OSPAR COMMISSION

Protecting and conserving the North-East Atlantic and its resources

Assessment document of land-based inputs of microplastics in the marine environment

Environmental Impact of Human Activities Series

2017

OSPAR Convention

The Convention for the Protection of the Marine Environment of the North-East Atlantic (the “OSPAR Convention”) was opened for signature at the Ministerial Meeting of the former Oslo and Paris Commissions in Paris on 22 September 1992. The Convention entered into force on 25 March 1998. The Contracting Parties are Belgium, Denmark, the European Union, Finland, France, Germany, Iceland, Ireland, Luxembourg, the Netherlands, Norway, Portugal, Spain, Sweden, Switzerland and the United Kingdom.

Convention OSPAR

La Convention pour la protection du milieu marin de l'Atlantique du Nord-Est, dite Convention OSPAR, a été ouverte à la signature à la réunion ministérielle des anciennes Commissions d'Oslo et de Paris, à Paris le 22 septembre 1992. La Convention est entrée en vigueur le 25 mars 1998. Les Parties contractantes sont l'Allemagne, la Belgique, le Danemark, l’Espagne, la Finlande, la France, l’Irlande, l’Islande, le Luxembourg, la Norvège, les Pays-Bas, le Portugal, le Royaume- Uni de Grande Bretagne et d’Irlande du Nord, la Suède, la Suisse et l’Union européenne

Acknowledgement

This report has been prepared by Anja Verschoor assisted by René van Herwijnen, Connie Posthuma, Kristina Klesse and Stefanie Werner (the Netherlands and Germany as lead-countries) with contributions from members for OSPAR ICG-ML actions 46 and 47.

Executive summary

The pollution of the marine environment with plastic (macro) litter and microplastic particles, is regarded as a major global environmental problem. Plastic material is a valuable material in our society and is used in a diverse range of applications, both short and long term, before potentially becoming waste. Loss of plastics to the environment may occur at any stage of the life cycle.

Global plastic production has increased from 50 million tons in 1976 to 315 million tons in 2015, leading to large quantities of mismanaged plastic waste, litter and broken down particles entering the marine environment. In this report, source emissions and potential impacts of microplastics in the OSPAR Maritime Area are described to facilitate prioritization of further actions to mitigate microplastic emissions to the environment. The sources were selected at an international stakeholder conference held by OSPAR in December 2015 in Rotterdam. This report is based on existing literature. It was agreed by the OSPAR Intersessional Correspondence Group on Marine Litter (ICG ML) that recommendations with respect to reduction targets, research needs, and mitigation measures, were outside of the scope of this report.

Sources, pathways and emissions

Microplastics - plastic particles <5mm - may be primary or secondary. Primary microplastics are those originally manufactured to be that small size; secondary microplastics are those that result from the breakdown of larger items.

This document concentrates on the land-based sources and emissions of plastics that are released to the marine environment in particles smaller than 5mm (as part of microlitter), as well as mismanaged plastic waste and litter such as bottles and packaging (as part of macrolitter), which subsequently break down in the sea to form microplastics. The microplastics sources include preproduction pellets, cosmetics, abrasive cleaning agents, rubber infill on artificial sports fields, road runoff of car tyre wear, laundry fibres, and paints. It also covers emissions of larger land-based litter. The estimated source emissions are shown in Figure 1.

Several studies show that sewage treatment plants remove microplastics from waste water streams, with an efficiency up to 99%, depending on the type of the sewerage system and conditions during sampling. The removal is often reached during the primary steps of the treatment process. The actual removal of microplastics from waste water is lower than the reported 99% because of overflows due to a limited capacity of the sewage treatment plants to deal with peak water flows and because not all households are connected to a sewage treatment plant. Sewage treatment plants are considered as part of the distribution pathway and not as a source of microplastics. Therefore, emissions of sewage treatment plants have not been quantified separately.

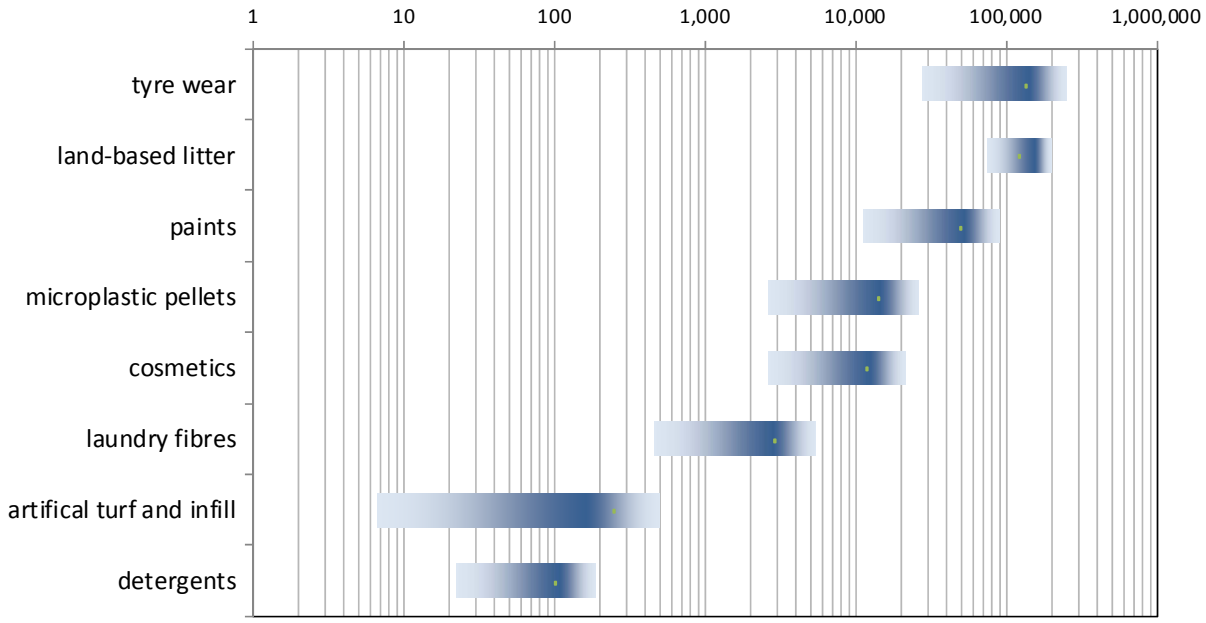


Figure 1: Estimated emissions of microplastics in OSPAR catchments (tonnes / year). The bars represent the uncertainty margins of the emission, white dots represent the midpoint.

The largest sources of microplastics to the OSPAR catchments are tyre wear and (macro) litter with estimated amounts of around 100,000 tons/year. The estimated emissions are difficult to validate with monitoring data because the origin of the microparticles found in monitoring studies can often not be traced back to a particular source.

Environmental concentrations

Microplastics are found in a range of marine and freshwater environments, including the water column, sediments and biota. The highest concentrations of microplastics are found in industrialized and urbanized areas. High concentrations are particularly found in harbours. Many marine species including species consumed by humans, contain microplastics. In fish microplastics are predominantly found in the intestines. There are no signs of accumulation or biomagnification; most of the microplastics are egested. In fish in the North and Baltic seas 10% or less (depending on the species) contain microplastics, whereas in freshwater fish in French rivers, 11-26% of the fish contain microplastics. The microplastic content was around 1-4 microparticles per fish. In mussels and oysters considerably higher amounts were reported than in fish; from 0.1 particles to more than 100 particles per gram wet weight. Microplastics have also been found in a variety of other aquatic species.

Environmental and human impacts

This report also presents an overview of potential impacts of microplastics on the marine ecosystem and on humans, as reported by studies up to 2015. Microplastics can act in two distinct ways. The first mode of action is that they act as particles or fragments that can negatively affect specific functions in aquatic organisms such as breathing, movement or food uptake. The other mode of action is via the associated chemicals. Microplastics contain or can attract chemicals, that could be released again in a different (cleaner) environment or in the organisms body. Whereas common polymers such as polyethylene are quite inert, additives or adsorbed environmental contaminants can be potentially toxic to the organisms. Based on the literature that was studied it was not possible to determine a level for safe environmental concentrations for microplastics.

The presence of microplastics in food could also potentially increase direct exposure of plastic-associated chemicals to humans. However, based on current evidence, the risk to human health appears to be no more significant than via other exposure routes.

Uncertainties

The bulk of the information provided by this report was derived in 2015 by an exhaustive literature search in peer reviewed papers, supplemented with relevant publicly available grey literature published on behalf of national governments, the EU or non-governmental organizations. However, some additional data and information were included later up to spring 2017.

The presented monitoring data suffer by a lack of harmonization of sampling and analytical methods and are therefore difficult to compare. Not all the studies used procedures to minimize the possibility of contamination of samples or techniques to confirm the synthetic nature of the particles. This could have led to overestimations of microplastic counts in certain cases. On the other hand, when specific colouring techniques or harsh purification methods were used, certain types of plastics could have been missed, leading to an underestimation of microplastic counts.

The models used to calculate emissions and distribution of microplastics are a simplification of reality. This has led to several assumptions for inherently variable model parameters and for parameters with data gaps. The reported range in source emissions reflects these uncertainties. Assumptions are described in the chapters and could be refined if better data become available.

Content

APPENDICES

1 Introduction

1.1 Microplastic: a global concern

Plastic products are valuable materials to modern society. Plastic products facilitate our safety, health, comfort and well-being. Plastics form an integral part in all sectors, including application in transportation, packaging, construction, medical services, sport, recreation, and in water, sanitation and energy infrastructure. Plastics offer unprecedented opportunities with respect to strength, functionality, safety, preservability, hygiene and energy efficiency of products in our daily lives. In Europe, around 4-6% of annual oil exploitation is used for plastic production [112].

Figure 2: European plastics demand by polymer type in 2013 [110]

It is therefore no surprise that in todays' world – with its increasing population and prosperity – more and more plastic is being used (see Figure 2 and Figure 3). Plastics are often eco-efficient and the benefits / burden balance for climate is very positive (factors 5 to 9 for all plastics). Use of plastic products is resulting in a substantial savings in other chains. A study by Denkstatt [109] has revealed that the CO_2 emissions caused by producing plastic packaging materials are compensated for with a five to nine fold energy saving during the usage stage (in terms of CO_2 emissions). Other examples are plastic insulating materials, whereby the energy needed to produce them is recovered after 6 months' use. However, current life cycle analyses do not take into account the ocean as a final sink for plastics.

Figure 3: Production of plastics worldwide from 1950 to 2014 (in million metric tons) [Statistica, 2016]

The downside to the use of plastics is that they are at risk of ending up in the natural environment; in fact they are found in large amounts in seas, oceans, rivers and lakes. This is caused by improper waste management, irresponsible human behaviour and also accidental losses or inevitable wear and tear. Seas and oceans are exposed to direct inputs of plastics from marine activities, or indirectly through riverine inputs or by wind. Plastic litter breaks down into microplastics, generally described as particles smaller than 5 mm. It is estimated that each year at least 8 million tonnes of plastic is lost into the environment on a global scale [38].

It is recently within the OSPAR Maritime Area that a sixth gyre with plastics has been discovered. Surface circulation models and field data show that the poleward branch of the Thermohaline Circulation transfers floating litter from the North Atlantic to the Greenland and Barents seas, which are a dead end for this plastic conveyor belt. Given the limited surface transport of plastic that accumulates here and the mechanisms acting for downward transport, the seafloor beneath this Arctic sector is hypothesized to be an important sink of plastic litter [24].

To tackle the problem of a still growing amount of waste and an increasing scarcity of natural resources, the EU supports a zero waste policy and promotes transition to a circular economy. The EU published a road map for a Strategy on plastics in a Circular Economy in January 2017 [34]. The **circular economy** is a generic term for an industrial economy that is, by design or intention, restorative and in which material flows are of two types, biological nutrients, designed to re-enter the biosphere safely, and technical nutrients, which are designed to circulate at high quality without entering the biosphere. Thus, the circular economy aims to address resource scarcity and environmental impacts. The concept is visualized in Figure 4.

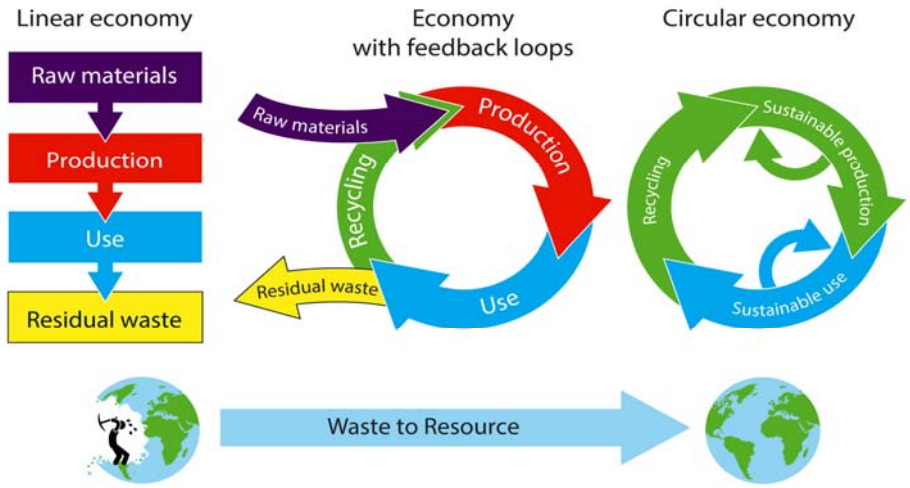


Figure 4: Visualization of the concept of transition to a circular economy: "Closing the loop"

1.2 Regional actions

It is acknowledged that plastic litter is a global problem. In fact plastic litter travels all over the world, with ocean currents, as was demonstrated by the accidental loss in 1992 of almost 30,000 yellow bath ducks from an oceans containership on its way from Hong Kong to the United States. Even now, the bath ducks are found on the shores of Hawaii, Alaska, South America, Australia and the Pacific Northwest; others have been found frozen in Arctic ice. Some have even made their way as far as Scotland and Newfoundland in the Atlantic [60]. Therefore one should not expect that OSPAR alone can solve the problem of plastics including the (size) fraction of microplastics in the North-East Atlantic (including the North Sea). Equal involvement by surrounding partners is needed to achieve results.

Altogether, more than 140 countries participate in at least one of the 18 Regional Seas Action Plans (or conventions) working for sustainable use and management of the ocean and coastal areas (Figure 5). In 12 of the Regional Seas, the Parties have also adopted a legally-binding convention setting out what governments must do to implement the Action Plan. These regional agreements have been effective in influencing participating governments' policies.

Thirteen regional seas programmes have been established under the auspices of the United Nations Environment Programme (UNEP). The Southeast Asia (COBSEA), Eastern Africa (Nairobi Convention), Mediterranean (Barcelona Convention), Northwest Pacific (NOWPAP), West and Central Africa (Abidjan Convention) and Wider Caribbean (Cartagena Convention) programmes are directly administered by UNEP. The Black Sea (Bucharest Convention), Northeast Pacific (Antigua Convention), Red Sea and Gulf of Aden (Jeddah Convention), Persian Gulf and Gulf of Oman (ROPME Sea, Kuwait Convention region), South Asian Seas (SAS, SACEP), Southeast Pacific (CPPS, Lima Convention) and South Pacific (SPREP, Noumea Convention) programmes are independently administered by their regional secretariats.

OSPAR covers the region of the North-East Atlantic. Furthermore, four other regional partner programmes are in place and relevant to achieve Good Environmental Status (GES) in the OSPAR Maritime Area: in the Antarctic (CCMLAR), Arctic (PAME), the Baltic Sea (Helsinki Convention, HELCOM), and Caspian (Tehran Convention). Plans for a new programme in the Southwest Atlantic are under consideration.

