck, J.R. and Law, K.L. (2017). Production, use, and fate of all plastics ever made. *Science Advances* 3(7), p.e1700782. https://doi.org/10.1126/sciadv.1700782. Accessed 13 September 2020.

Gies, E.A., LeNoble, J.L., Noël, M., Etemadifar, A., Bishay, F., Hall, E.R. and Ross, P.S. (2018). Retention of microplastics in a major secondary wastewater treatment plant in Vancouver, Canada. *Marine Pollution Bulletin* 133, 553-561. https://doi.org/10.1016/j.marpolbul.2018.06.006. Accessed 13 September 2020.

Gorgoglione, A. (2016). *Control and Modeling Non-Point Source Pollution in Mediterranean Urban Basins.* Ph.D. Thesis, Doctoral Program in Environmental and Territorial Safety and Control; Scuola Interpolitecnica di Dottorato – Politecnico di Bari, Bari, Italy. https://doi.org/ 10.13140/RG.2.1.4883.7520. Accessed 13 September 2020.

Gorgoglione, A. and Torretta, V. (2018). Sustainable management and successful application of constructed wetlands: A critical review. *Sustainability* 10, 3910. https://doi.org/10.3390/su10113910. Accessed 13 September 2020.

Gradus, R., van Koppen, R., Dijkgraaf, E. and Nillesen, P. (2016). *A Cost-Effectiveness Analysis for Incineration or Recycling of Dutch Household Plastics.* Tinbergen Institute, the Netherlands. https://papers.tinbergen.nl/16039.pdf. Accessed 13 September 2020.

Greenway, M. (2017). Stormwater wetlands for the enhancement of environmental ecosystem services: Case studies for two retrofit wetlands in Brisbane, Australia. *Journal of Cleaner Production* 163, S91-S100. https://doi.org/10.1016/j.jclepro.2015.12.081. Accessed 13 September 2020.

Guerranti, C., Martellini, T., Perra, G., Scopetani, C. and Cincinelli, A. (2019). Microplastics in cosmetics: *Environmental issues and needs for global bans. Environmental Toxicology and Pharmacology* 68,75-79. https://doi.org/10.1016/j.etap.2019.03.007. Accessed 14 September 2020.

Gugssa, B.T. (2012). The Cycle of Solid Waste: *A Case Study on the Informal Plastic and Metal Recovery System in Accra.* Master Thesis in Sustainable Development, Uppsala University, Sweden. http://www.diva-portal.org/smash/get/diva2:585668/FULLTEXT01.pdf. Accessed 14 September 2020.

Gündoğdu, S., Çevİk, C., Güzel, E. and KİlercİoĞlu, S. (2018) Microplastics in municipal wastewater treatment plants in Turkey: A comparison of the influent and secondary effluent concentrations. Environmental Monitoring and Assessment 190(11), 626. https://doi.org/10.1007/s10661-018-7010-y. Accessed 14 September 2020.

Guo, T., Englehardt, J. and Wu, T. (2014). Review of cost versus scale: Water and wastewater treatment and reuse processes. *Water Science and Technology* 69(2), 223-234. https://doi.org/10.2166/wst.2013.734. Accessed 14 September 2020.

Gupta, K. (2011). Consumer Responses to Incentives to Reduce Plastic Bag Use: Evidence from a Field Experiment in Urban India. South Asian Network for Development and Environmental Economics (SANDEE) Working Papers. http://www.sandeeonline.org/uploads/documents/publication/954_PUB_WP_65_Kanupriya_Gupta.pdf. Accessed 14 September 2020.

Gupta, S.K. and Singh, G. (2007). Assessment of the efficiency and economic viability of various methods of treatment of sanitary landfill leachate. *Environmental Monitoring and Assessmment* 135, 107-117. https://doi.org/10.1007/s10661-007-9714-2. Accessed 13 September 2020.

Hanning, A.-C. (2020). Looking for microplastic release hotspots from textiles – findings so far. *OECD Workshop on Microplastics from Synthetic Textiles in the Environment: Knowledge, Mitigation and Policy.* 11 February. https://www.oecd.org/water/Draft_Agenda_Public_OECD_Workshop_MP_Textile.pdf. Accessed 12 September 2020.

He, D., Luo, Y., Lu, S., Liu, M., Song, Y. and Lei, L. (2018). Microplastics in soils: Analytical methods, pollution characteristics and ecological risks. *TrAC Trends in Analytical Chemistry* 109, 163-172. https://doi.org/10.1016/j.trac.2018.10.006. Accessed 12 September 2020.

Headley, T.R. and Tanner, C.C. (2012). Constructed wetlands with floating emergent macrophytes: An innovative stormwater treatment technology. *Critical Reviews in Environmental Science and Technology* 42(21), 2261-2310. https://doi.org/10.1080/10643389.2011.574108. Accessed 12 September 2020.

Heberling, M.T., Nietch, C.T., Price, J.I., Thurston, H.W. and Elovitz, M. (2017). *Drinking Water Treatment Plant (DWTP) Costs and Source Water Quality: An Updated Case Study (2013-2016).* American Water Resources Association Annual Conference, 8 November 2017. https://cfpub.epa.gov/si/si_public_record_report.cfm?Lab=NRMRL&dirEntryId=338309. Accessed 12 September 2020.

Henry, B., Laitala, K. and Klepp, I.G. (2019). Microfibres from apparel and home textiles: Prospects for including microplastics in environmental sustainability assessment. *Science of The Total Environment* 652, 483-494. https://doi.org/10.1016/j.scitotenv.2018.10.166. Accessed 14 September 2020.

Herbort, A.F., Sturm, M.T., Fiedler, S., Abkai, G. and Schuhen, K. (2018a). Alkoxy-silyl Induced agglomeration: A new approach for the sustainable removal of microplastic from aquatic systems. *Journal of Polymers and the Environment* 26(11), 4258-4270. https://doi.org/10.1007/s10924-018-1287-3. Accessed 14 September 2020.

Herbort, A.F., Sturm, M.T. and Schuhen, K. (2018b). A new approach for the agglomeration and subsequent removal of polyethylene, polypropylene, and mixtures of both from freshwater systems – A case study. *Environmental Science and Pollution Research* 25(15), 15226-15234. https://doi.org/10.1007/s11356-018-1981-7. Accessed 14 September 2020.

Homolka, Z.A. (2018). *Treatment of Plastic Wastes using Plasma Gasification Technology.* University of Nebraska-Lincoln, United States. Theses Environmental Studies Program. https://digitalcommons.unl.edu/cgi/viewcontent.cgi?article=1081&context=honorstheses. Accessed 14 September 2020.

Hopewell, J., Dvorak, R. and Kosior, E. (2009). Plastics recycling: Challenges and opportunities. *Philosophical Transactions of the Royal Society B: Biological Sciences* 364(1526), 2115-2126. https://doi.org/10.1098/rstb.2008.0311. Accessed 14 September 2020.

Horton, A.A., Svendsen, C., Williams, R.J., Spurgeon, D.J. and Lahive, E. (2017). Large microplastic particles in sediments of tributaries of the River Thames, UK – Abundance, sources and methods for effective quantification. *Marine Pollution Bulletin* 114(1), 218-226. https://doi.org/10.1016/j.marpolbul.2016.09.004. Accessed 14 September 2020.

Hubbard, R.K. (2010). Floating vegetated mats for improving surface water quality. In *Emerging Environmental Technologies,* Vol. II. Shah V. (ed). Springer Netherlands. 211-244. https://www.springer.com/gp/book/9789048133512. Accessed 30 November 2020.