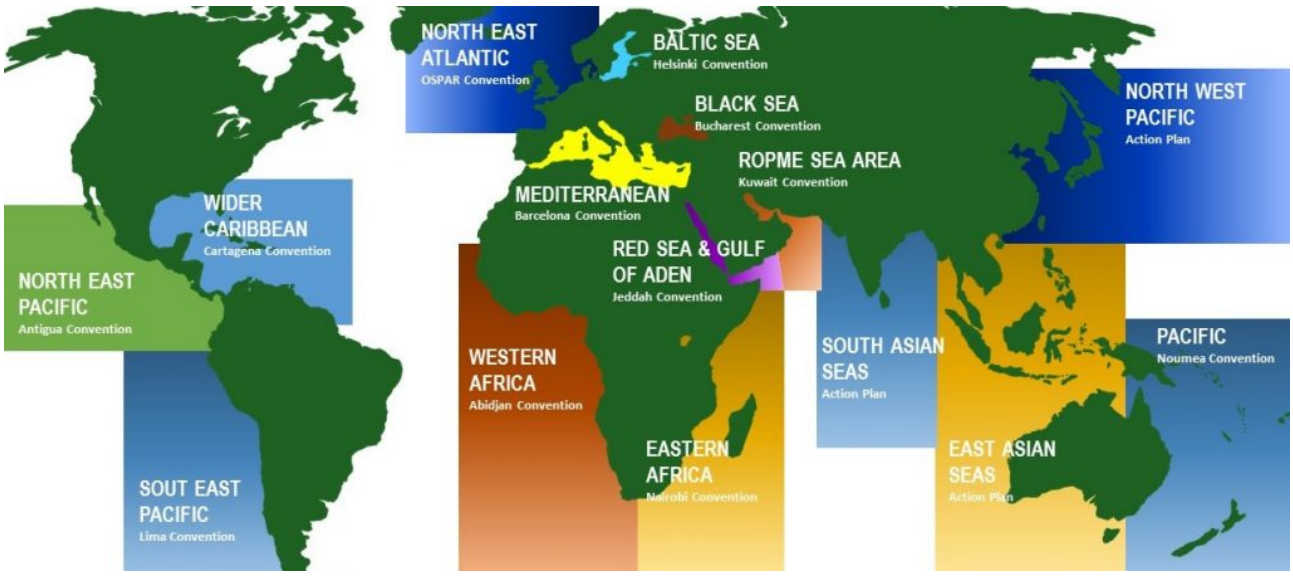


Figure 5: Seas and oceans covered by regional programmes or conventions ©UNEP

1.3 Background within OSPAR

OSPAR Ministers declared in 2010: "We note that quantities of litter in many areas of the North-East Atlantic are unacceptable, and therefore we will continue to develop reduction measures and targets, taking into consideration an ambitious target resulting in a reduction in 2020" (Bergen Statement).

The OSPAR objective with regard to marine litter, as laid down in the Strategy for the protection of the marine environment of the North-East Atlantic for the years 2010-2020, is *"to substantially reduce marine litter in the OSPAR maritime area to levels where properties and quantities do not cause harm to the marine environment"* [103]. The OSPAR objective, elaborated in the Regional Action Plan on Marine Litter (RAP ML) [104], is supportive of the Rio+20 global commitment to "*take action to, by 2025, based on collected scientific data, achieve significant reductions in marine debris to prevent harm to the coastal and marine environment"* and with the 2013 United Nations General Assembly resolution A/RES/68/70 in which States noted concern on marine litter.

The OSPAR objective is also in line with the definition of Descriptor 10 of the Marine Strategy Framework Directive (MSFD)[33], where GES can be regarded to be achieved, when "*properties and quantities of marine litter do not cause harm to the coastal and marine environment"*. It will also support the achievement of an EU-wide "quantitative reduction headline target" for marine litter, as agreed in the 7th Environment Action Programme [3]. The European Commission aims for a reduction goal of 30% until 2020 for litter findings on beaches and lost or abandoned fishing gear at sea. This goal is supposed to be part of the EU Plastics Strategy [34].

The sources of marine litter are diverse and ocean dynamics make marine litter a transboundary issue requiring collective action. Therefore, the OSPAR RAP ML aims to implement the commitments in the

North-East Atlantic Environment Strategy as well as to coordinate actions to deliver GES across the MSFD descriptors. Moreover, the RAP is contributing to the UNEP GPA[1] Global Partnership on Marine Litter, a global framework for prevention and management of marine litter, and the Honolulu Strategy developed at the 5th International Marine Debris Conference [95]. In that sense, the RAP for prevention and management of marine litter can be seen as an exemplar for regional effort supporting multiple regional and global commitments. Other bodies, such as the UN, G20 and G7, also developed goals and action plans; they recognize the urgent need for action to prevent and reduce marine litter in order to preserve human health and marine and coastal ecosystems, and mitigate marine litter's economic costs and impacts.

The OSPAR RAP ML in the North-East Atlantic has been adopted by OSPAR Contracting Parties as an 'OSPAR Other Agreement' and is designed as a flexible tool providing a set of actions to address marine litter from major sea-based and –land-based sources. It contains three types of actions:

1) Common OSPAR actions: actions requiring collective activity within the framework of the OSPAR Commission through, where applicable, OSPAR measures (i.e. Decisions or Recommendations) and / or other agreements such as guidelines and background documents;

2) Actions to be raised with other international organizations and competent authorities; and

3) Actions that Contracting Parties should consider in their national programmes of measures, including those under the MSFD. The approach regarding these national actions is based around the core principle that the RAP ML allows Contracting Parties to identify which of the measures and actions listed they have already taken forward (e.g. as a result of existing or planned national or European legislation or other initiatives) and to consider additional ones needed to further combat marine litter. It therefore provides guidance to Contracting Parties and a framework for regional cooperation.

All actions in the RAP ML aim to prevent further introductions and remove existing amounts of marine litter and will therefore potentially also lead to a reduction of microplastics originating from degradation. However, two collective actions in the RAP ML[104] deal directly with microplastics - Action 46 and Action 47:

> Action 46: Evaluate all products and processes that include primary microplastics and act, if appropriate, to reduce their impact on the marine environment.
>
> Action 47: Engage with all appropriate sectors (manufacturing, retail etc.) to explore the possibility of a voluntary agreement to phase out the use of microplastics as a component in personal care and cosmetic products. Should a voluntary agreement prove not to be sufficient, prepare a proposal for OSPAR to call on the EU to introduce appropriate measures to achieve a 100% phasing out of microplastics in personal care and cosmetic products.

1.4 Focus of this assessment document

This report gives an overview of potential sources and pathways of microplastics (Chapter 2), and summarizes the concentrations found in the marine environment (with a focus on the OSPAR Maritime Area) and describes potential impacts (Chapter 3). A first draft was written in 2015 based on an exhaustive

[1] Global Programm of Action for the Protection of the Marine Environment from land-based activities, from: http://www.unep.org/gpa/gpml/gpml.asp

literature search in peer reviewed papers, supplemented with relevant publicly available grey literature published on behalf of national governments, the EU or non-governmental organizations. The draft has been discussed at the meeting of OSPAR's Intersessional Correspondence Group on Marine Litter (ICG-ML) in November 2015, and was subsequently further refined and supplemented with new information and key publications, especially on topics for which data were scarce. The final draft was discussed by ICG-ML in May 2017 and comments have been processed in the final version of this assessment document.

1.5 Uncertainties

In the rapidly developing field of microplastics research new information is generated at a high rate. The bulk of the information provided by this report was derived in 2015 by an exhaustive literature search in peer reviewed papers, supplemented with relevant publicly available grey literature published on behalf of national governments, the EU or non-governmental organizations. However, some additional data and information were included after 2015, up until spring 2017.

The monitoring data considered within this report suffer a lack of harmonization in terms of sampling and analytical methods, and were therefore difficult to compare. Not all studies considered for this report used procedures to minimize the possibility of contamination of samples, or techniques to confirm the synthetic nature of the particles. This could have led to overestimations of microplastic counts in certain cases. On the other hand, when specific colouring techniques or harsh purification methods were used, certain types of plastics could have been missed, leading to an underestimation of microplastic counts.

The models used to calculate emissions and distribution of microplastics are a simplification of reality. This has led to several assumptions for inherently variable model parameters and for parameters with data gaps. The reported range in source emissions reflects these uncertainties. Assumptions are described in the chapters and could be refined if better data become available.

2 Identification of sources and pathways of microplastics to the marine environment

This chapter gives a description of microplastics (section 2.1) and identifies the most important microplastic sources. The microplastics released from these sources are quantified (section 2.3) and distribution pathways in the environment are described (section 2.4).

2.1 Definition and selection of sources

A legally binding definition of microplastics is not existent. Different "definitions" are used, depending on the purposes of a study, the available sampling and analytical equipment and the regulatory context. GESAMP gives a nice description of the development of plastics and microplastics terminology. At present, the lack of an agreed nomenclature, together with practical difficulties in sampling and measuring different size ranges in the field, has encouraged the widespread adoption of microplastics as a generic term for 'small' pieces of plastic [48].

A broad description of microplastics includes solid man-made materials that are made from fossil-based, bio-based or even mineral-based polymers. It concerns particles that are smaller than 5 mm. Plastics, rubbers, and other solid, insoluble polymers are considered as potential sources of microplastics. Physico-chemical specifications of the criteria for materials to be identified as microplastics are described in Appendix 1.

Further specification of the definition might be necessary once legal restrictions are considered. Verschoor [151] argues that polymers that are soluble or biodegradable in water **under ambient conditions** should not be considered as microplastics, because they change from solid particle to dissolved molecules. It should be noted that being biodegradable in an optimized composting installation does not mean that the plastics are degradable under natural (marine) conditions. Also, the specification of a lower size boundary for microplastics should be considered. Options for the definition of size boundaries, biodegradability, chemical composition and physical state should be described for further discussion within Europe [151].

Specification of the definition is necessary to enable law enforcement and to provide legal certainty to companies that produce or formulate microplastics. Moreover, a clear definition facilitates the harmonisation of monitoring methods and the comparability of studies. Currently, a definition is discussed in ISO TC 61, which is focussed on harmonisation of monitoring and analysis of microplastics. Non-Governmental Organisations (NGOs) increasingly use the phrase microplastics for dissolved fluid and waxy polymers. This demonstrates the difficulty to establish a lower limit and to define the potential additional threats to the marine environment. To avoid difficulties in interpretation of the term microplastics (especially the term plastic) alternative terms have been suggested such as polymer microlitter, man-made macro-molecular materials or micro polymers. However, this is only part of the solution, question about size, physical state and origin still need to be specified. In the present report, the term microplastics is used in the broad sense and what is meant for each individual source is specified.

There is an almost endless and still growing list of polymers and it is impossible to list them all in this background document. Tailor-made polymers are made by adjustment of the composition and molecular weight distribution of the molecules constituting the polymer, which has led to an enormous versatility of polymeric materials.

Being bio-based, does not necessarily make the plastic biodegradable. Many bio-based resins such as bio-PE or bio-PET are substitutes for the conventional plastic resins with the same structure. They are

developed to mirror the properties of their conventional counterparts to allow similar lifetime, applications and recycling capabilities. Therefore prevention by avoiding loss in to the environment is a primary goal. Degradation of lost bio-based materials is still a loss of resources at minimum and in addition a potential threat to the marine environment and therefore should not be considered as a solution.

This report focuses on the plastics with the highest use in Europe (EU28+Norway and Switzerland). The most widely used polymers are polyethylene (29.4%), polypropylene (19.1%), polyvinylchloride (10.1%), polyurethane (7.5%), polyethylene terephthalate (PET, 7.1%), polystyrene (6.9%) and others (19.1%) [111,112]. Tyre rubber (styrene butadiene rubber), fibres and paints are not covered by the statistics of Plastics Europe, but are also included in the assessment, because they demand high volumes of synthetic polymers that may significantly contribute to environmental exposure to polymers.

The distinction between primary and secondary microplastics is based on whether the particles were originally manufactured to be that size (primary) or whether they have resulted from the breakdown of larger items (secondary) [48].

Primary microplastics are intentionally engineered and added to products or used in production processes, for example exfoliating microbeads in cosmetic products, industrial 'scrubbers' used to blast clean surfaces, or plastic powders used in moulding. In addition, resin pellets, typically around 5 mm in diameter, are widely used during plastics manufacturing and transport of the basic resin 'feedstock' prior to production of plastic products.

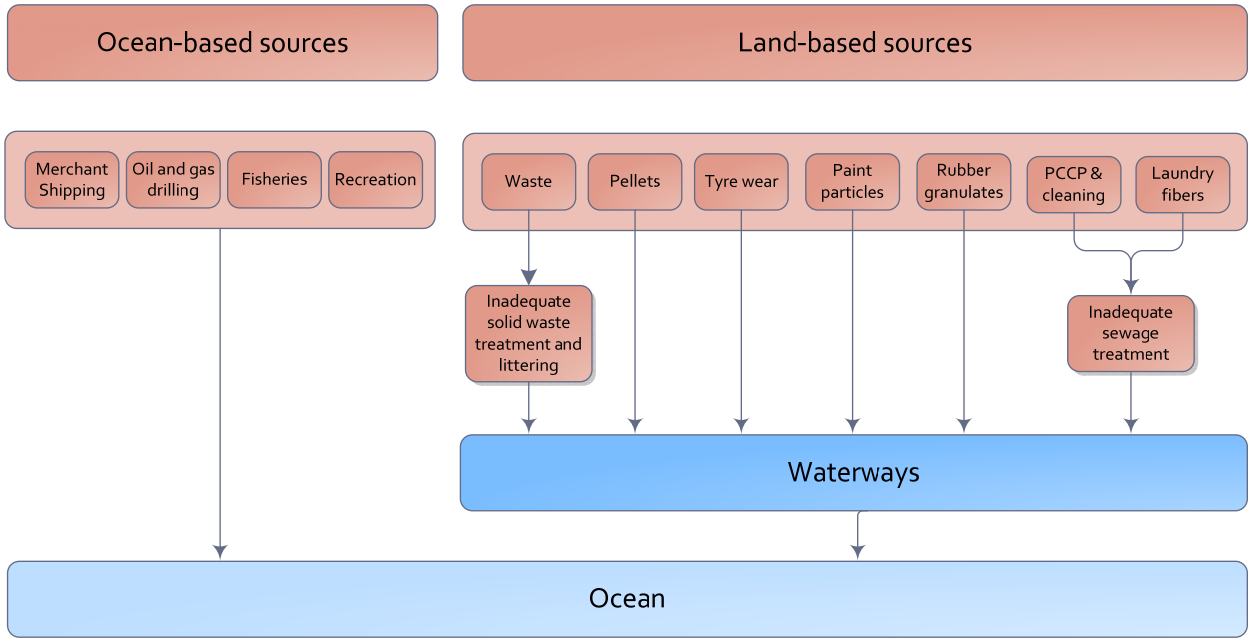
Secondary microplastics are microplastics that are created intentionally or unintentionally during production, use, or waste phases, but also through the littering of plastic items. An example of intentional creation of secondary microplastics is the breakdown of so-called OXO-degradable plastic bags. These bags are decomposed to invisible small plastic fragments, i.e. microplastics. Note that this process does not really reduce the amount of plastic in the environment and increases the bio-availability of plastics. Even the use of fully biodegradable plastics, used as packaging material or bags, is not the solution for behavioural littering. These bags are degradable in an industrial composting facility, but under natural outdoor conditions their biodegradability is poor. Unintentional creation of secondary microplastics is the result of breakdown and wear of many consumer goods, for instance the wearing of textiles, paints and car tyres. The distinction between primary and secondary microplastics is useful because it can help to indicate potential sources and identify mitigation measures to reduce their input to the environment.

Considerable progress has been made in the determination of the amount and location of plastic litter in our seas. How much plastics enter our seas and the relative contribution of different sources is more uncertain. Any plastic product is a potential source of microplastics, but the actual amount of microplastics released to the environment depends on how it is produced, transported, used and disposed of.

One common method is to classify marine litter sources as either land-based or ocean-based, depending on where the litter entered the sea. The MSFD defines marine litter as "any persistent, manufactured or processed solid material discarded, disposed of or abandoned in the (marine and coastal) environment" [45]. Plastic marine litter will contribute significantly to the release of microplastics in the marine environment, when nothing is done to remove the existing plastic mass and reduce the influx of litter. It is often stated that approximately 80% of marine litter arises from land-based sources and the remaining 20% come from sea-based sources [134-136]. This value has been repeated frequently with reference to UNEP and is used as a global average. Regional differences can lead to much higher or much lower percentages. It is likely that in high-income areas, such as the North Sea and the wider North-East Atlantic, have lower input of land-based litter than the lower income countries in, for example, Africa and South-East Asia [62]. Sea-based sources are

not considered within this report; only the emissions of the most important land-based sources of marine microplastics are estimated. These sources were identified at an international OSPAR stakeholder conference in Rotterdam in December 2015.

Figure 6: Overview of sources and pathways of microplastics to the marine environment. Only land-based source emissions are quantified in this report.



The following sources are addressed in this OSPAR document:

1. Land-based macro-litter
2. Preproduction pellets
3. Laundry fibres
4. Tyre wear
5. Rubber infill for artificial turf pitches
6. Paint wear and abrasion from buildings, shipyards and marinas
7. Personal care and cosmetic products
8. Cleaning agents

It should be mentioned that volume estimations and correlation are of indicative value and do have considerable uncertainties. Materials are used in diverse long term or short term applications before becoming waste, leading to a complex latency pattern of the source stream.

2.2 General methodology for quantification of sources

All sources of microplastics are related to human activities. The input of most of the sources to the aquatic environment has therefore been related to the population density. The emission of preproduction pellets has been related to the total plastic demand per country. Plastic demand refers to the amount of plastics used by industries.

For population density estimations there are three possibilities:

1. **Population in OSPAR countries:** The application of these data can be justified because potential measures will be taken at the national level. This report uses data from 2015 [138].
2. **Population in the catchments of the OSPAR Maritime Area:** This approach gives a better estimate of microplastics inputs to the North-East Atlantic and the North Sea than national population densities. Not all the catchments in Spain, France, Switzerland, Germany, Finland, Norway and Sweden discharge to the OSPAR Maritime Area. Moreover, some countries that do not join OSPAR have catchments that do discharge to the OSPAR seas; Russia, Greenland, Czech Republic and Liechtenstein also partly discharge to the OSPAR Maritime Area. The population in OSPAR catchments reported in the year 2000 [102] was used in combination with total population densities in the same year [138], and updated to reflect population densities in 2015. More details are provided in Appendix 2.
3. **Population in the coastal zones** of the OSPAR Maritime Area: This is useful for microplastics sources that are transported over relative short distances or that are distributed by wind. This approach has been applied for land-based litter. The population in the coastal zones (50 km zone) of 192 countries in the world has been determined by GIS-technique and described by Jambeck et al. [62], with a reference year of 2010. These figures have been extrapolated to 2015 and restricted to coastal zones in the OSPAR Maritime Area only. This was done by multiplication of the percentage population in the catchment and the percentage population in the coastal zone. For example, in Spain 52% of the population live in catchment areas that discharge into the OSPAR Maritime Area, whereas 49% of the population live in coastal zones. Therefore, the people that live in coastal areas of catchments that discharge to the OSPAR Maritime Area is estimated to be 52%*49%=25%. More details are provided in Appendix 3.

Demographic and market information for each country of the OSPAR Maritime Area is presented in Table 1. Although it seems reasonable to assume that land-based microplastics emissions will be partly captured or deposited on river beds or banks, this report does not account for this “reduction factor”. Given the uncertainties and lack of knowledge involved in accounting for transport and deposition processes in streams, we have chosen to estimate amounts that can potentially reach the ocean. This means that the potential input in the marine system is set equal to the amount of plastics that enters the rivers.

For plastic litter however, it is assumed that 15-40% of the emissions are captured along rivers, based on the study of Jambeck et al. [62].

Table 1: Demographic and market information of the OSPAR Maritime Area for the reference year 2015

Country	Total population [138]	Estimated population in catchment area of OSPAR Maritime Area		Estimated population in coastal zone of catchments that discharge to OSPAR Maritime Area		Industrial plastic demand [112]
		%		%		[million tonnes]
Contracting Parties						
Belgium	11,299,190	100%	11,299,190	43.00%	4,858,652	2.0
Denmark	5,669,080	41%	2,324,323	40%	2,254,593	0.4
Finland	5,503,460	0.2%	11,007	0.11%	6,054	0.4
France	64,395,350	71%	45,720,699	19%	12,344,589	4.6
Germany	80,688,540	89%	71,812,801	10%	7,899,408	12.0
Iceland	329,430	100%	329,430	92%	303,076	0.0
Ireland	4,688,470	100%	4,688,470	81%	3,797,661	0.2

Luxembourg	605,110	100%	605,110	0%	0	0.1
Norway	5,210,970	73%	3,804,008	61%	3,195,367	0.1
Portugal	10,349,800	100%	10,349,800	80%	8,279,840	0.7
Spain	46,121,700	52%	23,983,284	25%	11,751,809	3.8
Sweden	9,779,430	27%	2,640,446	18%	1,742,694	0.7
Switzerland	8,654,270	80%	6,923,416	0%	0	0.7
The Netherlands	16,924,930	100%	16,924,930	54%	9,139,462	1.9
United Kingdom	65,015,480	100%	65,015,480	69%	44,860,681	3.6
Total	**335,235,210**					31.2
Other countries:						
Greenland	10,543,186	7%	738,023	7.00%	738,023	0.0
Russian Federation	143,456,918	0.8%	1,147,655	0.80%	1,147,655	8
Liechtenstein	37,531	100%	37,531	0.00%	0	0.0
Czech Republic	10,543,186	63%[1]	6,642,207	0.00%	0	1.1
Total			**274,997,810**	43.00%	**112,319,564**	

[1] The QSR (OSPAR, 2000) suggests that 100% of Czech Republic discharges to the North Sea. However, Czech Republic has also catchments of the Danube (discharge to Black Sea) and Oder (discharge to Baltic Sea)
https://en.wikipedia.org/wiki/List_of_rivers_of_the_Czech_Republic#Rivers_flowing_to_the_North_Sea_.28Elbe_basin.29

2.3 Quantification of microplastics emissions

2.3.1 Litter and inadequately managed waste

Despite waste (water) management and application of all existing legislation, littering and accidental leaking of plastics takes place.

In 2015 the global initiative 'International Coastal Cleanup' counted tiny trash items (< 2.5 mm). Approximately 1.5 million plastic pieces and 1.3 million foam pieces were collected over a coastline stretch of 21,500 km [61]. It is acknowledged that marine and coastal litter is not just a beach problem; it is also a watershed problem. Garbage that has been disposed of in rivers, or in dry riverbeds during the summer months, is often washed away by autumn rains and found even on faraway beaches.

Litter travels into the ocean from streets, parking lots, and storm drains. Also smoking related items (cigarette filters, cigar tips, lighters, and tobacco product packaging), medical / personal hygiene items (ranging from tampons and disposable diapers to syringes), and legal and illegal dumping of domestic and industrial garbage can be considered land-based sources. Jambeck et al. [62] estimated that 275 million metric tons (MT) of plastic waste was generated by populations within 50 km distance from the coast in 192 coastal countries world-wide in 2010, with 4.8 to 12.7 million MT entering the ocean, which would be 1.8% to 4.7% of the global production in 2010. For OSPAR countries, the same approach is used in this report. For high income countries, such as the OSPAR countries, the percentage littering is assumed to be 2% [62]. The fraction of inadequately managed waste is negligible ($<10^{-8}$) compared to littering and is therefore not included in this report. The other input parameters are country-dependent variables. To estimate the country-dependent variables, the average of individual values reported by Jambeck et al. [62] for OSPAR countries was used (see Table 2).

Table 2: Summary of input parameters for calculation of plastic litter input to the sea

	Parameter	Value
A	total population	see Table 1
B	average waste production per person per day	2.12 kg/person/day range 1 .33-3.58 kg/person/day
C	percentage coastal or catchment population	see Table 3
D	average percentage of plastic in waste	11.8% (range 2 to 23%)
E	percentage littering	2%
F	fraction litter to sea	15-40%

It appears that waste production figures vary almost by a factor of 3, and the fraction of plastic in waste even varied by a factor of 11 within the OSPAR Maritime Area. It is surprising that such large differences exist in a relatively homogeneous and well-developed region. It was acknowledged that the collected solid waste data should be considered with a degree of caution due to global inconsistencies in definitions, data collection methodologies, and completeness [25]. For example, it was rarely disclosed at what stage the waste generation rates and composition were determined, and whether they were estimated or physically measured. The most accurate method measured the waste generated at the source before any recycling, composting, burning, or open dumping took place. However, the generation rate and composition are commonly calculated using waste quantities arriving at the final disposal site. This method of measurement does not fully represent the waste stream because waste can be diverted prior to final disposal. Given these uncertainties, it was decided not to differentiate between OSPAR countries and to use average solid waste production and composition values for the estimation of plastic emissions to the marine environment in the OSPAR Maritime Area. As a result, the calculated differences in plastic littering in the OSPAR Maritime Area are solely the result of differences in population density.

The fraction of plastic litter that will enter the watershed will be very dependent on local street cleaning practices and infrastructure but also on local geography. In mountainous areas large amounts of plastic litter will enter the watershed by storm water run-off, whereas in flat areas more plastic is retained on soil and in vegetation. The fraction of litter and mismanaged waste that ends up in the water, was measured in 71 municipalities in the San Francisco Bay, where the amounts of trash collected by street sweeping, trash collected in storm water catchments and trash collected in pump stations was monitored and compared with baseline trash loads (obtained from the period before these measures were implemented). The average percentage of uncollected waste, thus the fraction entering the watershed, was on average 61% (range 36-95%). Jambeck et al. [62] used a range of 15-40%, as a “conservative” estimate for the fraction of plastic entering the watershed, which was adopted in the present study.

The emissions of land-based litter is calculated with the following formula:

$$Landbased\ litter\ to\ sea\ = A \times B \times C \times D \times E \times F \times \frac{365}{1000} \qquad [\frac{tonnes}{year}]$$

The total plastic litter input to the sea was estimated between 31 and 199 kilotonnes per year, see Table 3. Consequently, the fraction of the plastic demand that enters the marine environment by land-based littering in the OSPAR Maritime Area is calculated to be 0.1-0.8%. Considering the differences in income, development and infrastructure between the OSPAR Maritime Area and the global scale, these estimates are in good agreement with the estimation of Van Sebille et al. [149] who computed an annual global release of plastic litter of around 1% of global plastics production.

Table 3: Estimated plastic litter input to the marine environment in the OSPAR Maritime Area from land-based sources for the reference year 2015

Country	Land-based littering (kilotonnes/year)	Estimated plastic litter input to the marine environment 2015 (tonnes/year)	
		Min (15%)	Max (40%)
OSPAR countries	607	91	243
OSPAR catchments	498	75	199

OSPAR coastal zone	203	31	81

An alternative approach, based on monitoring data in rivers, was followed by Lebreton et al [72] who estimated that 1.15-2.41 million tonnes of plastic waste enter the oceans from inland sources world-wide. The top 20 polluting rivers, mostly located in Asia, account for 67% of the global total. This implies that 380-795 kilotonnes are emitted by all the other rivers worldwide. The simple approach used for the OSPAR catchments between 75 and 199 kilotonnes per year, seems plausible in comparison with the estimations of Lebreton et al [72].

2.3.2 Pre-production pellets

Preproduction pellets are microplastics that serve as the base material for the conversion to other plastic products. Pellets are defined within ISO 472:2013 as "a small mass of preformed moulding material, having relatively uniform dimensions in a given lot, used as feedstock in moulding and extrusion operations". Pellets can differ in their shape (e.g. cylindrical, lentil or globular shaped). They are in most cases smaller than 5 mm in size. Plastic pellets are considered as primary microplastics. The plastics industry includes polymer producers, plastic converters and plastic recyclers. More details about the sector are provided in Appendix 4.

Plastic recyclates are, like virgin plastics, shaped in the form of flakes or pellets for subsequent conversion into a variety of plastic products. During handling, loading, transport and waste disposal, spills or accidental losses of pellets may occur. When the spill is outdoors, the spilled pellets can be blown or washed into drains or surface water. Pellets can be found and have been monitored on beaches and in biota for many decades. Mishandling of cargo and accidental spills are considered to be the main reasons why high levels of microplastics have been found in some harbour sediments, particularly resin pellets [49].

Various studies used greatly differing pellet losses, ranging from 1% down to 0.0003% (see Table 4). Reasons for these differences could be:

- Most of the data were collected by interviews or questionnaires, and not by measurements;
- The number of companies in the studies is relatively low;
- Different phases of the plastic cycle are involved (transport and production);
- Different companies may be involved (producers and converters);
- Difference in definition of pellet loss: some respondents seem to focus on the total pellet spill, while other respondents focus on the fraction pellets that actually washed to the drains or surface waters. An unknown fraction of the lost pellets will be collected and disposed of with solid waste; and
- Different study designs. For example, the German study estimated resource efficiency (production yield) by comparison of the mass of the feedstock purchased and the mass of the final product sold, whereas in other studies the mass of pellet spills was estimated based on observations.

The German study estimated the highest pellet losses (0.1-1%), but this reflects total plastic available after the polymerisation process and likely includes other losses such as the formation of waste gasses.

The Eunomia study [123] for EU28+N+CH used the emission factor of Sundt et al. [126] and assumed that the amount emitted during transport is further reduced with 0-57% to account for removal by waste water treatment plants, the overall emission factor for the transport phase is thus between 0.022 and 0.05%. This seems reasonable, and maybe a little conservative, because the Danish study [71] estimated a lower emission factor for the transport phase of 0.0013%. Also for the production phase, Eunomia further reduced the emission of the production phase by 10-50% to account for processes that capture or trap the pellets on their way to the ocean. This assumption is not supported by evidence, and is not applied for other sources. Considering all available studies and the uncertainties involved, emission factors for plastic pellets to the ocean of 0.01% to 0.1% of total plastic demand in OSPAR countries was selected.

Table 4: Overview of studies that estimated pellet loss by monitoring, interviews and market data. N is number of companies involved in the study.

	Pellet loss compared to total plastic demand	N	EF used for RA	Reference
Original studies				
UK	0.0003%-0.009%	1	0.001-0.01%	[21]
Denmark	mean total spill: 0.04% max emission to the drains: 0.0013%	8	0.0005-0.01%[a]	[71]
Norway	0.05% (transport)[b] 0.04% (production)	1	0.09%	[126]
Minnesota, USA	0.41%	1	-	[139]
Germany	0.3% - 0.9%	n.r.	0.1-1%	[41]
Studies who used existing information				
Sweden	0.0005-0.01% (transport) 0.04% (production) Overall range: 0.04-0.05% (transport + production)			[86]
EU28+N+CH	0.05% (transport) * (1-removal rate in WWTP[c]) = 0.022 - 0.05% 0.04% (production) * (1- removal rate) = 0.02 - 0.036 Overall range : 0.042 – 0.086% (transport + production)			[123]
OSPAR	0.01-0.1% (transport + production)			This study

[a] a range of 0.0005% to 0.01% was derived by application of uncertainty factors of 0.5 and 10 on the emission factor of 0.0013%.

[b] This value was derived by taking USEPA emission factor of 5 gram/kg handled from transport divided by a factor 10, because it was assumed that spill control measures have been put in place in a large proportion of the industrial sites.

[c] For production losses the same 0—57% range for the capture of particles in waste water treatment as has been assumed for other primary emissions. For transport, 10—50% of microplastic emissions are assumed to be captured in some way before they reach the oceans [123].

We assume that pellet loss is directly related to plastic demand as follows:

$$Pellet\ loss\ [tonnes] = plastic\ demand\ [tonnes] \times Emission\ Factor$$

Emission factors in the range of 0.01% to 0.1% are applied (see Table 4).

The calculated total pellet loss is summarized in Table 5. Details are provided in Appendix 5. For the OSPAR catchment is was assumed that the distribution of companies is similar to the distribution of people; so for certain countries that only partly discharge to OSPAR marine waters a reduction factor was applied as mentioned in Table 1.

Table 5: Estimated preproduction pellet loss in the OSPAR Maritime Area for the reference year 2015

	Plastic demand [million tonnes]	Calculated pellet loss [kilotonnes/year]
OSPAR countries	31	3.1-31
OSPAR catchment	26	2.6-26
OSPAR coastal zone	Not calculated	

2.3.3 Laundry fibres

A recent overview of laundry fibres in the environment is published by Salvador Cesa et al. [119], however emission estimates were not provided.