Hunter, R.G., Day, J.W., Lane, R.R., Shaffer, G.P., Day, J.N., Conner, W.H. *et al.* (2018). Using natural wetlands for municipal effluent assimilation: A half-century of experience for the Mississippi River Delta and surrounding environs. In *Multifunctional Wetlands: Pollution Abatement and Other Ecological Services from Natural and Constructed Wetlands.* Nagabhatla, N. and Metcalfe, C.D. (eds.). Cham: Springer. 15-81. https://www.springer.com/gp/book/9783319674155.

International Rubber Study Group (IRSG) (2017). *Rubber Statistical Bulletin* 71, 10-12. http://www.rubberstudy.com/storage/uploads/contentfile/18600/bMEY90Ab2r.pdf. Accessed 5 September 2020.

Jacobs, D.F., Salifu, K.F. and Seifert, J.R., (2005). Growth and nutritional response of hardwood seedlings to controlled-release fertilization at outplanting. *Forest Ecology and Management* 214(1-3), 28-39. https://doi.org/10.1016/j.foreco.2005.03.053. Accessed 14 September 2020.

Jafarinejad, S. (2017). Cost estimation and economical evaluation of three configurations of activated sludge process for a wastewater treatment plant (WWTP) using simulation. *Applied Water Science* 7, 2513-2521. https://doi.org/10.1007/s13201-016-0446-8. Accessed 14 September 2020.

Jambeck, J.R., Geyer, R., Wilcox, C., Siegler, T.R., Perryman, M., Andrady, A. *et al.* (2015). Plastic waste inputs from land into the ocean. *Science* 347(6223), 768-771. https://doi.org/10.1126/science.1260352. Accessed 14 September 2020.

Jarosiewicz, A. and Tomaszewska, M. (2003). Controlled-release NPK fertilizer encapsulated by polymeric membranes. *Journal of Agricultural and Food Chemistry* 51(2),413-417. https://doi.org/10.1021/jf020800o. Accessed 14 September 2020.

JFE Engineering Corporation (2018). *Waste to Energy Plant for Yangon City in Myanmar – Final Report.* City-to-City Collaboration Project for Low Carbon City Development. FY 2017. https://www.env.go.jp/earth/coop/lowcarbon-asia/english/project/data/EN_MMR_2017_03.pdf. Accessed 14 September 2020.

Joint Group of Experts on the Scientific Aspects of Marine Environmental Pollution (GESAMP) (2015). *Sources, Fate and Effects of Microplastics in the Environment: A Global Assessment.* Kershaw, P.J. (ed.). http://www.gesamp.org/publications/reports-and-studies-no-90. Accessed 13 September 2020.

Joint Group of Experts on the Scientific Aspects of Marine Environmental Pollution (GESAMP) (2016). Sources, *Fate and Effects of Microplastics in the Marine Environment: Part Two of a Global Assessment.* Kershaw, P.H. and Rochman C.M. (eds.). http://www.gesamp.org/publications/microplastics-in-the-marine-environment-part-2. Accessed 13 September 2020.

Jönsson, C., Levenstam Arturin, O., Hanning, A.-C., Landin, R., Holmström, E. and Roos, S. (2018). Microplastics shedding from textiles – Developing analytical method for measurement of shed material representing release during domestic washing. *Sustainability* 10, 2457. https://doi.org/10.3390/su10072457. Accessed 13 September 2020.

Kalčíková, G., Alič, B., Skalar, T., Bundschuh, M. and Gotvajn, A.Ž. (2017). Wastewater treatment plant effluents as source of cosmetic polyethylene microbeads to freshwater. *Chemosphere* 188, 25-31. https://doi.org/10.1016/j.chemosphere.2017.08.131. Accessed 13 September 2020.

Karlsson, K. (2006). *Pathways of Pollutants in Stormwater Systems.* Licentiate thesis. Department of Civil and Environmental Engineering, Luleå University of Technology, Sweden. https://www.diva-portal.org/smash/get/diva2:990591/FULLTEXT01.pdf. Accessed 13 September 2020.

Karlsson, K. (2009). *Characterisation of Pollutants in Stormwater Treatment Facilities.* Doctoral thesis. Department of Civil, Mining and Environmental Engineering, Luleå University of Technology, Sweden. http://ltu.diva-portal.org/smash/get/diva2:990416/FULLTEXT01.pdf. Accessed 13 September 2020.

Kato, Y., Takemon, Y. and Hori, M. (2009). Invertebrate assemblages in relation to habitat types on a floating mat in Mizorogaike Pond, Kyoto, Japan. *Limnology* 10, 167-176. https://doi.org/10.1007/s10201-009-0274-8. Accessed 13 September 2020.

Keating, K., Kitchen, A. and Pettit, A. (2014). *Cost Estimation for Culverts – Summary of Evidence.* Environment Agency of the United KIngdom. https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/411174/Cost_estimation_for_culverts.pdf. Accessed 13 September 2020.

Kelkar, V.P., Rolsky, C.B., Pant, A., Green, M.D., Tongay, S. and Halden, R.U. (2019). Chemical and physical changes of microplastics during sterilization by chlorination. *Journal of Water Research* 163, 114871. https://doi.org/10.1016/j.watres.2019.114871. Accessed 13 September 2020.

Kerr-Upal, M., Seasons, M. and Mulamoottil, G. (2000). Retrofitting a storm water management facility with a wetland component. *Journal of Environmental Science and Health, Part A: Toxic/Hazardous Substances and Environmental Engineering* 35(8), 1289-1307. https://doi.org/10.1080/10934520009377037. Accessed 13 September 2020.

Kettenmann, S. (2016). Nationwide ban on plastic microbeads in cosmetics. *Natural Resources and Environment* 31(1): 58-59. https://www.bdlaw.com/publications/nationwide-ban-on-plastic-microbeads-in-cosmetics/. Accessed 13 September 2020.

Klein, S., Dimzon, I.K., Eubeler, J. and Knepper, T.P. (2018). Analysis, occurrence, and degradation of microplastics in the aqueous environment. In *Freshwater Microplastics. The Handbook of Environmental Chemistry*. Wagner, M. and Lambert S. (eds.). Cham: Springer. 58, 51-67. https://doi.org/10.1007/978-3-319-61615-5_3. Accessed 13 September 2020.

Koelmans, A.A., Nor, N.H.M., Hermsen, E., Kooi, M., Mintenig, S.M. and De France, J. (2019). Microplastics in freshwaters and drinking water: Critical review and assessment of data quality. *Water Research* 155, 410-422. https://doi.org/10.1016/j.watres.2019.02.054. Accessed 13 September 2020.

Koop, F. (2019) Recycling plastic could become easier thanks to this new technology. ZME Science, 30 October. https://wwastewater.zmescience.com/science/recycling-plastic-easier-new-technology/. Accessed 13 September 2020.

Kosuth, M., Mason, S.A. and Wattenberg, E.V. (2018). Anthropogenic contamination of tap water, beer, and sea salt. *PLoS One* 13 (4), e0194970. https://doi.org/10.1371/journal.pone.0194970. Accessed 13 September 2020.

Kržan, A. and Zupan, M. (2020). PlanetCare fibre filters: A solution for today. *OECD Workshop on Microplastics from Synthetic Textiles in the Environment: Knowledge, Mitigation and Policy.* 11 February. https://www.oecd.org/water/Draft_Agenda_Public_OECD_Workshop_MP_Textile.pdf. Accessed 12 September 2020.