One of the first studies on fibre release from laundry in relation to environmental exposure was published by Browne et al. [13]. They found that up to more than 1900 fibres per garment, per wash, or 100-300 fibres per litre effluent could be released. Additional assumptions were needed to extrapolate these data to the total emission of synthetic fibres in OSPAR countries. Several other studies used the same data and made assumptions for the missing parameters, for example for the weight of a garment, the size and weight of

fibres, the weight of a wash, the frequency of washing and the removal rate of fibres in sewage treatment plants [71, 126, 123, 10]. They all used similar approaches and assumptions, such as:

- Weight of a wash: 4 kg [105]. More recent data of the same research institute show that the 84% of the washing cycles is done fully loaded. Given a washing machine capacity of 6 to 8 kg , and 84% of the machines are fully loaded an average washing cycle would contain at least 6 kg [1];
- Percentage synthetic garments in developed countries: 50% [124]; and
- Frequency of washing is approximately 165 cycles per household per year, or 55-75 cycles per person per year [105]. Statistic of [1] show that the average washing frequency in the EU is 3.1 cycles per household per week in the reference year 2014. The average household size in the EU is 2.3 persons per household (Eurostat, data 2015).

One key assumption to translate the number of fibres to total mass is the assumption of the weight of one laundry fibre. The above-mentioned studies explicitly or implicitly assumed that a single laundry fibres weight is 0.15 mg. This value was originally derived by Sundt et al. [126] assuming a fibre length of 5 mm and a linear density of 300 dtex (equal to 0.03 mg/mm). The assumption on fibre length is recently confirmed by microscopic measurements [93]. However, the linear density seems to be overestimated. The assumption of a linear density of 300 dtex is not supported by evidence. The linear density of a fibre is amongst others dependent on the thickness of the fibre. The thickness of laundry fibres is approximately 10 µm [93]. Records from the textile industry indicate that synthetic fibres with a thickness of 10 µm have a linear density of approximately 1 dtex [90]. Napper and Thompson determined fibre weights of 1.8 dtex for acrylic fibres and 1.5 dtex for polyester fibres (pers. Comm. Thompson, dd. 16-4-2017). It is therefore very likely that previous assumptions of a linear density of 300 dtex are approximately a factor 200 too high. For the emission estimates, we propose to use a linear density of 1-2 dtex.

The removal rate of microfibres from laundry effluents in the sewage treatment plant is not a fixed value. In reality, the removal rate will be very dependent in the technical specifications of the sewage treatment plant and the volume input of other sources, for instance storm water. Leslie et al. [79] studied influent-concentrations and effluent-concentrations of microplastics of 7 municipal waste water treatment plants in the Netherlands and found a mean retention of microplastics in sewage sludge of 72% (standard deviation 61%). Removal of microlitter from wastewater during different treatment steps of mechanical, chemical and biological treatment (activated sludge) and biologically active filter (BAF) in a large (population equivalent 800,000) advanced Wastewater Treatment Plant (WWTP) was examined by Talvitie et al. [129]. Most of the microlitter (97%) was removed during the pre-treatment, and activated sludge treatment further decreased the microlitter concentration. The overall retention capacity of studied WWTPs was over 99% and was achieved after secondary treatment. These results are in line with a study of Carr et al. [17]. A conservative estimate of the removal rate of 72%, is used to estimate fibre emissions to surface water. Also the number of household that are (not) connected to a sewage treatment plant will affect the input of laundry fibres to surface water.

For emission calculations, the fibre counts of Napper and Thompson [93] were used, because this study documented the weight of the garments from which the fibres were released and the linear densities of the fibres. A maximum of 700,000 fibres were released from 6 kg acrylic laundry. Generic input parameters for the emission calculation are given in Table 6.

Table 6: Generic input parameters for calculation of laundry fibre emissions from households in the present study

Parameter	Value	Source
Number of synthetic fibres per 6 kg wash	496,030-728,789	Napper and Thompson [93]
Fibre lengths	Resp. 7.8 and 5.4-mm	idem
Fibre diameter	Resp. 12-14µm	idem
Linear density (dtex[a])	1-2	Minifibers Inc. [90]
Fraction synthetic clothing	50%	Shui and Plastina [124]
Removal efficiency	72%-99%	Leslie et al. [79] Carr et al. [17]

[a] dtex is expressed as g/10000 m. 1 dtex equals 10^{-4} g/m or mg/mm, which equals 0.1 µg/mm.

The country-specific input parameter such as population numbers, washing frequency and connection of households to sewage treatment plants are listed in Appendix 6. A summary of the estimated emission of laundry fibres is given in Table 7. The total amount of laundry fibres from households to surface water in the OSPAR catchment is estimated to be approximately 1,281 tonnes per year. This is a much lower value than would be expected based on previous assessments (studies mentioned above). As explained before, this is caused by the fact that a lower linear density is used (factor 300 lower).

It is estimated that approximately two-thirds of laundry fibres are retained in sewage sludge. Depending on national policies on the spreading of sewage sludge on land, these emissions could enter the environment, and could be redistributed to surface water through run-off.

Table 7: Estimated laundry fibre emissions [tonnes/year] in the OSPAR Maritime Area for the reference year 2015

	OSPAR countries	**OSPAR catchment**
fibres in laundry effluent	**10,400** (1,600-19,200)	**8,500**(1,300-15,700)
fibres directly to surface water	**1,100** (160-1,900)	**800** (120-1,400)
fibres in STP effluent	**2,600** (410-4,800)	**2,200**(340-4,000)
total laundry fibre emission to water	**3,700** (570-6,800)	**2,900** (460-5,400)
fibres in sewage sludge	**6,700** (1,000-12,400)	**5,600** (880-10,300)

Just before this report was finalized, the results of the EU-funded Mermaids project were published [74]. The authors estimated a total emission of 29,215 tonnes synthetic fibres from domestic laundry in the EU. After correction for population numbers this would be approximately 15,700 tonnes in laundry effluents in the OSPAR catchment area. This is 1.3 times higher than the highest emission predicted in this report based on the study of Napper and Thompson [93]. This is caused by the higher release estimates and a slightly different scenario (load and percentage synthetic clothing). The Mermaids project noticed that the determination of microfibres release had much interference with other elements that contributed to the weight of the filter (detergent residue, water line and others). This implies that the fibre weight may be overestimated. In the study of Napper and Thompson [93] no detergents were used. On the other hand, the use of soap may increase the release of fibres. In order to account for these uncertainties the estimates of the Mermaid baseline scenario have been adopted as upper limit in the uncertainty range in Table 7.

Some laundry is done by professional laundry companies, however there is little data available on this aspect. Sundt et al. [126] used an estimate from Finland which suggests that 10% of garments are washed commercially. The Mermaids study estimated that 3.75 tonnes of synthetic fibres are released from industrial laundries in Spain and Italy. For the OSPAR catchment region this would imply an amount of 9 tonnes synthetic fibres per year (extrapolation based on population numbers). This amount is negligible ($\sim 0.1\%$) compared to the release of fibres from domestic washings.

Based on our estimations the overall emission factor for laundry fibres to surface waters within the OSPAR Maritime Area is approximately 11 (2-20) gram laundry fibres per person per year.

2.3.4 Tyre wear

Tyre wear particles are due to their high polymer content, particle size, low solubility and low degradability considered as microplastics. The particle size of tread wear particles may range from less than 10 up to several hundred micrometres [70]. In the tyre thread, other substances are present such as fillers, anti-oxidants, UV absorbers and residues of substances used or generated during the production process. Some studies correct the emission of microplastics of tyre wear for the rubber content, using percentages of 50% [123] or 60% [126]. A proposal for definition of microplastics [151] includes composite particles that contain a certain amount of additives, and considers the whole particle as microplastics. A cut-off value for a minimum polymer content is not yet decided upon. In this OSPAR background document tyre wear particles are therefore not corrected for the polymer content.

Tyre wear is usually expressed in grams per vehicle kilometres. It is determined experimentally by sales approach or a distance travelled approach [106, 118], or the weight difference between new and end-life-tyres. The total amount of tyre tread material lost per kilometre varies largely, and depends on several parameters such as:

- tyre characteristics with the most significant being size (radius / width / depth), tread depth, construction, pressure and temperature, contact patch area, chemical composition, accumulated mileage and alignment;
- vehicle characteristics such as weight, distribution of load, location of driving wheels, engine power, electronic braking systems, suspension type and state of maintenance;
- road surface characteristics with the most significant being material (asphalt / concrete), texture pattern and wavelength, porosity, condition, wetness and surface dressing; and
- vehicle operation such as speed, linear acceleration, radial acceleration, frequency and extent of braking and cornering.

Because of all those influencing factors, many countries distinguish between several vehicle types for their emission registrations. In the Netherlands additionally three types of road systems are distinguished: urban, rural and highways, because the driving style and speed associated with them affects the emission factor [66, 131]. For the refinement of emissions factors over road types a ratio of emission factors 1:3:5 respectively for highways, regional roads and urban roads was initially assumed [9]. The distance travelled on several road types also affected the derivation of the refined emission factors. A good agreement was found between estimates obtained by the sales approach and by the refined distance travelled approach. Later, a ratio of 1:1:2 for emission factors of different road types was employed [131]. In the National Pollution Transfer Register annual updates of the Dutch emission factors are published. Emission factors have been refined over the years, although the original data and the reasons for adjustment have not been well documented. Additionally different drainage and sewerage systems are connected to different road types and highways may have a different road surface (open asphalt concrete), which captures tyre dust up to 90%. Open asphalt concrete is not suitable for Nordic countries, because this type of road surface is easily damaged by frost. In countries with dry climate, the advantages of open asphalt concrete are not so obvious. An overview of emissions factors applied in different studies is shown in Table 8. By comparison of the average weight of new and used tyres it was estimated that between 10-17% of the rubber is removed by abrasion [39, 71, 106, 126].

Table 8: Overview of tyre wear emission factors (mg per vehicle kilometres)

	The Netherlands Klein et al. [65]			Germany Hillenbrand et al. [59]	Russia [118]	Sweden Gustavsson [55]	UK Pant and Harrison [106]
Road type	**Urban**	**Rural**	**Highway**		Unknown		Unknown
Moped	13	9	10	22.5			100
Motorcycle	60	39	47	45			
Passenger car	132	85	104	90	33	50	
Van	159	102	125		51		
Lorry	850	546	668	700	178		
Busses	415	267	326	700		700	
Truck	658	423	517	1200			
Light duty vehicle	159	102	125				
Heavy duty vehicle	850	546	668				
Others				180			

The environmental distribution is very dependent on the local or national situation.

Based on Dutch data, approximately 30% of the tyre wear is generated on rural roads, 30% on urban roads and approximately 40% on highways, based on registered kilometres driven on these type of roads in combination with the emission factor per vehicle and road type [152]. In urban areas 60% of storm water run-off will end-up in sewerage systems, of which 90% is actually treated. A rationale for this distribution is described in Appendix 7. Since removal rates are quite uncertain, a rate of 10-90% was assumed. In urban areas no direct emissions of tyre wear are assumed to occur. In rural areas and along highways however, 10% of the released tyre wear is assumed to be directly emitted to nearby surface waters. Along with the rubber particles, other tyre rubber components that enter the surface water are zinc, copper and cobalt, PAH's and phthalates. Considering the vehicle kilometres driven in The Netherlands and the distribution pattern over environmental compartments, a total emission of tyre particles to surface water of 1800 tonne/year was estimated here.

For Denmark, the surface area with run-off to combined sewers is 38%, to separate sewers 45% and without sewers 16% [71]. In the area with connections to sewers it was assumed that 30-50% of the tyre wear generated in these areas enters the sewerage system. The remaining part is retained by soil and road borders. They further assumed a removal efficiency of 75-85% in the sewage treatment plant. The direct emission of tyre wear to surface water in areas without sewers is assumed to be 2-5%. Overall, based on kilometres driven and the environmental distribution as described, the total emissions of tyre wear to surface water in Denmark is estimated to be 500-1700 tonne/year.

In Norway, according to the Norwegian Public Roads Administration (Statens Vegvesen), dusts from roads mostly end up in nature. In the inland countryside, road dust would to a large extent be absorbed and trapped in soil along the roads, while in urban environments with impervious surfaces most particles both from building weathering and roads will get washed into sewers and transported towards the sea. A 50:50 division between emissions to soil and to water was suggested for Norwegian road dust [126]. Overall, this would imply a tyre wear emission to surface water of 3750-4750 tonnes/year.

It is difficult to use the same calculation method, with the same distribution pattern for all OSPAR countries, because the necessary statistics were not available. Vehicle kilometres per road type are present, but the environmental distribution might differ, because the vicinity of surface water and the connection to drains and sewage treatment plants will differ between countries. Moreover, the ratio between urban,

rural and highway kilometres may also be different. To calculate a total emission for OSPAR it is therefore assumed that overall the emission per capita is constant within the OSPAR Maritime Area.

Table 9: Overview of emissions of tyre wear particles to surface water in several assessment reports

Country (reference)	Population (million)	Emission (tonnes)	Based on:	Total tyre wear (kg/capita/year)	Emission to water (tonnes/year)	Emission Factor (EF) to surface water (kg/capita/year)
Denmark ([71])	5.7	1915 - 4600	Russian EF - Tyre consumption	0.74-0.81	600-1900[b]	0.10-0.33
Germany ([41])	80.7	60000 - 111000	Industry information UK EF	0.78-1.38	unknown	
Norway ([126])	5.2	7500 - 9500[a]	Russian EF - Tyre consumption	1.44-1.82	3750-4750	0.7-0.9
The Netherlands ([152])	16.9	15452	Dutch EF	0.9	1100-2400	0.06-0.14
Sweden ([98]) ([86])	9.8	10,000 13,000	UK EF Swedish EF	1.0 1.34	Unknown 8,190	0.8 kg/capita/year
EU ([123])	504	465,554	Dutch EF	0.92	50,000-116,000[c]	0.10-0.23

[a] Recalculated values. The study used 60% of this value, to correct for the polymer content. As discussed in paragraph 2.1 we consider the whole tyre wear particle including additives as microplastic.

[b] Recalculated values. Lassen et al mention a range of 500-1700 ton/year, which could not be reproduced based on their descriptions.

[c] Recalculated values. The study used 50% of this value, to correct for the polymer content. We consider the whole tyre wear particle including additives as microplastic. A removal efficiency of 0-57% was assumed [123].

Given the high uncertainties in environmental distribution and geographical and infrastructural variations between countries, an overall average emission of tyre wear particles to water in a range of 0.1 to 0.9 kg/per capita/year (see Table 8) was used, then multiplied with the population densities mentioned in Table 1.

Table 10: Overview of estimated microplastic emissions [x1000 tonnes/year] of tyre wear particles. Reference year 2015.

	Average estimate (kilotonnes)	**Range (kilotonnes)**
OSPAR Countries	167	34-302
OSPAR catchment	137	28-247
OSPAR coastal zone	56	11-101

2.3.5 Rubber infill on synthetic turf sports pitches

Synthetic turf is typically used in regions where climatic conditions do not allow the efficient usage and maintenance of natural grass fields. A very rainy or extremely dry climate may provide reasons for installing synthetic turf. However, other strategic and economic reasons may also play a role. In order to keep the synthetic fibres in an upright position and provide the desired elasticity of the field, granulates are used as infill. The most common infill material is styrene butadiene rubber (SBR), which is retrieved from end-of-life vehicle tyres. This practice is typical for synthetic football and synthetic rugby pitches. The rubber infill has

a mean particle size of approximately 2 mm (0.7-3 mm). SBR infill is considered as microplastics because of its size and substantial polymer content. Besides that SBR contains many potentially toxic components, such as metals, nitrosamines and PAH's, which gave rise to concerns and debates about the safety of SBR infill. For The Netherlands, the association of the tyre industry (VACO/RecyBEM) estimated that 95% of the artificial turf football pitches was filled with SBR (pers. comm. 2016). In Norway, the market share of SBR was 90%, TPE and EPDM are mostly used for the remaining 10%. Alternatives for SBR can be used, including coated rubber, EPDM, and TPE (thermoplastic elastomer), which are all considered as microplastics. Microplastics free, organic infills (for instance based on cork or coconut shells) and sand are also available and are becoming increasingly popular for instance in the Netherlands. They all have their specific advantages and disadvantages. New synthetic fibres, which do not require infill, are also available, but because they are more expensive, the application is still limited. In Figure 7, the composition of a synthetic football pitch is shown.

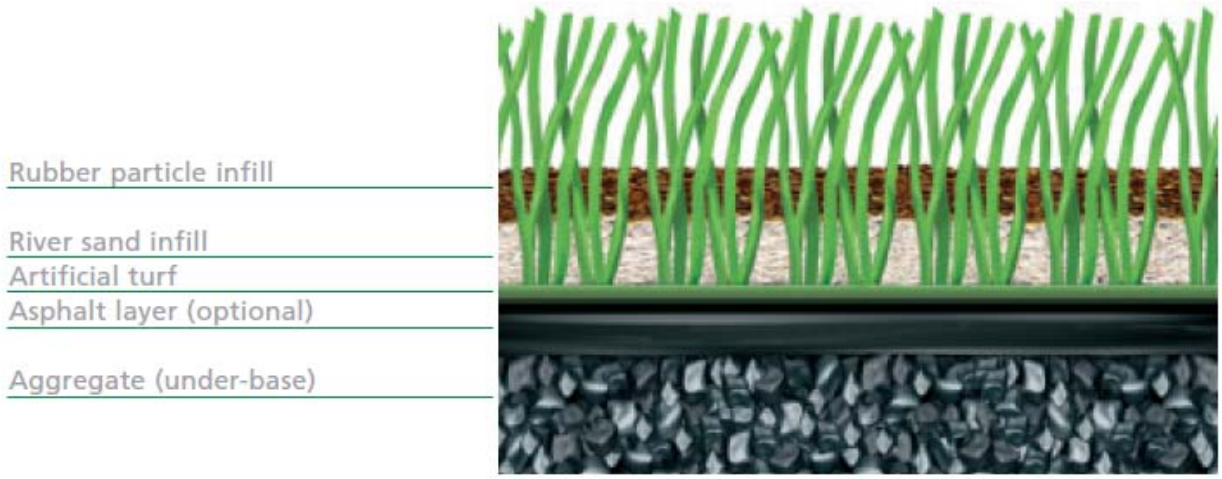


Figure 7: Composition of a synthetic football pitch (picture from FIFA)

Beneath the artificial turf, an under-base of rubber is applied, to provide the necessary elasticity and bouncing conditions for a sports field. As the rubber under-base is covered by artificial turf, loss to the surrounding environment does not occur. It is during the waste phase, or during reconstruction or replacement of the synthetic turf that loss to the environmental can occur. Replacement of a synthetic sports field is expected between a lifetime of 10 and 20 years.

However part of the rubber infill is lost to the environment directly, or attached to clothes and shoes and taken home where it ends up in the washing machine. The following release pathways are considered for infill granulate and abrasions from artificial grass fibres [71]:

- Release to surrounding soil area;
- Release to paved areas surrounding the field, and subsequently release to sewerage system via grates (includes releases from shoes and clothing);
- Release of infill particles to the indoor environment, as the particles get stuck in sports-bags, shoes and clothing where they 1) are removed by vacuum cleaning or 2) are released to sewerage system via discharges from washing machines; and

- Release to drainage via drainage water. The fate of the drainage water is: 1) downward seepage; 2) release to sewerage system or 3) release to nearby streams due to heavy rainfall.

The Danish EPA has estimated the release of microplastic particles from artificial sports fields in Denmark [71]. Around 100 tonnes of rubber infill granulates are used for a regular football field. Parts of the infill granulates will disappear from the field to the surrounding area and must therefore be continuously replaced, while replacement sometimes is necessary due to compression of the infill granulates on the field. In the Danish report it is estimated that the consumption of infill granulates is 3-5 ton per year for a standard football field. It is assumed by Lassen et al. [71] that the release to the environment is equal to 50% of the consumption of infill granulate. Direct distribution to the surface water is considered negligible, whereas 5-20% is assumed to end up in the sewage system. Because the granules are relatively large (>0.3 mm) it is assumed that most of them end up in sewage sludge, and only 3-6% of the fraction that enters the sewerage would be released to surface water. Overall, the percentage of the supplemented infill that may end up in surface water is 0.1-0.6%, which for Denmark is estimated to be 1-20 ton/year. In Denmark, sludge may go to agricultural soil. This is not the case in some other OSPAR countries.

In The Netherlands a lower supplement rate of 0.5-1 ton/year was assumed by VACO the Dutch car tyre association, based on interviews with field managers, maintenance companies and sports associations. Moreover, in the Netherlands, it is recognized that release to the environment is only partly responsible for the annual supplemental infill needs. Two major additional processed contribute to the "loss" of infill: compaction and cleaning. Compaction of the infill layer occurs in certain intensely played areas. The infill is not distributed to the surrounding environment, but becomes more compact. As a consequence the infill layer becomes thinner and needs to be supplemented. It is estimated that this accounts for 10% of the annual infill. Maintenance guidelines prescribe that fields are cleaned 6 times per year to remove fallen leaves and dirt. This is done by a specialized vacuum cleaner, which also removes the sand and rubber infill. The infill is sieved and returned to the field. Though a fraction of the infill is discarded as waste, along with the other dirt. Approximately 80% of the annual infill supplement is needed to compensate this loss. Distribution to the surrounding pavement is assumed to be not more than 5%, and 1% is assumed to end up in surface water through storm water sewage systems and run-off. It is assumed that 5% is transported to the indoor environment (pers. comm. VACO 15-03-2016). A follow-up study by SWECO in the Netherlands estimated that 20-50% of the annual infill is distributed to the environment, accounting for 50 to 460 kg infill per field per year [156]. How much ends up in surface water is very dependent on the local situation. The highest amount distributed to surface water in the five fields included in this study was 70 kg, in the other fields emissions to surface water were very low.

A Swedish study assumed an annual infill supplement of 2-3 tonnes [86]. There are no monitoring data to verify the assumptions, made by the Netherlands, Sweden or by Denmark. The Norwegian study of Sundt et al. [126] assumes that the emissions of rubber infill to surface water are negligible. Distribution percentages to surface water may be very dependent on local geographical and meteorological conditions. Therefore, an uncertainty range is taken into account of 500 to 5,000 kg infill supplement per field per year, and 0.1% to 1% release to surface water. With these data, emissions of rubber infill to surface waters on land in the OSPAR area are estimated as follows:

$$Release\ granulates\ to\ surface\ water = Number\ of\ synthetic\ turf\ pitches \times supplement\ rate \times \%to\ surface\ water$$

Furthermore, microplastics will be released from the artificial grass fibres due to wear and tear. It is estimated that 5-10% of the grass fibres are abraded and released per year [71]. This seems somewhat

overestimated, since the durability of an artificial turf field is at least 10 years. We assume that 40-60% of the fibres is released during a period of 10 years. Similar to the distribution of infill, and taking into account the uncertainties in the distribution pattern, it is assumed that 0.1 to 1% of the released fibres end up in surface water. According to a report from the Norwegian Institute for Water Research (NIVA) (Källquist 2005), the amount of grass fibres is equal to 0.8 kg/m². A standard football field is 7,140 m²; therefore, the amount of fibres is 5,712 kg/field.

$$Release\ granulates\ to\ surface\ water = Number\ of\ synthetic\ turf\ pitches\ \times\ fiber\ release \times \%to\ surface\ water$$

Table 11: Input for calculation of emissions of rubber infill to surface water

	Lowest	Highest
infill supplement rate (ton per field per year)	0.5	5
fibres release (% of total fibres)	4%	6%
% to surface water	0.1%	1%

The European Synthetic Turf Organisation has published statistics of synthetic turf fields in several countries (www.theesto.com). A compilation is shown in Table 12. The median number of synthetic turf pitches per club is 0.12. For Finland, Portugal and Luxembourg no statistics on synthetic turf pitches could be found; in these cases the number of synthetic turf pitches is assumed to be equal to the $number\ of\ clubs\ \times 0.12$.

Note that the number of pitches has rapidly grown over the past years, and is expected to grow even more in the coming years. Since some of the data are from 2009 and 2012, the number of synthetic turf fields mentioned in Table 12 is an underestimation of the actual current number of fields. Moreover, the statistic focussed on football pitches, and other synthetic turf applications have been ignored. However, artificial football pitches are considered the most relevant source, because it concerns the outdoor and open application of unbounded granules, which can be easily spread to the surroundings. Other applications in sports often concern bound granulates, such as in athletic tracks, or under-base applications that are hardly spread to the surroundings.

Table 12: Indicative number of synthetic football pitches in OSPAR countries

	no. synthetic pitches (size 7140m^2/field)	Ref. year	ref.
Belgium	280	2012	[a]
Denmark	311	2016	[c]
Finland	151		[e]
France	2157	2012	[a]
Germany	2500	2009	[b]
Iceland	28	2012	[a]
Ireland	48	2012	[a]
Luxembourg	5		[e]
Norway	897	2012	[a]
Portugal	76		
Spain	2000	2009	[a]
Sweden	789[f]	2015	[86]

Switzerland	113	2012	[a]
Netherlands	2072	2014	[d]
UK	896	2009	[b]
Total	**12638**		

a. Data 2012 (www.theesto.com).
b. Data 2009 (www.stadia-magazine.com/news.php?NewsID=15608)
c. Data 2016 (accessed 1 March) www.dbu.dk/klubservice/Kampe_og_baner/kunstgraes_fodboldbaner/Find_en_kunstgraesbane
d. http://www.telegraaf.nl/telesport/voetbal/21994325/__Veel_amateurs_op_kunstgras__.html
e. Extrapolated value
f. Sweden has in total about 1,336 artificial football fields (in 2016) of which 697 are for teams of 11 players, 235 smaller fields for 5, 7 or 9 players, 81 indoor arenas and 323 other sport fields [86]. The total area of outdoor fields was around 5,845,980 m^2 in 2016 [86]. This is equivalent to 789 standard football fields of 7140 m^2.

Table 13: Estimated emissions of microplastics from artificial turf soccer fields

	Estimated number of artificial pitches	**Estimated emission to water (tonnes/year)**		
		Rubber granulate	Fibres	Sum
OSPAR Countries	11230	6-618	3-42	9-660
OSPAR catchment	8921	5-465	2-32	7-497
OSPAR coastal zone	3725	2-199	1-14	3-213

2.3.6 Paint emissions from shipyards, marinas, buildings and roads

Paint particles are considered as microplastics because they have a backbone of polymers, which become solid after application. In some cases solid microplastic powders are added to paints as a colour enhancer or to obtain a matting, glitter or structure effect, or to decrease the density of the paints [71]. Paints are also used to increase the durability of surfaces. Paints form a water repellent layer, provide resistance against UV, and in the shipping sector paints serve as antifouling agent.

The Organisation for Economic Co-operation and Development (OECD) has generated estimates for how much of this paint is discharged to water during the life of the coat of paint: 1.8% during painting, 1% due to weathering and 3.2% during removal, totalling 6% [100]. Two national assessments on paints emissions have been published: one for Norway and one for the Netherlands, both took the OECD approach as a starting point. The Norwegian study used the OECD emission factors estimates, but also adds to this the OECD estimate of emissions to soil of 5%, thereby assuming that all emissions to soil end up in the marine environment, which may well be an over estimate [126]. An EU-wide assessment was also based on the OECD emission factors, and like the Norwegian study, it applies a reduction by a factor 4 to account for a polymer content of 25% [123]. The Dutch study used the differentiated solid content of different paint types for microplastics in the range of 20-80% [152]. Moreover the EU study, like the Dutch study, takes into account that part of the emissions will enter the sewerage system and will be partly retained in the sewage sludge.

Table 14: Overview of studies that estimated paint emissions

Country (reference)	Marine paints	Building	Road paint	Total (tonnes)	Overall Emission Factor (EF) (kg/capita/year)
Norway ([126])	**770** $EF_{commercial}$=22% $EF_{recreational}$=100% Microplastics= 25%	**148** $EF_{application}$ = 5% EF_{wear} = 3% (wear) Microplastics= 25%	**320** EF_{wear} =100% Microplastics= 25%	1,270	0.18
The Netherlands ([152])	**200** $EF_{commercial}$=1% $EF_{recreationl}$=5% Microplastics =60%	**487** Application: Professional= 3.2% DIY = 6.4% Wear Professional= 3% DIY wood stains = 15% Microplastics= dependent on paint type 20-80% To WWTP = 72% Removal rate = 50%	Not assessed. Car paint lacquers were responsible for 4% of the paint sales.	687	0.04
EU ([123])	**825-4056** $EF_{commercial}$=6% $EF_{recreationl}$=6-100% Microplastics= 25%	**12,300-28,600** EF=3.3% 60% to WWTP 0-57% removal Microplastics= 25%	**700-18,000** EF=3.2% 60% to WWTP 0-57% removal Microplastics= 25%	13,825 - 50,656	0.03-0.10

In The Netherlands lower emissions factors were used than in the Norwegian and EU assessments for the shipping sector because it was argued that mitigation measures are in place in the building and shipping sectors that reduce the paint emissions [152]. In The Netherlands, all of the shipyards collect and purify waste water and recreational vessels are mostly maintained at marinas. Paint sales statistics in The Netherlands showed that paint consumption for recreational ships were only 3% of total paint sails in the shipping sector, whereas the Norwegian and EU assessment assumed that this percentage is 25%. These market shares may be different between countries and have an effect on the total emission estimates.

Considering the uncertainties in the environmental distribution and national differences in paint practices and dust mitigation methods we assume a range of 0.03-0.18 kg/per capita/year. As differentiated national sales figures were not available, a preliminary extrapolation is done based on average emission factors bases on population density. The estimated overall emission factors and total emissions are shown in Table 15.

Table 15: Estimated paint emissions to surface water in the OSPAR Maritime Area for the reference year 2015

	Building	**Shipping[1]**	**Road paint**	
Emission factor (gram/capita/year)	24-56	2-148	1-61	
				Sum
OSPAR Countries	8 -19	3-50	0.5-30	50 (11-89)
OSPAR catchment	7-15	2-41	0.4-17	41 (9-73)
OSPAR coastal zone	3-6	2-41	0.2-7	29 (5-54)

[1] Emissions of the shipping sector in coastal zones are assumed to be identical to emissions in the catchments, because shipyards and marinas are predominantly found in the coastal regions. A reduction based on population density is therefore not justified.

2.3.7 Scrubs and exfoliating cosmetics

Discussions to determine an appropriate definition for microplastics are taking place, especially with regard to cosmetics and personal care products. This includes the questions whether waxes should be considered as solid particles and whether plastic particles in the nano-range should be also considered. This report focuses on so-called microbeads, which are mostly particles with an abrasive or decorative (glitter) function in shampoo, facial scrubs, soaps and bath products. Their size is generally >50 micrometres. Several estimates of microplastics releases (concerning plastic microbeads) from cosmetic and personal care products are available. In 2011 Gouin et al. [53] estimated a daily use of 2.4 mg microbeads per person per day, based on sales of 195 million litres of liquid soaps and shower gels in the United States in 2009. The sales data indicated that 15% of the market is shared by companies that use micro-plastic beads in their liquid soaps. It is further assumed that these companies use microplastics in 10% of their brands. This implies a market penetration of microbeads containing products of 1.5%. Based on the U.S. patent describing the use of microplastics in liquid soaps, a typical formulation may include a maximum of 10% of polyethylene. The density of polyethylene is 0.9 g·cm-3.

In 2015 Gouin et al. [52] performed a new survey on the use and release of microbeads from scrubs and exfoliating products. The difference with the previous one is that here European sales data of 2012 were used. A total amount of 4,360 tonnes of microbeads were used across all European Union countries, plus Norway and Switzerland in 2012. Polyethylene microbeads represented 9% of the total amount. Polyethylene micro beads in the size range of 450-800 µm was the dominating material reported in the survey. It was estimated that micro-plastic beads originating from cosmetic products represent between 0.1%–1.5% of the total amount of plastic litter emitted to the North Sea marine environment.

Napper et al. [92] measured microplastics in six different facial scrubs and estimated that between 4,594 and 94,500 particles could be released into the environment per use. They also estimated that the UK population is emitting 40.5–215 mg of PE per person per day into surface water, resulting in a total of 16–86 tonnes per year. Assumptions were that 1.1 million women (=3% of all females in the UK) use 5 g of facial scrub daily, that the market penetration of microbeads containing facial scrubs is 75%, and that 25% of the microbeads is retained in a sewage treatment plant.

Table 16: Comparison of estimated daily emission of microplastics by the use of facial scrubs. Measured data are printed in bold, the other data are based on assumptions.