Lam, C.-S., Ramanathan, S., Carbery, M., Gray, K., Vanka, K.S., Maurin, C. *et al.* (2018). A comprehensive analysis of plastics and microplastic legislation worldwide. *Water, Air and Soil Pollution* 229(11), 1-19. https://doi.org/10.1007/s11270-018-4002-z. Accessed 13 September 2020.

Lambert, S. and Wagner, M. (2018). Microplastics are contaminants of emerging concern in freshwater environments: An overview. In *Freshwater Microplastics. The Handbook of Environmental Chemistry,* Wagner, M. and Lambert S. (eds.). Cham: Springer. 58, 1-23. https://link.springer.com/chapter/10.1007/978-3-319-61615-5_1. Accessed 13 September 2020.

Landis, T.D. and Dumroese, R.K. (2009). Using polymer-coated controlled-release fertilizers in the nursery and after outplanting. *Forest Nursery Notes.* Winter, 5-12. https://www.fs.usda.gov/treesearch/pubs/34172. Accessed 13 September 2020.

Lares, M., Ncibi, M.C., Sillanpää, M. and Sillanpää, M. (2018). Occurrence, identification and removal of microplastic particles and fibers in conventional activated sludge process and advanced MBR technology. *Water Research* 133, 236-246. https://doi.org/10.1016/j.watres.2018.01.049. Accessed 13 September 2020.

Lassen, C., Hansen, S.F., Magnusson, K., Hartmann, N.B., Jensen, P.R., Nielsen, T.G. and Brinch, A. (2015). *Microplastics: Occurrence, Effects and Sources of Releases to the Environment in Denmark.* Danish Environmental Protection Agency. https://www2.mst.dk/Udgiv/publications/2015/10/978-87-93352-80-3.pdf. Accessed 14 September 2020.

Lebreton, L. and Andrady, A. (2019). Future scenarios of global plastic waste generation and disposal. *Palgrave Communications* 5, 6. https://doi.org/10.1057/s41599-018-0212-7. Accessed 14 September 2020.

Lechner, A., Keckeis, H., Lumesberger-Loisl, F., Zens, B., Krusch, R., Tritthart, M. *et al.* (2014). The Danube so colorful: A potpourri of plastic litter outnumbers fish larvae in Europe's second largest river. *Environmental Pollution* 188, 77-181. https://doi.org/10.1016/j.envpol.2014.02.006. Accessed 14 September 2020.

Lee, H. and Kim, Y. (2018). Treatment characteristics of microplastics at biological sewage treatment facilities in Korea. *Marine Pollution Bulletin* 137, 1-8. https://doi.org/10.1016/j.marpolbul.2018.09.050. Accessed 14 September 2020.

Leslie, H.A. (2015). *Plastic in Cosmetics: Are we polluting the Environment through our Personal Care? Plastic Ingredients that Contribute to Marine Microplastic Litter.* Commissioned by the UNEP Global Programme of Action for the Protection of the Marine Environment from Land-based Activities. http://wedocs.unep.org/bitstream/handle/20.500.11822/9664/-Plastic_in_cosmetics_Are_we_polluting_the_environment_through_our_personal_care_-2015Plas.pdf?sequence=3&isAllowed=y. Accessed 14 September 2020.

Leslie, H.A., Brandsma, S.H., van Velzen, M.J. and Vethaak, A.D. (2017). Microplastics en route: Field measurements in the Dutch river delta and Amsterdam canals, wastewater treatment plants, North Sea sediments and biota. *Environment International* 101, 133. https://doi.org/10.1016/j.envint.2017.01.018. Accessed 14 September 2020

Li, X., Chen, L., Mei, Q, Dong, B., Dai, X., Ding, G. and Zeng, E.Y. (2018). Microplastics in sewage sludge from the wastewater treatment plants in China. *Water Research* 142, 75-85. https://doi.org/10.1016/j.watres.2018.05.034. Accessed 14 September 2020.

Li, X.N., Song, H.L., Li, W., Lu, X.W. and Nishimura, O. (2010). An integrated ecological floating-bed employing plant, freshwater clam and biofilm carrier for purification of eutrophic water. *Ecological Engineering* 36(4), 382-390. https://doi.org/10.1016/j.ecoleng.2009.11.004. Accessed 14 September 2020.

Liebezeit, G. and Liebezeit, E. (2014). Synthetic particles as contaminants in German beers. *Food Additives and Contaminants: Part A* 31(9), 1574-1578. https://doi.org/10.1080/19440049.2014.945099. Accessed 14 September 2020.

Lithner, D., Larsson, Å. and Dave, G. (2011). Environmental and health hazard ranking and assessment of plastic polymers based on chemical composition. *Science of The Total Environment,* 409(18), 3309-3324. https://doi.org/10.1016/j.scitotenv.2011.04.038. Accessed 14 September 2020.

Liu, F., Olesen, K.B., Borregaard, A.R. and Vollertsen, J. (2019a). Microplastics in urban and highway stormwater retention ponds. *Science of The Total Environment* 671, 992-1000. https://doi.org/10.1016/j.scitotenv.2019.03.416. Accessed 14 September 2020.

Liu, X., Yuan, W., Di, M., Li, Z. and Wang, J. (2019b). Transfer and fate of microplastics during the conventional activated sludge process in one wastewater treatment plant of China. *Chemical Engineering Journal* 362, 176-182. https://doi.org/10.1016/j.cej.2019.01.033. Accessed 14 September 2020.

Long, Z., Pan, Z., Wang, W., Ren, J., Yu, X., Lin, L. *et al*. (2019). Microplastic abundance, characteristics, and removal in wastewater treatment plants in a coastal city of China. *Water Research* 155, 255-265. https://doi.org/10.1016/j.watres.2019.02.028. Accessed 14 September 2020.

Lucas, N., Bienaime, C., Belloy, C., Quéneudec, M., Silvestre, F. and Nava-Saucedo, J.-E. (2008). Polymer biodegradation: Mechanisms and estimation techniques – A review. *Chemosphere* 73. 429-42. https://doi.org/10.1016/j.chemosphere.2008.06.064. Accessed 14 September 2020.

Luís, I.P. and Spínola, H. (2010). The influence of a voluntary fee in the consumption of plastic bags on supermarkets from Madeira Island (Portugal). *Journal of Environmental Planning and Management* 53(7), 883-889. https://doi.org/10.1080/09640568.2010.490054. Accessed 14 September 2020.

Lusher, A., Hollman, P. and Mendoza-Hill, J. (2017). *Microplastics in Fisheries and Aquaculture: Status of Knowledge on Their Occurrence and Implications for Aquatic Organisms and Food Safety.* Food and Agriculture Organization of the United Nations (FAO) Fisheries and Aquaculture Technical Paper No. 615. http://www.fao.org/3/a-i7677e.pdf. Accessed 14 September 2020.

Lv, X., Dong, Q., Zuo, Z., Liu, Y., Huang, X. and Wu, W.-M. (2019). Microplastics in a municipal wastewater treatment plant: Fate, dynamic distribution, removal efficiencies, and control strategies. *Journal of Cleaner Production* 225, 579-586. https://doi.org/10.1016/j.jclepro.2019.03.321. Accessed 14 September 2020.

Lyn, D.A., Cooper, T.J., Condon, C.A. and Gan, L. (2007). *Factors in Debris Accumulation at Bridge Piers.* Joint Transportation Research Program, Indiana Department of Transportation and Purdue University. West Lafayette, Indiana, United States. https://doi.org/10.5703/1288284313364. Accessed 14 September 2020

Lynn, H., Rech, S. and Samwel-Mantingh, M. (2017). *Plastics, Gender and the Environment: Findings of a Literature Study on the Lifecycle of Plastics and Its Impacts on Women and Men, from Production to Litter.* WECF – Women Engage for a Common Future. Women Engage for a Common Future. https://www.wecf.org/wp-content/uploads/2018/11/PlasticsgenderandtheenvironmentHighRes-min.pdf. Accessed 30 November 2020.