Reference	Fraction of microplastic in product	Market penetration	Usage	Removal by STP	Emission (mg microbeads per capita per day)
Gouin et al. [53]	10%	15%	Sales US, 2009	Not included	2.4
Gouin et al. [52]	10%	6%	Sales EU27+CH+N, 2012	Not included	17.5 (EU27+N+CH) Range: 12-51 mg/capita/day
Napper, Bakir [92]	2-8%	75%	Use 5 g scrub per day by 3% of the women	25%	40.5-215

The market share of plastic microbeads containing personal care and cosmetic products in the above mentioned studies varies from 6 to 75%, all based on assumptions that were not supported by evidence. The market share is an influential assumption that determines the estimated emission factor. Because we found no evidence for better specification of the market share we used the range of country specific emission factors reported by Gouin et al. [53] and the maximum emission factor of Napper et al. [92]. The resulting range of microbead emissions is given in Table 17.

Table 17: **Estimated emissions of microbeads from personal care and cosmetic product in tonnes/year**

	minimum	**maximum**	**average**
Emission factor [mg/capita/day]	17.5-51	215	1-61
		Emission (tonnes/year):	
OSPAR Countries	3,225	65,531	34,378
OSPAR catchment	2,647	21,580	12,114
OSPAR coastal zone	1,143	8,814	4,407

2.3.8 Abrasive cleaning agents

Detergents are divided in five different groups: laundry detergents, dishwasher detergents, bathroom cleaners, bleaching and surface cleaners. Abrasives are mainly used to clean hard surfaces. Abrasive plastic particles are covered by the term microbeads.

Microplastics in detergents are only expected for the cleaning of hard surfaces, because the microbeads have an abrasive function. Additionally, microplastics are only used when other, cheaper alternatives such as calcium carbonate, silica or aluminium are not suitable, because these may cause scratches on delicate surfaces. A survey of The Netherlands showed that detergents are a very small source of microplastics [152]. More than 400 (abrasive) cleaning agents of six market leading companies were screened resulting in ten products that were suspected to contain microplastics. It concerned products for cleaning of floors. Further verification by contacting the companies involved, and eventually followed by chemical and microscopic analysis should confirm the precise nature and function of these ingredients. It was estimated that the total emission of microplastics from cleaning agents is 2.4-2.6 ton/year, which is almost completely discharged to the sewer. Essel et al. [41] estimated an amount of <100 tons for Germany. For the emission estimates a

certain fraction of households that are not connected to a sewage treatment plant is taken in to account, as was done for laundry fibres (see Appendix 6).

Table 18: **Estimated emissions of microplastics from abrasive cleaning agents**

	minimum	maximum	average
Emission factor [mg/capita/day]	0.4	3.4	1.9
		Emission (tonnes/year):	
OSPAR Countries	28	229	129
OSPAR catchment	23	186	104
OSPAR coastal zone	9	76	43

The emission of microplastics from laundry detergents are not included in the figures in Table 18. A large European research project in to laundry detergents has been conducted by Italian and Spanish scientists in the Mermaids project, which showed that persistent and non-soluble polymers (for instance polypropylene terephthalate) are present in certain laundry detergents. Amounts or concentrations of microplastics in laundry detergents were not reported [89].

2.3.9 Overview of quantified sources of microplastics

An overview of estimated emissions of microplastics in the OSPAR catchment areas from different sources is shown in Table 19. The overview shows large differences between emissions of different sources. The lowest estimated emission is for abrasive cleaning agents (detergents), which is a factor 1000 lower than the emissions of the highest source: tyre wear. There are some recent studies which show that microplastics (for example fibres, tyre dust) are ubiquitous in urban air [30, 31]. Reports that estimate the contribution of atmospheric deposition to marine microplastics is currently not available. The estimations of the source emissions go with considerably uncertainties due to knowledge gaps (assumption have to be made) and due to inevitable variations in for instance geography and consumer behaviour. The uncertainties and variations in the input parameters of the source emission calculations are addressed as a range. As a result the uncertainties (see ratio max/min in Table 19) of source emissions is mostly lower than a factor of 10. The highest uncertainty is present for microplastics emissions by artificial turf and infill, and the lowest and highest estimates vary by more than a factor of 70. This is caused by a lack of knowledge about the amount of microplastics that actually reach the watershed and about the wear and tear of fibres and by variations in the amount of annual infill supplements.

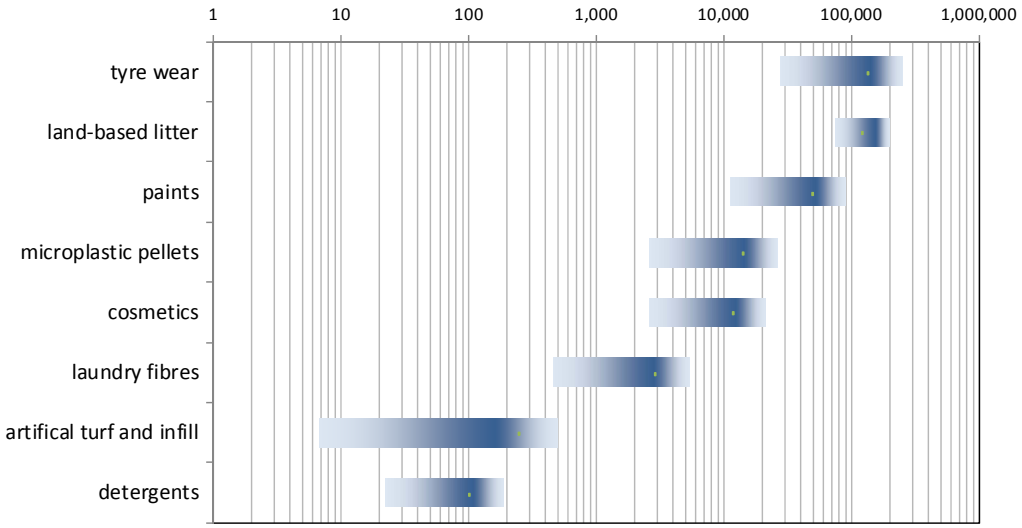


Figure 8: Estimated emissions of microplastics in OSPAR catchments (tonnes/year). The bars represent the uncertainty margins of the emission, white dots represent the midpoint.

Table 19: Overview of estimated emissions of microplastics within the OSPAR catchment areas [tonnes/year] for the reference year 2015. Values smaller than 1000 are rounded to the nearest decade, and values larger than 1000 are rounded to the nearest hundred.

type	average	min	max	Ratio max/min
tyre wear	137,000	27,500	247,000	9
land-based litter	122,000	74,700	199,000	3
paints	50,100	11,200	89,000	8
microplastic pellets	14,400	2,600	26,100	10
cosmetics	12,100	2,600	21,600	8
laundry fibres	2,900	460	5,400	12
artificial turf and infill	250	10	500	73
detergents	100	20	190	8

2.4 Pathways of microplastics to the marine environment

The spatial distribution and accumulation of litter, and probably also of microplastics in the ocean is influenced by hydrography, geomorphological factors, prevailing winds and anthropogenic activities. Hotspots of litter accumulation include shores close to populated areas, particularly beaches, but also submarine canyons, where litter originating from land accumulates in large quantities [108].

Microplastics can reach the marine waters directly from marine activities or recreational activities at sea, on shores / beaches, or indirectly through riverine inputs or discharges of sewage treatment plants. An unknown amount of plastic may reach the marine environment though the air: for example microplastic "dust" from traffic, or balloons, which are frequently found objects in beach litter monitoring programs, and that are subject to further fragmentation in to micropartcles. Figure 9 shows the main potential pathways for microplastic particles entering the environment. Measures intended to reduce microplastic emissions can intervene at particular points in the contamination route or address the source directly. The pathways are described in greater detail below.

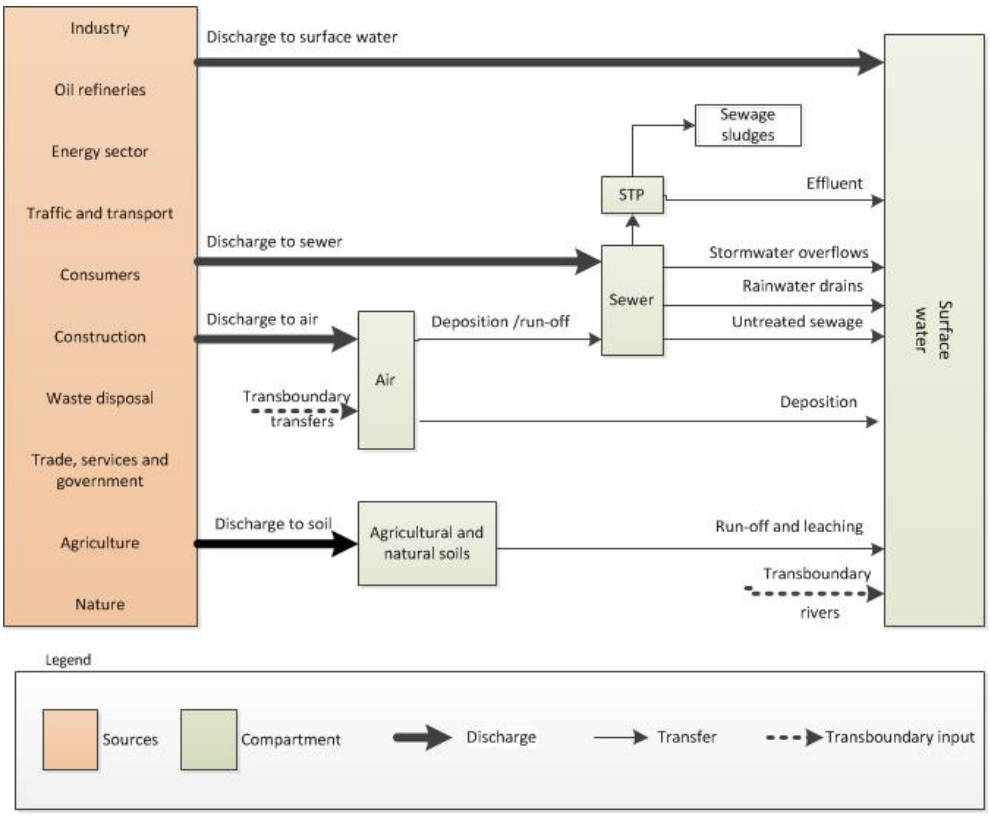


Figure 9: Sources and routes of plastics/microplastics to surface water (www.emissieregistratie.nl).

In some countries a licence is required in order to discharge waste water (whether treated or not). This licence sets standards for the quality of the discharged water. These quality requirements govern, amongst others, heavy metals, some organic micro pollutants and nutrients, but not microplastics. In principle, household waste water is treated at sewage treatment plants (STP). In most cases, industries have their own treatment installations or filters at their disposal. Microplastics are not recycled and due to their limited size, it is difficult for sewage treatment plants to filter all microplastics out of the water. Only limited data is available on the treatment efficiency of sewage treatment plants regarding microplastics. In a study conducted by the Institute of Environmental Studies (IVM) at VU University Amsterdam, in collaboration with Deltares, Delft University of Technology and the Hollandse Delta Water Board, research was conducted into the presence of microplastics in various flows at the Heenvliet sewage treatment plant [77]. In this exploratory study that only included a few samples, 90% of the microplastics were removed by the treatment process. The remaining 10% enters the surface water, from where it can reach the sea. In a follow-up study, the number of sewage treatment plants was increased to three and a larger number of samples were taken. Microplastics were detected in the effluent (on average 39-89 microplastic particles per litre). This confirms that microplastics are not entyrely removed from water by sewage treatment plants [11]. Microplastics were also detected in the influent. It turned out that the concentration of microplastics varied greatly with time, and that effluent concentrations were not always lower than influent concentrations. The previous estimate for treatment efficiency (90%) was not confirmed by the follow-up study. A recent study conducted by four Dutch sewage-treatment plants show average microplastic concentrations of 48 to 55 particles per litre [78].

In Belgium an STP near Gent was examined. Influent samples had an average microplastic content of 17±7 particles/L, which was reduced to 5±1 particle/L in the effluent. This corresponds to a removal efficiency of 80%. Extrapolating the results of this STP, Van Cauwenberghe et al estimated a daily discharge of 1400 microplastics particles/inhabitant/day [146].

In Germany, treated waste water, sewage sludge and separated light solids (oil, grease) were sampled from 12 sewage plants of the OOWV in Lower Saxony [91]. The contained microplastics (> 10 µm) were determined using micro- FTIR (Fourier Transform Infrared Spectroscopy) and ATR- (Attenuated Total Reflectance) FTIR technique.
For sampling the treated waste water, all material larger than 10 µm was retained on a stainless steel cartridge filter. Depending on the amount of suspended particles 390 – 1000 litres were filtered. From the sewage sludge and the light solids one kilogram was taken.

To enable the measurement with micro- FTIR all samples had to undergo a complex sample preparation. While filtering the waste water a variety of natural materials like algae, sand, lipids, insect and plant remains were also kept on the filter. These were removed by applying a plastic-preserving, enzymatic-oxidative degradation with a following density separation using zinc chloride. The remaining material was analyzed using ATR- FTIR, micro- FTIR and "chemical imaging". Microplastic particles and fibres were detected in all waste water samples. Predominantly the particles were in a size range of 50 – 100 µm. The quantity of microplastics varied considerably: Between 86 m-3 (Neuharlingersiel) and 714 m-3 (Essen) particles and 98 m-3 (Burhave) to 1479 m-3 (Schillig) fibres were enumerated. A final installed filtration system in the sewage plant in Oldenburg reduced the amount of microplastics from 1131 m-3 to 29 m-3 (97 %). Compared to the other eleven plants, the sewage plant in Holdorf showed a much higher amount of microplastics (8851 particles m-3 und 4808 fibres m-3). Depending on the sewage flow the sewage plants release 93 million to 8.2 billion microplastic particles and fibres in the rivers per year.

To examine the potential to retain microplastics, a subsample of the sludge was degraded by using 10 molar NaOH. Polyethylene (PE), polypropylene (PP), polystyrene (PS) and polyamide (PA) particles were detected and resulted in 1,041 to 24,129 particles per kilogram dried sludge and 1.2 – 5.7 billion microplastic particles per year and plant. Since the subsamples were relatively small, the amounts have to be seen as indicative values. The samples of the light solids could not be purified enough to enable the analysis with micro- FTIR. However, the light densities of the most common polymers (PE, PP) and detected macroplastics (cotton buds, contact lenses) indicate a potential removal of microplastics too.

Beside this study, only two other studies published data of microplastic concentrations in treated waste water. Leslie et al. (2013) detected microplastic particles in a range from 9 to 91 particles per litre in the treated waste water of three Dutch plants [78]. One plant had an additional installed Membrane bioreactor, but no influence on the amount of microplastics was proved. HELCOM examined the impact of the sewage plant in St. Petersburg on the microplastics concentration while purifying the waste water [58]. Compared to the incoming amounts, the released portion was reduced by 96% to 16 L-1 fibres, 7 L-1 synthetic and 125 L-1 black particles. Both results show much higher concentrations in the treated waste water than the mean of < 1 L-1 microplastics that was detected in this study.

Due to the early stage of this topic the results gained in this study are only indicative. More studies are necessary that examine further sewage plants and the microplastics concentrations that are transported in rivers. Only then the importance of punctual and diffuse sources can be interpreted correctly and appropriate sanctions developed that will help to reduce disposal of microplastics.

Sewage treatment plants have insufficient capacity to cope with periods of heavy rainfall, meaning that untreated waste water will enter the surface water via a sewer overflow. This particularly occurs at plants that do not have a separated waste water system, i.e. rainwater and domestic waste water are combined and treated at the plant. In the case of separated waste-water collection, rainwater and domestic waste water are treated separately. The advantage of this system is that overflows are less frequent, due to the

fact that the amount of domestic waste water is relatively consistent. The disadvantage is that urban runoff water is not treated, and hence plastics / microplastics from litter can directly enter the surface water. The proportion of households that are not connected to the sewer network and as a result directly discharge untreated waste water amounts to 0.3%: roughly 23,000 households [114]. Direct untreated discharge of domestic waste water can also originate from the shipping industry. The Emission Register calculated that in 2012, 12% of the waste water from recreational shipping was collected. For chartered ships and passenger ships, this figure was 5%, while 0% of waste water from inland shipping is collected. However, there was an increase in the number of vessels that collect their waste water in tanks and deposit it at the port [115].