Ma, B., Xue, W., Ding, Y., Hu, C., Liu, H. and Qu, J. (2019a). Removal characteristics of microplastics by Fe-based coagulants during drinking water treatment. *Journal of Environmental Sciences* 78, 267-275. https://doi.org/10.1016/j.jes.2018.10.006. Accessed 14 September 2020.

Ma, B., Xuea, W., Hua, C., Liud H., Qua, J. and Lif, L. (2019b). Characteristics of microplastic removal via coagulation and ultrafiltration during drinking water treatment. *Chemical Engineering Journal* 359, 159-167. https://doi.org/10.1016/j.cej.2018.11.155. Accessed 14 September 2020.

MacDonald, D., Walker, C., Lucke, T., Flipp, R., Covey, K. and Shadforth, P. (2016). *Floating Wetland Treatment System in Residential Development: Assessing the Benefits for Residents, Local Authorities, and Developers.* Ninth International Conference on planning and technologies for sustainable management of water in the city (Novatech 2016). https://www.researchgate.net/publication/306255661. Accessed 14 September 2020.

Magni, S., Binelli, A., Pittura, L., Avio, C.G., Della Torre, C., Parenti, C.C. *et al.* (2019). The fate of microplastics in an Italian Wastewater Treatment Plant. *Science of The Total Environment* 652, 602-610. https://doi.org/10.1016/j.scitotenv.2018.10.269. Accessed 14 September 2020.

Magnusson, K. and Norén, F. (2014). *Screening of Microplastic Particles In and Down-stream a Wastewater Treatment Plant.* IVL Swedish Environmental Research Institute. https://www.diva-portal.org/smash/get/diva2:773505/FULLTEXT01.pdf. Accessed 14 September 2020.

MakeInBusiness (2018). Plastic waste recycling plant – business plan, profit & cost estimation. 10 April. https://makeinbusiness.com/starting-plastic-recycling-plant/. Accessed 14 September 2020.

Marsden Jacob Associates (2016). *Plastic Bags Ban Options – Cost Benefit Analysis. Final Report.* Prepared for the Victorian Department of Environment, Land, Water and Planning, Australia. https://s3.ap-southeast-2.amazonaws.com/hdp.au.prod.app.vic-engage.files/1915/0580/1564/Plastic_Bags_Ban_Options_-_Cost_Benefit_Analysis_Report.pdf. Accessed 14 September 2020.

Marshalls CPM (2019). *Water Management.* https://pdf.archiexpo.com/pdf/cpm-group-ltd/water-management/69767-381643.html. Accessed 15 September 2020.

Mason, S.A., Garneau, D., Sutton, R., Chu, Y., Ehmann, K., Barnes, J. *et al.* (2016). Microplastic pollution is widely detected in US municipal wastewater treatment plant effluent. *Environmental Pollution* 218, 1045-1054. https://doi.org/10.1016/j.envpol.2016.08.056. Accessed 15 September 2020.

Mason, S.A., Welch, V. and Neratko, J. (2018). Synthetic polymer contamination in bottled water. *Frontiers in Chemistry* 6, 407. https://doi.org/10.3389/fchem.2018.00407. Accessed 30 November 2020.

McKinsey and Company and the Ocean Conservancy (2015). *Steaming the Tide: Land-based Strategies for a Plastic-free Ocean.* https://oceanconservancy.org/wp-content/uploads/2017/04/full-report-stemming-the.pdf. Accessed 30 November 2020.

Mehta, S. (2019). New recycling technologies can help solve the plastic waste problem – Giving new life to old plastic. Wood Mackenzie, 13 August. https://www.woodmac.com/news/editorial/new-recycling-technologies-can-help-to-solve-the-plastic-waste-problem/. Accessed 15 September 2020.

Mesdaghinia, A., Nasseri, S., Mahvi, A.H., Tashauoei, H.R. and Hadi, M. (2015). The estimation of per capita loadings of domestic wastewater in Tehran. *Journal of Environmental Health Science and Engineering* 13, 25-32. https://doi.org/10.1186/s40201-015-0174-2. Accessed 14 September 2020.

Michielssen, M.R., Michielssen, E.R., Ni, J. and Duhaime, M. (2016). Fate of microplastics and other small anthropogenic litter (SAL) in wastewater treatment plants depends on unit processes employed. *Environmental Science: Water Research and Technology* 2(6), 1064-1073. https://doi.org/10.1039/C6EW00207B. Accessed 14 September 2020.

Mintenig, S.M., Int-Veen, I., Löder, M.G.J., Primpke, S. and Gerdts, G. (2017). Identification of microplastic in effluents of waste water treatment plants using focal plane array-based micro-Fourier-transform infrared imaging. *Water Research* 108, 365-372. https://doi.org/10.1016/j.watres.2016.11.015. Accessed 14 September 2020.

Mintenig, S.M., Löder, M.G.J., Primpke, S. and Gerdts, G. (2019). Low numbers of microplastics detected in drinking water from ground water sources. *Science of The Total Environment,* 648, 631-635. https://doi.org/10.1016/j.scitotenv.2018.08.178. Accessed 14 September 2020.

Mitsch, W.J. (1992). Landscape design and the role of created, restored and natural riparian wetlands in controlling nonpoint source pollution. *Ecological Engineering* 1, 27-47. https://doi.org/10.1016/0925-8574(92)90024-V. Accessed 14 September 2020.

Mourgkogiannis, N., Kalavrouziotis, I.K. and Karapanagioti, H.K. (2018). Questionnaire-based survey to managers of 101 wastewater treatment plants in Greece confirms their potential as plastic marine litter sources. *Marine Pollution Bulletin* 133, 822-827. https://doi.org/10.1016/j.marpolbul.2018.06.044. Accessed 14 September 2020.

Munir, D., Irfan, M.F. and Usman, M.R. (2018). Hydrocracking of virgin and waste plastics: A detailed review. *Renewable and Sustainable Energy Reviews* 90, 490-515. https://doi.org/10.1016/j.rser.2018.03.034. Accessed 14 September 2020.

Murphy, F., Ewins, C., Carbonnier, F. and Quinn, B. (2016). Wastewater treatment works (WwTW) as a source of microplastics in the aquatic environment. *Environmental Science and Technology* 50(11), 5800-5808. https://doi.org/10.1021/acs.est.5b05416. Accessed 14 September 2020.

Murphy, M. (2017). *Microplastics Expert Workshop Report: Trash Free Waters Dialogue Meeting*. United States Environmental Protection Agency (US EPA) Office of Wetlands, Oceans and Watersheds. https://www.epa.gov/trash-free-waters/microplastics-expert-workshop-report. Accessed 14 September 2020.

Napper, I.E. and Thompson, R.C. (2016). Release of synthetic microplastic plastic fibers from domestic washing machines: Effects of fabric type and washing conditions. *Marine Pollution Bulletin* 112(1/2), 39-45. https://doi.org/10.1016/j.marpolbul.2016.09.025. Accessed 14 September 2020.

Nizzetto, L., Futter, M. and Langaas, S. (2016). Are agricultural soils dumps for microplastics of urban origin? *Environmental Science and Technology* 50(20), 10777-10779.https://doi.org/10.1021/acs.est.6b04140. Accessed 14 September 2020.

Noack, T. (2018). *Costs and Other Considerations for Constructed Wetlands.* Constructed Wetlands Workshop, Caesar Kleburg Wildlife Center, Texas A&M University, Kingsville, Texas, United States, 7 March 2018. https://tamuk-isee.com/wp-content/uploads/2018/03/Costs-and-Other-Considerations-for-Constructed-Wetlands-Tim-Noack.pdf. Accessed 14 September 2020.

Notten, P. (2019). Addressing marine plastics: A systemic approach - Recommendations for action. United Nations Environment Programme. Nairobi, Kenya. https://gefmarineplastics.org/publications/addressing-marine-plastics-recommendations-for-action. Accessed 14 December 2020.

Norwegian Water Institute (2019). Microplastics in Road Dust – *Characteristics, Pathways and Measures*. https://www.miljodirektoratet.no/globalassets/publikasjoner/M959/M959.pdf. Accessed 5 December 2020.

Novotna, K., Cermakova, L., Pivokonska, L., Cajthaml, T. and Pivokonsky, M. (2019). Microplastics in drinking water treatment – Current knowledge and research needs. *Science of The Total Environment* 667, 730-740. https://doi.org/10.1016/j.scitotenv.2019.02.431. Accessed 14 September 2020.

Núñez, L., Fraga, F., Nunez, M.R. and Villanueva, M. (2000). Thermogravimetric study of the decomposition process of the system BADGE (n=0)/1,2 DCH. *Polymer* 41, 4635-4641. https://doi.org/10.1016/S0032-3861(99)00687-4. Accessed 14 September 2020.

Oßmann, B.E., Sarau, G., Holtmannspötter, H., Pischetsrieder, M., Christiansen, S.H. and Dicke, W. (2018). Small-sized microplastics and pigmented particles in bottled mineral water. *Water Research* 141, 307-316. https://doi.org/10.1016/j.watres.2018.05.027. Accessed 14 September 2020.

Pajares, E.M., Valero, L.G. and Sánchez, I.M.R. (2019). Cost of urban wastewater treatment and ecotaxes: Evidence from municipalities in Southern Europe. *Water* 11, 423-435. https://doi.org/10.3390/w11030423. Accessed 14 September 2020.

PETCO (2018). *PET Recycling Company NPC T/A PETCO (Registration Number: 2004/032347/08) – PET Industry Waste Management Plan - Shared-Cost Plan rev00. PETCO NPC, 5 September 2018*. https://static1.squarespace.com/static/54b408b1e4b03957d1610441/t/5ba4cef1f9619a23ee295c46/1537527545092/201809_PETCO+IndWMP+Shared+Cost+Plan+rev00.pdf. Accessed 15 September 2020.

Pinto da Costa, J., Paço, A., Santos, P.S.M., Duarte, A.C. and Rocha-Santos, T. (2019). Microplastics in soils: Assessment, analytics and risks. *Environmental Chemistry* 16(1), 18-30. https://doi.org/10.1071/EN18150. Accessed 14 September 2020.

Pivokonsky, M., Cermakova, L., Novotna, K., Peer, P., Cajthaml, T. and Janda, V. (2018). Occurrence of microplastics in raw and treated drinking water. *Science of The Total Environment* 643, 1644-1651. https://doi.org/10.1016/j.scitotenv.2018.08.102. Accessed 14 September 2020.

Planete Energies (2014). L'incinération : le pouvoir calorifique des ordures. https://www.planete-energies.com/fr/medias/decryptages/l-incineration-le-pouvoir-calorifique-des-ordures#:~:text=la%20production%20de%20chaleur.,thermiques%20par%20tonne%20d'ordures. Accessed 30 November 2020.

Plappally, A.K. and Lienhard, J.H. (2012). Costs for water supply, treatment, end-use and reclamation. *Desalination and Water Treatment* 51(1-3), 200-232. https://www.tandfonline.com/ doi/abs/ 10.1080/19443994.2012.708996?journalCode=tdwt20. Accessed 14 September 2020.

PlasticsEurope (2019). *Plastics – The Facts 2019*. https://www.plasticseurope.org/application/files/9715/7129/9584/FINAL_web_version_Plastics_the_facts2019_14102019.pdf. Accessed 11 December 2020.

PlasticsEurope (2020). Recycling and energy recovery. https://www.plasticseurope.org/en/focus-areas/circular-economy/zero-plastics-landfill/recycling-and-energy-recovery. Accessed 14 September 2020.

Porcu, A., Sollai, S., Marotto, D., Mureddu, M., Ferrara, F. and Pettinau, A. (2019). Techno-economic analysis of a small-scale biomass to-energy BFB gasification-based system. *Energies* 12, 494. https://doi.org/10.3390/en12030494. Accessed 14 September 2020.

Prata, J.C. (2018). Microplastics in wastewater: State of the knowledge on sources, fate and solutions. *Marine Pollution Bulletin* 129(1), 262-265. https://doi.org/10.1016/j.marpolbul.2018.02.046. Accessed 14 September 2020.

Quinn, P. (2018). Vehicle tyres are probably the biggest source of plastic pollution in our rivers and seas, according to a new report commissioned by Friends of the Earth. https://friendsoftheearth.uk/plastics/tyres-and-microplastics-time-reinvent-wheel. Accessed 14 September 2020.

Ragaert, K., Delva, L. and Geem, K.V. (2017). Mechanical and chemical recycling of solid plastic waste. *Waste Management* 69, 24-58. https://doi.org/10.1016/j.wasman.2017.07.044. Accessed 14 September 2020.

Ranieri, E., Gorgoglione, A., Montanaro, C., Iacovelli, A. and Gikas, P. (2014). Removal capacity of BTEX and metals of constructed wetlands under the influence of hydraulic conductivity. *Desalination and Water Treatment* 56, 1-8. https://doi.org/10.1080/19443994.2014.951963. Accessed 14 September 2020.

Rhodes, C.J. (2018). Plastic pollution and potential solutions. *Science Progress* 101(3), 207-260. https://doi.org/10.3184/003685018X15294876706211. Accessed 14 September 2020.

Riggs, C.N. and Naito, C. (2012). *Debris Control for Bridges and Culverts*. Pennsylvania Local Technical Assistance Program (LTAP) Information Sheet No. 152. United States. https://gis.penndot.gov/BPR_pdf_files/Documents/LTAP/TS_152.pdf. Accessed 15 September 2020.

Ritchie, H. and Roser, M. (2018). Plastic pollution. Our World in Data. https://ourworldindata.org/plastic-pollution. Accessed 15 September 2020.

Rollins, S. (2019). *Neighborhood Stormwater Pond Maintenance Log and Resources*. Master of Environmental Studies program, College of Charleston, Clemson Carolina Clear, and Clemson University Cooperative Extension Service, United States. https://www.clemson.edu/extension/water/stormwater-ponds/resources/final_binder_pond_maintenance.pdf. Accessed 15 September 2020.

Ross, P.S. (2020). Tackling microfiber pollution at source a solution oriented partnership across public and private sectors. *OECD Workshop on Microplastics from Synthetic Textiles in the Environment: Knowledge, Mitigation and Policy.* 11 February. https://www.oecd.org/water/Draft_Agenda_Public_OECD_Workshop_MP_Textile.pdf. Accessed 12 September 2020.

Royal Society (2019). *Microplastics in Freshwater and Soil: An Evidence Synthesis*. London. https://royalsociety.org/-/media/policy/projects/microplastics/microplastics-evidence-synthesis-report.pdf. Accessed 30 November 2020.