2.5 Fate and distribution of microplastics in the aquatic environment

Once in the aquatic system, microplastics can distribute in five main oceanic compartments: 1) in sediments and shorelines, 2) in water columns, 3) in surface waters 4) on the sea floor and 5) in biota. The movement of microplastics in and between different compartments is complex and driven by different chemical, physical and biological processes. The distribution and aggregation behaviour is also influenced by different plastics properties including polymer type, size and shape (GESAMP 2016).

The fluxes of microplastics in and between different compartments are displayed in Figure 10.

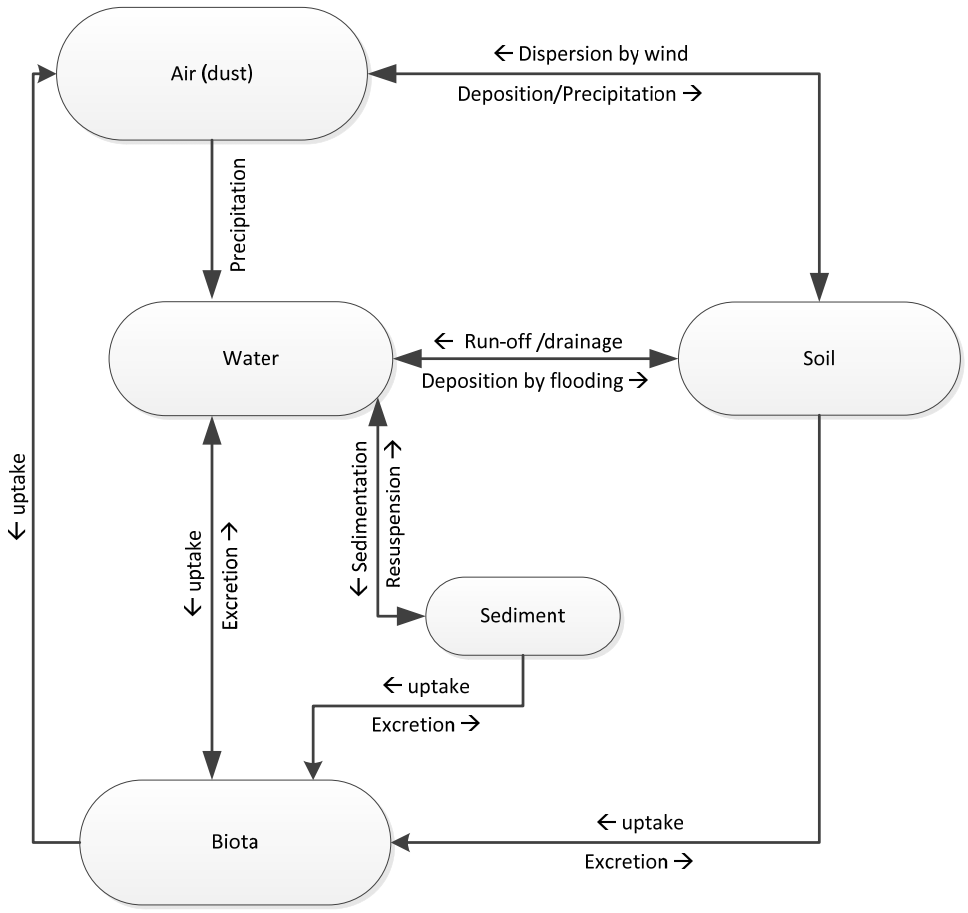


Figure 10: Potential distribution pathways of microplastics

The distribution and behaviour of microplastics is dependent on the specific gravity. Plastics types that are heavier than water (> kg/L) will sink and end up in the sediment on a relative short distance from the effluent point, whereas lighter plastics will float downstream and are distributed over large distances (Table 20). However, the properties of plastic may gradually change as a result of weathering processes, which will in turn again influences their distributive behaviour.

Table 20: Overview of densities of different plastics

		Specific gravity (kg/L)
Most likely to float on water:		
Polyethylene (PE)	Plastic bags, bottles, six-pack rings, gear, cages and pipes for fish farming	0.91-0.94
Polypropylene (PP)	Rope, bottle caps, gear, strapping	0.90-0.92
Styrene Butadene Rubber SBR	Roofing felt and car tyre	0.94
Polystyrene (expanded) (EPS)	Bait boxes, floats, cups , expanded packaging	0.01-1.05
Most likely to end-up in sediment:		
Polystyrene (PS)	Utensils, containers, packaging	1.04-1.09
Acrylic	Paints, packaging	1.09-1.20,
Polyvinyl chloride (PVC)	Film, pipe, containers , buoys	1.16-1.30
Polyamide or nylon (PA)	Gear, fish farming nets, rope	1.13-1.15
Polyurethane (PUR)	Insulation	1.2
Poly(lactic acid) (PLA)	Packaging, cups, mulch film	1.21-1.43
Cellulose acetate	Cigarette filters	1.22-1.24
Polyethylene terephthalate (PET)	Bottles, strapping, gear	1.34-1.39
Polyester resin + glass fibres	Textiles, leisure boats	>1.35
Polytetrafluorethylene PTFE (aka Teflon)	Personal care products	2.2

Once microplastics are released in to the environment, they can be colonised by algae; a process which is called fouling. Larger pieces of plastic can be colonised by larger organisms, such as mussels and worms. By colonisation the weight of the particles can increase, and a floating particle can precipitate.

3 Concentrations and impacts

3.1 Concentrations of microplastic in marine samples

Microplastics abundance depends on population density and human activities such as recreation, fishing and maritime transport (see Chapter 2). With respect to these factors, the five OSPAR regions are described as follows (*http://www.ospar.org/convention/the-north-east-atlantic*).

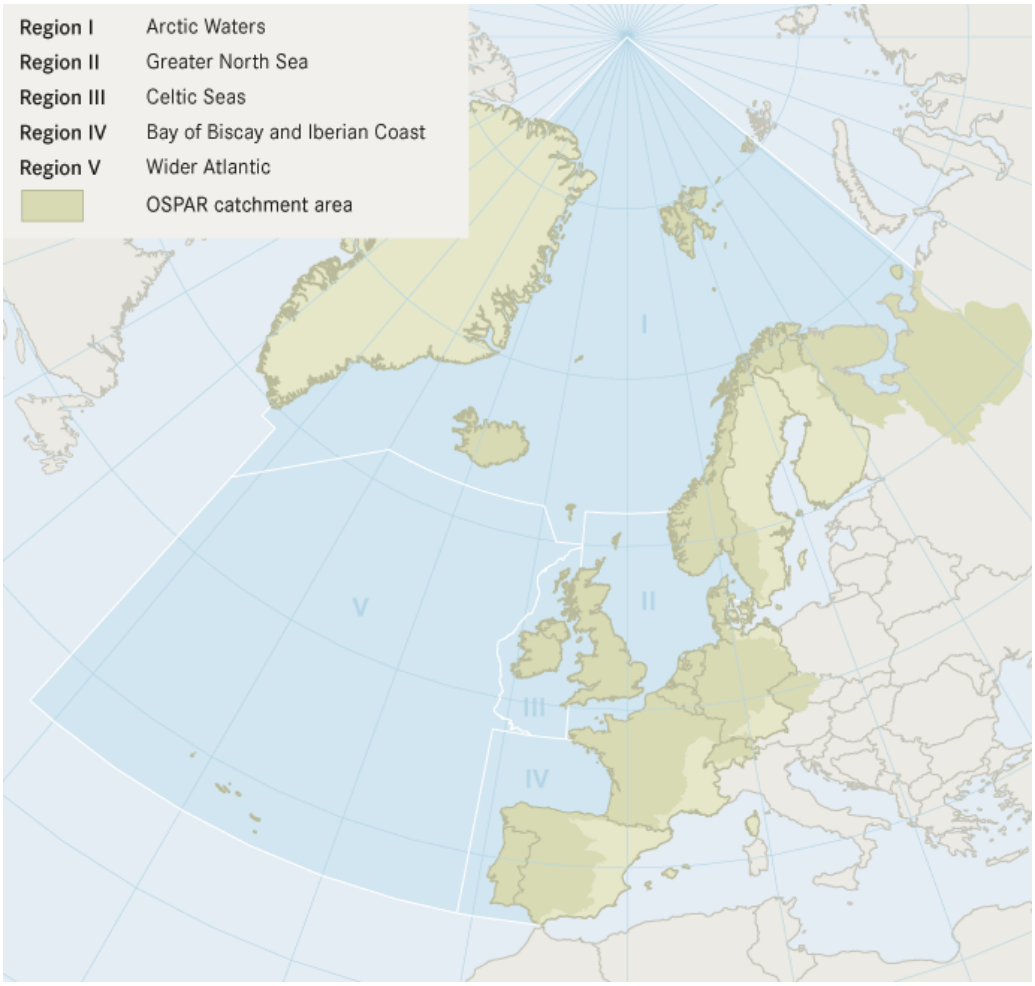


Figure 11: Map of the five regions of the OSPAR Maritime Area: I. Arctic water, II Greater North Sea, III Celtic seas, IV Bay of Biscay and Iberian Coast, and V. Wider Atlantic

Region I: Arctic Waters

Region I is the most northerly OSPAR region, characterized by its harsh climate and ice coverage, although the ecosystems of this region are still rich. In spite of its low population density, human activities such as fishing and offshore petroleum production remain significant.

Region II: Greater North Sea

The Greater North Sea is one of the busiest maritime areas. Offshore activities related to the exploitation of oil and gas reserves, and maritime traffic are very important. Two of the world's largest ports are situated on the North Sea coast, and the coastal zone is used intensively for recreation. The Greater North Sea is surrounded by densely populated, highly industrialized countries.

Region III: Celtic Seas

The Celtic Seas region contains wide variations in coastal topography, from fjordic sea lochs, to sand dunes, bays, estuaries and numerous sandy beaches. The large range of habitats in the region supports a diverse fish fauna. Although traditional maritime activities, such as fishing, take place in the Celtic seas, there is ongoing development of tourism.

Region IV: Bay of Biscay and Iberian Coast

The bottom topography of Region IV and coastlines are highly diversified, including the continental shelf and slope and parts of the abyssal plain. Ecosystems in Region IV are very rich, support a rich fish fauna and have a particular importance for migratory birds. Main human activities in Region IV are fishing, maritime transport and tourism.

Region V: Wider Atlantic

Region V represents the deep waters of the North-East Atlantic extending across the abyssal plain and the Mid-Atlantic Ridge, and including many seamounts. There have been recent discoveries of several different fragile deep-sea habitats (such as hydrothermal vents, carbonate mounds, coral gardens and sponge communities). Human population in the region is restricted to the Azores Archipelago. The main human activities are fishing and maritime transport.

A literature search was performed in 2015 to collect data from monitoring studies in water, sediment and biota in the OSPAR Maritime Area. The literature search provided results from multiple monitoring studies. Most of the studies contained information on microplastics monitored in regions II and III. Study results are subdivided into three classes: water, sediment and biota. The results of the studies are presented respectively in Appendix 8 to

Appendix 10. Per location the type, number and weight of particles were studied and reported, when available. Weight data were only available in one sediment monitoring study [19]. Types of microplastics found in the studies were fibres, granules, planar / granular fragments, plastic films, black carbon particles and foam.

3.1.1 Montoring in water

Seven studies were published towards the end of 2015, the same time that this literature search was being undertaken. The studies are summarized in Appendix 8. Data are available for regions II, III, IV and V.

The average number of particles in the different studies ranged from 0 – 102,550 particles per m^3.

The lowest average number was found in breakwater sampled along the French, Belgian and Dutch North Sea coast, and averaged 0.004 particles/m^3 (range 0 – 0.008) [141].

The highest number of particles in the OSPAR Maritime Area was reported in the Skagerak (North Sea area) in Sweden with a maximum of 102,550 particles/m^3 [97]. This number, found in only 1 of the 12 sampling points, might originate from careless handling in connection with loading of plastic pellets from the plastics industry in the nearby Stenungsund. The number of particles in the other 11 sampling points ranges from 167 – 2,400 particles/m^3.

A large scale study in sub surface seawater, covering a track length of 12,700 km of the Northeast Atlantic Ocean was performed by Lusher et al. [83]. The track length sampled included the shelf margins of the Atlantic continental shelf, Rockall Trough, Porcupine Bank, Celtic shelf, Celtic Sea and the Hebridean shelf and slope. Average microplastic numbers were of 2.46 ± 2.43 particles per m^3. The target of this study were microplastics over 250 µm in size. Furthermore, a method was developed to find out which of the following variables: wind speed, sea surface temperature, sea surface salinity and depth affected the number of microplastics sampled. Temperature and wind were the only variables to have significant effects ($p < 0.1$) on the distribution and abundance of microplastics in the North-East Atlantic. Furthermore, offshore samples had a significantly higher median number of microplastics than the other ecoregions as is presented in Figure 12 [83]. In four of the five ecoregions, fibres were the major type of microplastics found with reported percentages ranging from 61 – 95.9% and that the range is small: 0 – 2.46 particles/m^3.

There are indications of a loss of floating microplastics from the sea surface, suggesting there are removal mechanisms at play. These include UV degradation, biodegradation, ingestion by organisms, decreased buoyancy due to fouling organisms, entrainment in settling detritus, and beaching [40].

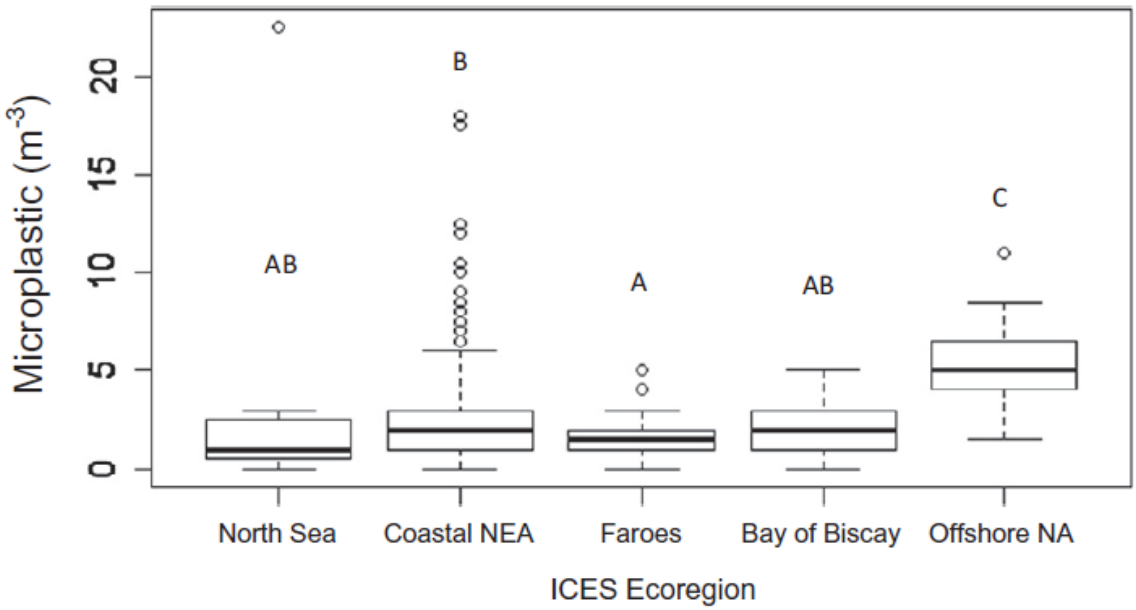


Figure 12: Number of particles collected per m^3 sub-surface sea water in the five different ecoregions, within the OSPAR Maritime Area. Areas sharing the same letter are not significantly different (Mann Whitney, p > 0.005) (Lusher, 2014).

3.1.2 Monitoring in sediment

The range of microplastic particles in sediment reported in different studies ranged from 0 – 49,600 granules and 100-1400 fibres/kg dry sediment. The variability of microplastic contents in sediments is high and strongly dependent of sample location. This can be harbours, deep sea sediments, tidal flats, or coastal sediments. Ten studies were available, which are summarized in Appendix 9. Areas sampled in the available studies were situated in regions II, III and IV.