Ruan, Y., Zhang, K., Wu, C., Wu, R. and Lam, P.K.S. (2019). A preliminary screening of HBCD enantiomers transported by microplastics in wastewater treatment plants. *Science of The Total Environment* 674, 171-178. https://doi.org/10.1016/j.scitotenv.2019.04.007. Accessed 15 September 2020.

Ryberg, M., Laurent, A., Hauschild M. (2018). *Mapping of Global Plastics Value Chain and Plastics Losses to the environment – with a Particular Focus on Marine Environment*. United Nations Environment Programme. Nairobi, Kenya. https://gefmarineplastics.org/publications/mapping-of-global-plastics-value-chain-and-plastics-losses-to-the-environment-with-a-particular-focus-on-marine-environment. Accessed 30 November 2020.

Saeed, T. and Sun, G. (2012). A review on nitrogen and organics removal mechanisms in subsurface flow constructed wetlands: Dependency on environmental parameters, operating conditions and supporting media. *Journal of Environmental Management* 112, 429-448. https://doi.org/10.1016/j.jenvman.2012.08.011. Accessed 15 September 2020.

Salvador Cesa, F., Turra, A. and Baruque-Ramos, J. (2017). Corrigendum to "Synthetic fibers as microplastics in the marine environment: A review from textile perspective with a focus on domestic washings" [Sci. Total Environ. 598 (2017) 1116-1129]. *Science of the Total Environment* 603-604, 836. https://doi.org/10.1016/j.scitotenv.2017.06.179. Accessed 15 September 2020.

SAMCO (2016). How much does a wastewater treatment system cost? (pricing, factors, etc.). 18 May. https://www.samcotech.com/cost-wastewater-treatment-system/. Accessed 15 September 2020.

SAMCO (2019). How much do biological wastewater treatment systems cost? 9 March. https://www.samcotech.com/how-much-do-biological-wastewater-treatment-systems-cost-pricing/SAMCO (2016). How much does a wastewater treatment system cost? (pricing, factors, etc.). 18 May. https://www.samcotech.com/cost-wastewater-treatment-system/. Accessed 15 September 2020.

Sample, D.J.. Wang, C.-Y. and Fox, L. (2013). Innovative Best Management Fact Sheet No. 1, Floating treatment wetlands. Virginia Cooperation Extension. Virginia Polytechnic Institute and State University (Virginia Tech), United States. https://vtechworks.lib.vt.edu/handle/10919/70627. Accessed 15 September 2020.

Saur, T. (2020) Global assessment of microplastic pollution in wastewater treatment plants. *OECD Workshop on Microplastics from Synthetic Textiles in the Environment: Knowledge, Mitigation and Policy.* 11 February. https://www.oecd.org/water/Draft_Agenda_Public_OECD_Workshop_MP_Textile.pdf. Accessed 12 September 2020.

Schnurr, R.E.J., Alboiu, V., Chaudhary, M., Corbett, R.A., Quanz, M.E., Sankar, K. *et al.* (2018). Reducing marine pollution from single-use plastics (SUPs): A review. *Marine Pollution Bulletin.* 137, 157-171. https://doi.org/10.1016/j.marpolbul.2018.10.001. Accessed 15 September 2020.

Schymanski, D., Goldbeck, C., Humpf, H.-U. and Fürst, P. (2018). Analysis of microplastics in water by micro-Raman spectroscopy: Release of plastic particles from different packaging into mineral water. *Water Resources* 129, 154-162. https://doi.org/10.1016/j.watres.2017.11.011. Accessed 15 September 2020.

Science Advice for Policy by European Academies (SAPEA) (2019). *A Scientific Perspective on Microplastics in Nature and Society.* https://www.sapea.info/topics/microplastics/. Accessed 15 September 2020.

Scientific Advice Mechanism of the European Commission (SAM) (2018). *Microplastic Pollution: The Policy Context - Background Paper.* https://ec.europa.eu/research/sam/pdf/topics/microplastic_pollution_policy-context.pdf. Accessed 15 September 2020.

Scott, K.M. (2012). *Investigating Sustainable Solutions for Roadside Gully Pot Management.* Doctoral thesis, University of Hull, United Kingdom. https://hydra.hull.ac.uk/assets/hull:7115a/content. Accessed 15 September 2020.

Sea My Thoughts (2020). Plastic to fight plastic: Can Seabins save our oceans? https://seamythoughts1916708.wordpress.com/2019/02/28/plastic-to-fight-plastic-can-seabins-save-our-oceans/. Accessed 11 August 2020.

Sheeder, S.A. and Johnson, P.A. (2008). *Controlling Debris at Pennsylvania Bridges.* Pennsylvania Department of Transportation, United States. https://gis.penndot.gov/BPR_PDF_FILES/Documents/Research/Complete%20Projects/Design/Controlling%20Debris%20at%20PA%20Bridges.pdf. Accessed 11 August 2020.

Silow N. (2019) New Swedish investigation about taxation of chemicals in clothing and shoes. University of Gothenburg, Sweden. 13 June. https://fram.gu.se/news-events/n/new-swedish-investigation-about-taxation-of-chemicals-in-clothing-and-shoes.cid1632742. Accessed 11 August 2020.

Simon, M., Alst, N.V. and Vollertsen, J. (2018) Quantification of microplastic mass and removal rates at wastewater treatment plants applying Focal Plane Array (FPA)-based Fourier Transform Infrared (FT-IR) imaging. *Water Research* 142, 1-9. https://doi.org/10.1016/j.watres.2018.05.019. Accessed 11 August 2020.

Solis, M. and Silveira, S. (2020) Technologies for chemical recycling of household plastics – A technical review and TRL assessment. *Waste Management* 105, 128-138. https://doi.org/10.1016/j.wasman.2020.01.038. Accessed 11 August 2020.

Statistics Canada (2019). Municipal wastewater systems in Canada, 2017. https://www150.statcan.gc.ca/n1/pub/11-627-m/11-627-m2019023-eng.htm. Accessed 15 August 2020.

Somda, Z., Katuta, F., Gweshe, J., Corner, B., Forsythe, S., Hamilton, M. and Alkenbrack, S. (2013). *Cost of Behavior Change Communication Interventions in Namibia: Mass Media, Community Mobilization and Interpersonal Communications.* United States Agency for International Development (USAID). http://www.healthpolicyplus.com/archive/ns/pubs/hpi/Documents/1557_1_BCC_Cost_GenPop_FINAL.pdf. Accessed 15 August 2020.

Stormwater Partners Washington (2020). Infiltration basin. https://www.stormwaterpartners.com/facilities-infiltration-basin. Accessed 15 August 2020.

Subash, R., Carbery, M., Kuttykattil, A., Senathirajah, K., Subashchandrabose, S.R., Evans, G. and Thavamani, P. (2018). Transport and fate of microplastics in wastewater treatment plants: Implications to environmental health. *Reviews in Environmental Science and Bio/Technology,* 17(4), 637-653. https://doi.org/10.1007/s11157-018-9480-3. Accessed 15 August 2020.

Sun, J., Dai, X., Wang, Q., van Loosdrecht, M.C.M. and Ni, B.-J. (2019). Microplastics in wastewater treatment plants: Detection, occurrence and removal. *Water Research* 152, 21-37. https://doi.org/10.1016/j.watres.2018.12.050. Accessed 15 August 2020.

Sun, X., Liu, T., Zhu, M., Liang, J., Zhao, Y. and Zhang, B. (2018). Retention and characteristics of microplastics in natural zooplankton taxa from the East China Sea. *Science of The Total Environment* 640, 232e242. https://doi.org/10.1016/j.scitotenv.2018.05.308. Accessed 15 August 2020.

Sundt, P., Schulze, P.E. and Syversen, F. (2014). *Sources of Microplastic-pollution to the Marine Environment.* Mepex for the Norwegian Environment Agency. https://www.miljodirektoratet.no/globalassets/publikasjoner/M321/M321.pdf. Accessed 15 August 2020.

Sutton, R., Mason, S.A., Stanek, S.K., Willis-Norton, E., Wren, I.F. and Box, C. (2016). Microplastic contamination in the San Francisco Bay, California, USA. *Marine Pollution Bulletin* 109(1), 230-235. https://doi.org/10.1016/j.marpolbul.2016.05.077. Accessed 15 August 2020.

Swartz, C.D., Swanepoel, G., Welz, P.J., Muanda, C. and Bonga, A. (2017). *NATSURV 8. Water and Wastewater Management in the Laundry Industry (Edition 2).* Water Research Commission of South Africa. https:/doi.org/10.13140/RG.2.2.32149.58086. Accessed 15 August 2020.

Talvitie, J., Heinonen, M., Pääkkönen, J.P., Vahtera, E., Mikola, A., Setälä, O. and Vahala, R. (2015). Do wastewater treatment plants act as a potential point source of microplastics? Preliminary study in the coastal Gulf of Finland, Baltic Sea. *Water Science and Technology* 72(9), 1495-1504. https://doi.org/10.2166/wst.2015.360. Accessed 15 August 2020.

Talvitie, J., Mikola, A., Koistinen, A. and Setälä, O. (2017a). Solutions to microplastic pollution – Removal of microplastics from wastewater effluent with advanced wastewater treatment technologies. *Water Research* 123, 401. https://doi.org/10.1016/j.watres.2017.07.005. Accessed 15 August 2020.

Talvitie, J., Mikola, A., Setälä, O., Heinonen, M. and Koistinen, A. (2017b). How well is micro litter purified from wastewater? - A detailed study on the stepwise removal of micro litter in a tertiary level wastewater treatment plant. *Water Research* 109, 164-172. https://doi.org/10.1016/j.watres.2016.11.046. Accessed 15 August 2020.

Tanner, C.C. and Headley, T.R. (2011). Components of floating emergent macrophyte treatment wetlands influencing removal of storm water pollutants. *Ecological Engineering* 37(1), 474-486. https://doi.org/10.1016/j.ecoleng.2010.12.012. Accessed 30 November 2020.

Textile Exchange (2018). *Preferred Fiber and Materials Market Report 2018*. https://store.textileexchange.org/product/2018-preferred-fiber-and-materials-market-report/. Accessed 30 November 2020.

The Tyre Collective (2020). https://www.thetyrecollective.com. Accessed 13 September 2020.

Thompson, R.C. (2015). Microplastics in the marine environment: Sources, consequences and solutions. In *Marine Anthropogenic Litter*. Bergmann, M., Gutow, L. and Klages, M. (eds.). Cham: Springer. 185-200. https://link.springer.com/book/10.1007/978-3-319-16510-3. Accessed 11 August 2020.

Tiwari, M.K. (2018). *Wastewater Treatment and Recycling*. Kharagpur: Indian Institute of Technology. https://nptel.ac.in/courses/105/105/105105178/. Accessed 11 November 2020.

Tramoy, R., Gasperi, J., Dris, R., Colasse, L., Fisson, C., Sananes, S. *et al.* (2019). Assessment of the plastic inputs from the Seine basin to the sea using statistical and field approaches. *Frontiers in Marine Science* 6, 1-10. https://doi.org/10.3389/fmars.2019.00151. Accessed 11 August 2020.

Tullo, A.H. (2019). Plastic has a problem; is chemical recycling the solution? *Chemical and Engineering News* 97(39). https://cen.acs.org/environment/recycling/Plastic-problem-chemical-recycling-solution/97/i39. Accessed 11 August 2020.

Tullo, A.H. (2018) Should plastics be a source of energy? *Chemical & Engineering News* 96(38). https://cen.acs.org/environment/sustainability/Should-plastics-source-energy/96/i38. Accessed 29 October 2020.

Turchet, T. (2015). L'incinération des déchets. Zero Waste France, 12 March 2015. https://www.zerowastefrance.org/lincineration-des-dechets/ Accessed 29 October 2020.

Tyndall, J. and Bowman, T. (2016). Iowa Nutrient Reduction Strategy Best Management Practice Cost Overview Series: Constructed Wetlands. Department of Ecology and Natural Resource Management, Iowa State University, United States. https://www.nrem.iastate.edu/bmpcosttools/files/page/files/2016%20Cost%20Sheet%20for%20Constructed%20Wetlands.pdf. Accessed 11 August 2020.

Tyler, R.N. (2011). *River Debris: Causes, Impacts, and Mitigation Techniques*. Prepared for the Ocean Renewable Power Company by the Alaska Center for Energy and Power. http://acep.uaf.edu/media/89819/2011_4_13_AHERC-River-Debris-Report.pdf. Accessed 11 August 2020.

Uhl, W., Eftekhardadkhah, M. and Svendsen, C. (2018). *Mapping Microplastic in Norwegian Drinking Water*. Norwegian Water Report. http://www.eureau.org/resources/publications/3100-norsk-vann-report-on-microplastics-in-drinking-water-1/file. Accessed 11 August 2020.

United Nations Environment Programme (UNEP) (2016). *Frontiers 2016 Report: Emerging Issues of Environmental Concern*. https://www.unenvironment.org/resources/frontiers-2016-emerging-issues-environmental-concern. Accessed 11 August 2020.

United Nations Environment Programme (UNEP) (2018). *Single-use Plastics: A Roadmap for Sustainability*. https://www.unenvironment.org/resources/report/single-use-plastics-roadmap-sustainability. Accessed 11 August 2020.

United Nations Environment Programme (UNEP) (2019). *Gender and Waste Nexus. Experiences from Bhutan, Mongolia and Nepal. Nairobi and Osaka*. https://www.unenvironment.org/resources/report/gender-and-waste-nexus-experiences-bhutan-mongolia-and-nepal. Accessed 30 November 2020.

United Nations Environment Programme (UNEP) 2020. Legislative guide on the regulation of single use plastic. United Nations Environment Programme (UNEP)/World Resources Institute (WRI). In press.

United States Environmental Protection Agency (US EPA) (2000a). *Constructed Wetlands Handbook: A Guide to Creating Wetlands for Agricultural Wastewater, Domestic Wastewater, Coal Mine Drainage and Stormwater in the Mid-Atlantic Region*. https://www.epa.gov/sites/production/files/2015-10/documents/constructed-wetlands-handbook.pdf. Accessed 30 November 2020.

United States Environmental Protection Agency (US EPA) (2000b). *Guiding Principles for Constructed Treatment Wetlands: Providing for Water Quality and Wildlife Habitat*. https://www.epa.gov/wetlands/guiding-principles-constructed-treatment-wetlands-providing-water-quality-and-wildlife. Accessed 15 August 2020.

United States Environmental Protection Agency (US EPA) (2000c). *Wastewater Technology Fact Sheet: Free Water Surface Wetlands*. https://www3.epa.gov/npdes/pubs/free_water_surface_wetlands.pdf. Accessed 11 August 2020.

United States Environmental Protection Agency (US EPA) (2002). *Wastewater Technology Fact Sheet: Trickling Filter Nitrification*. https://www3.epa.gov/npdes/pubs/trickling_filt_nitrification.pdf. Accessed 11 August 2020.

United States Environmental Protection Agency (US EPA) (2004). *Wastewater Technology Fact Sheet: Constructed Treatment Wetlands*. https://nepis.epa.gov/Exe/ZyPDF.cgi/30005UPS.PDF?Dockey=30005UPS.PDF. Accessed 11 August 2020.

United States Environmental Protection Agency (US EPA) (2020a). Combined Sewer Overflow Technology Fact Sheet: Chlorine Disinfection. https://www3.epa.gov/npdes/pubs/chlor.pdf. Accessed 15 September 2020.

United States Environmental Protection Agency (US EPA) (2020b). *The Sources and Solutions: Wastewater*. https://www.epa.gov/nutrientpollution/sources-and-solutions-wastewater. Accessed 15 September 2020.

University of Arkansas Community Design Center (2020). *Low Impact Design: A Design Manual for Urban Areas*. http://uacdc.uark.edu/work/low-impact-development-a-design-manual-for-urban-areas. Accessed 5 December 2020.

Verschoor, A.J. (2015). *Towards a Definition of Microplastics: Considerations for the Specification of Physico-chemical Properties*. National Institute for Public Health and the Environment of the Netherlands. https://www.rivm.nl/bibliotheek/rapporten/2015-0116.pdf. Accessed 13 September 2020.

Wagner, M., Scherer, C., Alvarez- Muñoz, D., Brennholt, N., Bourrain, X., Buchinger, S. *et al.* (2014). Microplastics in freshwater ecosystems: What we know and what we need to know. *Environmental Sciences Europe* 26(1), 1-9. https://doi.org/10.1186/s12302-014-0012-7. Accessed 13 September 2020.

Wang, F., Talaue McManus, L. and Xie, R. (eds.) (2019). *Addressing Marine Plastics: A Roadmap to a Circular Economy*. United Nations Environment Programme (UNEP), Nairobi, Kenya. https://gefmarineplastics.org/publications/addressing-marine-plastics-a-roadmap-to-a-circular-economy. Accessed 30 November 2020.

Wang, W., Gao, H., Jin, S., Li, R., Na, G. (2019). The Eco toxicological effects of microplastics on aquatic food web, from primary producer to human: A review. *Ecotoxicology and Environmental Safety* 173, 110-117. https://doi.org/10.1016/j.ecoenv.2019.01.113. Accessed 13 September 2020.

Wang, Z., Taylor, S.E., Sharma, P. and Flury, M. (2018). Poor extraction efficiencies of polystyrene, nano- and microplastics from bio solids and soil. *PLoS One*. 13(11), e0208009. https://doi.org/10.1371/journal.pone.0208009. Accessed 13 September 2020.

Wastewater Gardens (WWG) (2012). *Constructed Wetlands to Treat Wastewater: Framework and Schematic Overview*. Wastewater Gardens International Information Sheet IS20120105. https://www.wastewatergardens.com/pdf/WWG_AboutConstructedWetlands.pdf. Accessed 11 September 2020.

Wiśniowska, E., Moraczewska-Majkut, K. and Nocoń, W. (2018). Efficiency of microplastics removal in selected wastewater treatment plants – preliminary studies. *Desalination and Water Treatment* 134, 316-323. https://doi.org/10.5004/dwt.2018.23418. Accessed 13 September 2020.

World Health Organization (WHO) (2003). *Acrylamide in Drinking Water: Background Document for Development of WHO Guidelines for Drinking-water Quality. Geneva*. https://apps.who.int/iris/bitstream/handle/10665/75373/WHO_SDE_WSH_03.04_71_eng.pdf?sequence=1&isAllowed=y. Accessed 11 September 2020.

World Health Organization (WHO) (2019). *Microplastics in Drinking-water*. Geneva. https://www.who.int/water_sanitation_health/publications/microplastics-in-drinking-water/en/. Accessed 11 September 2020.

Woroniuk, B. and Schalkwyk, J. (1998). *Waste disposal & equality between women and men*. Swedish International Development Cooperation Agency (Sida) Equality Prompt 7. http://www.oecd.org/dac/gender-development/1849277.pdf. Accessed 21 October 2020.

Wright, S. and Kelly, F. (2017). Plastic and human health: A micro issue? *Environmental Science and Technology* 51, 1(12):6634-6647. http://doi.org/10.1021/acs.est.7b00423. Accessed 11 September 2020.

Wright, S., Thompson, R.C. and Galloway, T.S. (2013). The physical impacts of microplastics on marine organisms: A review. *Environmental Pollution* 178, 483-492. https://doi.org/10.1016/j.envpol.2013.02.031. Accessed 11 September 2020.

Xu, X., Hou, Q., Xue, Y., Jian, Y. and Wang, L. (2018). Pollution characteristics and fate of microfibers in the wastewater from textile dyeing wastewater treatment plant. *Water Science and Technology* 78(10), 2046-2054. https://doi.org/10.2166/wst.2018.476. Accessed 11 September 2020.

Xu, X., Hou, Q., Xue, Y., Jian, Y. and Wang, L. (2018). Research progress on the transference and pollution characteristics of microplastics in wastewater treatment plants. *China Environmental Science* 38(11), 4393-4400. http://www.zghjkx.com.cn/EN/abstract/abstract16012.shtml#. Accessed 11 September 2020.

Yang, D.Q., Shi, H.H., Li, L., Li, J.N., Jabeen, K. and Kolandhasamy, P. (2015). Microplastic pollution in table salts from China. *Environmental Science and Technology* 49 (22), 13622-13627. https://doi.org/10.1021/acs.est.5b03163. Accessed 11 September 2020.

Yang, Z., Lü, F., Zhang, H., Wang, W.,Shao, L., Ye, J. and He, P. (2021). Is incineration the terminator of plastics and microplastics? *Journal of Hazardous Materials* 401, 123429. https://doi.org/10.1016/j.jhazmat.2020.123429. Accessed 30 November 2020.

Ziajahromi, S., Drapper, D., Hornbuckle, A., Rintoul, L and Leusch, F. (2020). Microplastic pollution in a stormwater floating treatment wetland: Detection of tyre particles in sediment. *Science of The Total Environment* 713, 136356. https://doi.org/10.1016/j.scitotenv.2019.136356. Accessed 11 September 2020.

Ziajahromi S., Neale, P.A and Leusch, F.D.L. (2016). Wastewater treatment plant effluent as a source of microplastics: Review of the fate, chemical interactions and potential risks to aquatic organisms. *Water Science and Technology* 74(10), 2253-2269. https://doi.org/10.2166/wst.2016.414. Accessed 11 September 2020.

Ziajahromi, S., Neale, P.A., Rintoul, L. and Leusch, F.D.L. (2017). Wastewater treatment plants as a pathway for microplastics: Development of a new approach to sample wastewater-based microplastics. *Water Research* 112, 93. https://doi.org/10.1016/j.watres.2017.01.042. Accessed 11 September 2020.

Zubris, K.A.V. and Richards, B.K. (2005). Synthetic fibers as an indicator of land application of sludge. *Environmental Pollution* 138(2), 201-211. https://doi.org/10.1016/j.envpol.2005.04.013. Accessed 11 September 2020.

Zuccarello, P., Ferrante, M., Cristaldi, A., Copat, C., Grasso, A., Sangregorio, D. *et al.* (2019). Exposure to microplastics (<10 μm) associated to plastic bottles mineral water consumption: The first quantitative study. *Water Research* 157, 365-371. https://doi.org/10.1016/j.watres.2019.03.091. Accessed 11 September 2020.