There is a lack of consistency in sampling and extraction techniques used to quantify microplastics in sediment. Because of the large variety in techniques applied, comparison of reported microplastic concentrations between studies is difficult and often requires additional calculations based on assumptions. Many of the inconsistencies can be related to (i) differences in the lower and upper size limit implemented, (ii) the sensitivity of the applied extraction technique and (iii) differences in sampling technique leading to a wide variety of reporting units.

The highest numbers were reported by Liebezeit and Dubaish [80] and derived from samples taken at the high water line on Kachelotplate, an emerging sandbank island in the Wadden Sea. Because of its exposed position the island is strongly affected by storm surges occurring several times per year thus preventing the development of a stable dune belt so far. The findings are however considered to be controversial because of the visual method used, no confirmation by FT-IR and difficulties to reproduce the results.

In deep sea sediment an average of 1 particle per 25 cm^3 was found [143]. This equals 31 particles/kg sediment, assuming the weight of the sediment to be approximately 1.3 g/cm^3.

The mean number of granules reported in beach samples from German Wadden Sea islands is 201/kg dry weight and mean number of fibres 461/kg dry weight. At the Belgium coast an average number of 98.2 ± 37.2 particles/kg dry weight were found in beach samples [19].

The lowest average numbers of microplastics in sediment was 1.3 – 2.3 items/kg dw in beach sediment from the North Sea island of Norderney [28].

In general, number of particles found on the Kachelotplate (although controversial) is the highest followed by the number in the harbour. Lowest (estimated) numbers are found in the deep sea sediment.

In 2016 Spain started the MSFD subprogram BM-6 on microplastics in beaches. In the preparatory screening phase 10 beaches in the OSPAR Maritime Area were monitored. The concentrations detected ranged between 0 and 141 particles/kg with an average of 78.4 particles/kg. In the first routine monitoring conducted in autumn 2016, on four of those beaches, levels of 3.2 to 24.2 particles/kg and 36.7 to 248.1 particles/m^2, with an average of 12.9 particles/kg and 133.2 particles/m^2, were detected.

A large-scale study generally showed that deep-sea sediments seem to be most likely the main sinks for microplastics compared to surface waters. The abundance of microplastics (fibres) were higher in deep-sea sediments from the Atlantic Ocean, Mediterranean Sea and Indian Ocean than in contaminated sea surface waters [158].

3.1.3 Monitoring in biota

Nine studies were available. Results have been summarized in Appendix 3. Biota samples of regions II, III and IV were found. Species examined were: whale, multiple fish species, shrimp, oyster, mussel, crab and lugworm. In most of the studies the number of particles per weight unit was reported.

In all monitoring studies concerning fish species, gills of the fish were not examined. In female beaked whales stranded on the coast in Ireland [84], and in a baleen whale stranded in The Netherlands [5], microplastics were retrieved from the intestines. The estimated number of particles in the total digestive tract were estimated as 80 – 160 and 88, respectively.

In fish, the average number of particles in the gastrointestinal tract of fish (whiting, horse mackerel, poor cod, John Dory, red gumard, dragonet, redband fish, solenette and thickback sole) caught at the west coast of Ireland was 1.90 ± 0.10 [85]. Particles found were fragments, fibres, beads and film. Foekema et al. [43] reported a range of 1 – 4 particles in the digestive tract of fish (herring, gray gumard, whiting, horse mackerel, haddock, atlantic mackerel and cod). Rummel et al. [117] investigated the gastrointestinal tracts of 290 individuals of demersal fish (cod, dab and flounder) and pelagic fish (herring and mackerel) species from the North and Baltic Sea. Plastic particles were detected in 5.5% samples, with 74% being microplastics (mainly polyethylene). Pelagic feeders had a mean no. of 0.19+/-0.61 items per fish (mean + S.D.), while demersal feeders had a mean no. of 0.03+/-0.18 items per fish (mean + S.D.). Plastic ingestion was recorded in 3.4% of the demersal and 10.7% of the pelagic individuals, showing a higher frequency for pelagic feeders.

The number of microplastics in mussel *Mytillus edulis* was determined in four studies. The two highest values were found in field-collected mussels from the Oosterschelde [78] and Ter Heide North Sea, with 105 and 19 particles/g dw, respectively. In the other three studies the number of particles ranged from 0.1 – 0.36 particles/g ww. In one of these studies [141], the authors report that due to the use of concentrated HNO_3 during tissue digestion, the microplastic concentration reported could be an underestimation. Concentrated HNO_3 has a detrimental effect on (nylon) fibres, resulting in the total destruction of this type of microplastic during extraction (see 3.1.2.1). A second mussel species (*Mytillus galloprovincialis*) was also monitored and in this species an average of 0.12 ± 0.04 particles/g ww was found [150].

A review on microplastic quantification in other aquatic animals showed an average load of 0.13±0.14 particles g^{-1}ww in mussel meat [157]. An average value of 1.2 parts/g tissue was found in the lugworm *Arenicola marina* ; 0.68 ± 0.55 microplastics/ g ww and of 1.03 fibres/g ww were found in the shrimp *Crangon crangon* [29]. An average of 0.47 ± 0.16 particle/g tissue without depuration and 0.35 ± 0.05

particles/g tissue after depuration were found in the oyster *Crassostrea giga* [145]. The average number of particles/g dw in periwinkle (20), amphipod (11), pacific oyster from the Oosterschelde (87), blue mussel from the Oosterschelde (105), sand crab (0), pacific oyster from the Botlek harbour (30) and blue mussels from Ter Heide, North Sea (19) [7]. Many of the particles detected were fibres.

3.2 Concentrations of microplastic in rivers

Input of microplastics to the marine environment from land-based sources could be estimated from measured riverine concentrations and fluxes. A literature search was done in 2015 for selected rivers that float to seas of the OSPAR Maritime Area. The reliability depends on the accuracy of the measurement and sampling techniques applied. As described in the previous paragraphs there are also uncertainties related to the measurements. As an alternative way of estimating microplastic concentrations, effluent concentrations of sewage treatment plants have been used in combination with population density. The results are indicative, as literature that was published after 2015 has not been included.

3.2.1 Extrapolation from sewage treatment effluents

Rivers form a major pathway for land-based microplastics to the marine environment. The microplastic load depends on their discharges and on the number of inhabitants, sewage treatment plants and industrila activities in their catchment areas. Some rivers cross several countries, which requires an international approach for an effective approach to reduce microplastics.

Important rivers (rivers with a length of more than 200 km) that directly discharge to the OSPAR seas are listed in Table 21.

Table 21: Major rivers with direct discharges into OSPAR Maritime Area.

Draining into :	Countries	Length (km)	Catchment area (km^2)	Catchment inhabitant X 10^6	Discharge (m^3/s)
Atlantic ocean					
Tagus	Spain, Portugal	1,038	80,100	9	500
Loire	France	1,013	117,000	3.2	835
Douro	Spain, Portugal	897	97.290	2.2	714
Seine	France	776	78,650	15	500

Guadiana	Spain, Portugal	742	67,733	2.0	600
Guadalquivir	Spain	657	56,978	3.4	164
Shannon	Ireland	361	15,695	0.67	208
Severn	UK	354	11,420	0.75	61
Mondego	Portugal	234	6,645	0.67	80
North sea					
Rhine	Switzerland, Germany, Austria, France, Netherlands, Liechtenstein	1,236	185,000	60	2900
Elbe	Czeck republic, Germany	1,091	148,268	24.5	711
Meuse	France, Belgium, The Netherlands	925	34,548	9	350
Glomma	Norway	621	42,000	0.6	720
Scheldt	France,Belgium	360	21,863	6.5	120
Thames	UK	346	12,935	1.5	66

Taking the estimate a daily discharge of 1400 microplastic particles/person [146], and the population in the different river basins, it is estimated that via 52 billion particles per person per day are released via the sewage system to rivers that enter the Atlantic, and 141 billion particles are released to rivers that enter the North sea.

3.2.2 Monitoring in water

In total eight studies were available. In five of the studies surface water was sampled, in one study biota and sediment were monitored, in one study microplastics in biota and in one study microplastics in sediment were monitored. Rivers studied were: Rhine, Meus, Main, Seine, Bath, Ems, Elbe, Dalalven, Hamble, Itchen and Test.

The number of particles found in the rivers monitored is reported in

Table **22**.

Table 22: Overview of monitoring data in riverine samples

Location	Amount	Reference
Southampton water and its tributaries: rivers Hamble, Itchen and Test.	Total microplastics of four trials (trawl with manta net, 5 minutes downstream and 5 minutes upstream at 3 knots) per location: Itchen: 1155 Test: 348 Hamble: 296 Southampton water: 960	[46]
Rivers Meuse and Rhine	Meuse: 9.7 particles/m^3 Rhine: 56 particles/m^3	[140]
Rhine (Rotterdam, (NL), Dalalven (SE),	Dalalven: 4.5 particles/m^3 Rhine (2nd sampling): 252 - 589 particles/m^3 Rhine (3rd sampling): 35 - 101 particles/m^3	[148]
Rhine surface	Basel-Rotterdam: average 4.96/m^3 Basel-Mainz: average 1.13/m^3 Bad Honnef-Leverkusen: 3.97/m^3 Duisburg-Zuilichem: 11.89/m^3 Rotterdam: 1.59/m^3 Total average: 5.60 particles/m^3	[87]
Maas (at Eijsden), Rhine (Lobith and Bimmen)	Lobith, 4900 ± 540 particles/kg dw SPM Eijsden 1400 ± 520 particles/kg dw SPM Bimmen 1700 ± 390 particles/kg dw SPM, Recalculated to concentration in water based on a suspended matter (SPM) content in the Meuse of 9 mg/L [4]: Lobith, 123 particles/m^3 Eijsden 22 particles/m^3 Bimmen 51 particles/m^3	[11]
Scheldt, Ghent (Belgium)	6000-7000 particles /m^3	[146]

Overall, the number of particles found in the surface water ranged from 1.13 to 589 particles/m^3.

The highest number reported was 589 particles/m^3 in the Rhine (collected with a manta net with mesh size 330 µm) at location Verkeerspost Stad along the Nieuwe Maas in the centre of Rotterdam

The lowest number reported was 1.13 particles/m^3 also found in the Rhine between Basel-Mainz, sampled with Manta net, 300 µm mesh size [87].

The number of microplastics generally increase downstream [47, 87] . Most likely causes are sinks and retention of particles, turbulences, still waters and drift to the river banks where particles are washed ashore. Major microplastic retention may take place when the slope of the riverbed decreases, slowing down water flow velocity, whereby sedimentation rates increase for particles with a specific density. This could be enhanced when microplastics are settled by fouling organisms, increasing the overall density. Due to tidal dynamics and low discharge, the residence time of microplastics could be sufficient for biofouling and subsequent sedimentation. Sedimentation could further be enhanced by the tidal exchange with heavier, brackish water. Overall, the majority of particles analysed were identified as polyethylene. The type of particles found were spherules, pellets (flattened spherules), fragments and fibres.

The number of fibres in the Rhine was significantly smaller than in the other rivers monitored in the same study (approx. a factor 10 smaller than the Po river and a factor 20 smaller than the Danube river) [148]. The authors state that the fibres are most likely emitted through wastewater treatment and possibly atmospheric deposition. Above the sampling site on the Danube, there are two wastewater treatment

plants. The low population density in the Swedish Dalalven basin might mean that some households discharge their waste water after a very limited treatment into the river. It is likely that the results obtained give an indication of the quality of water emissions treatment before entering the river, combined with a significant burden in wastewaters. The number of small pellets (<1 mm diameter) found in this study was substantial. Since the shape was perfectly round, the authors believe that they were manufactured as such and not the result of fragmentation. Consumers (cosmetics) and industries are reported as potential sources of pellets.

An in-depth study on the abundance and composition of microplastics in the Dutch parts of the European rivers Meuse and Rhine was carried out in 2014 [140]. This study included an extending series of synchronized samples, collected at exactly the same locations and using the same method, in two running European rivers.

From January 10th 2014 up to June 23rd 2014, 17 weekly samples were taken by leading river water through a cascade of soil sieves for 72 hours. Two size fractions were acknowledged: 0.125-0.250 mm and 0.250-5 mm. The samples were cleared from organic litter, inorganic particulate matter and occasionally coal. Hereto a method is developed comprising successively the following steps: digestion with hydrogen peroxide, an interim filtration step using a mini-sieve, sample splitting, density separation with sodium chloride and sonication. Microplastics were visually identified, counted, and sorted out into four groups: films, white spherules, transparent spherules and miscellaneous microplastics. Occasionally scrubs were sorted out from the latter group as well. Fibres were not taken into account. For each sample, the individual groups were weighed separately.

Raman and Fourier Transform spectroscopy were used in combination with Principal Component Analysis (PCA) to identify the composition of the handpicked particles. Differences between both rivers were observed as in the Meuse no spherules were found. In both rivers films, scrubs and the majority of the miscellaneous microplastics were identified as polyethylene. The white spherules in the Rhine were verified as polystyrene, just as the transparent spherules up to 0.250 mm. For the larger transparent spherules, temporal variations in composition were observed comprising polyethylene, polypropylene and polystyrene. Also, a yet unidentifiable polymer was observed.

For the size range of 0.125-5 mm, average concentrations of 0.14 mg or 9.7 microplastics per m^3 were calculated for the Meuse, and 0.56 mg or 56 microplastics per m^3 for the Rhine. These figures form a lower limit, as particles can become lost and unevenly distributed during the laboratory processing and demonstrated is that even with secure visual selection, microplastics can be overlooked.

3.2.3 Monitoring in sediments

On average 2.03 x 10^3 particles per kg dw were measured in sediments from four different European rivers; Elbe, Bath, Seine and Ems [64]. The results from the four rivers was significantly higher than the results from subsequent sampling in the area around Amsterdam / Rotterdam. The highest amount was found in Hoek van Holland and the lowest in Westhaven (canal) but there was no significant difference between different aquatic environments. Fibres were found in all sediment samples and pellets were found in 72% of the sediment samples and made up 17% of the total microlitter found, with the highest amounts detected in the rivers. Black and blue particles made up 18% and 14% respectively, of the total amount of identified microlitter.

In another study a range of 786 – 1368 and 228 – 3,763 particles per kg were determined, in sediments from Main and Rhine respectively [67].

3.2.4 Monitoring in biota

Microplastics were monitored in a number of invertebrate species of biota sampled in Lake Veere, Hoek van Holland, IJmuiden and Amstel canal [64]. Microlitter was found in 9 out of 10 species and in 85 % of the samples. Filter-feeders contained a significantly higher amount of particles compared to other functional groups. In bivalves the concentration was a thousand-fold higher compared to the surrounding sediment and water. Average concentrations ranged from 5.82x 10^3 to 73.6 x 10^3 particles per kg dw. Both the lowest and highest average concentration were found in bivalves. The only species without microplastics was the grazer *Patella vulgata*. Of the microplastics found in biota, 28% were blue particles, 28% were black particles and 25% were fibres. An optimized method for sediment analysis increased particle recovery from 64% to 82%. Recovery for biota extraction showed an increase from 34% to 97%. Blue particles were found in 88% of the samples, black particles in 71% and in 94% of the samples fibres were present.

Fish species *Gobio gobio* sampled in French rivers (discharging in OSPAR waters) contained microplastics in their intestines; 11 to 26% of the 186 fish samples contained microplastics. Among 11 investigated stations, microplastics were clearly observed in fish from seven sites. Microplastics were not detected in fish from sites with low anthropogenic pressure such as from the upper area of river basins [120].

3.3 Effects of microplastics in (marine) ecosystems

The purpose of this chapter is to give a short overview of possible effects of microplastics to the marine ecosystem. For this purpose we will cite relevant publications. It is not the intention to give a full overview of all publications available for an effect discussed. For a more extensive literature overview the reader is referred to more extensive reviews [48, 84]. Many different studies, mainly on bivalves, crustaceans, annelids or fish have been conducted to demonstrate (sub-lethal) impacts of microplastics, however, most of them are under laboratory conditions rather than field studies. General findings are that microplastics may reduce the health, feeding, growth and survival of organisms from lower trophic levels [49].

3.3.1 Different kinds of exposure and their relation to potential effects

In order to assess the effects of microplastics on individual species or a population, different kinds of exposure should be considered. The kinds of exposure to organisms considered in this chapter are:

- External exposure adsorption of the microplastics to the exterior of an organism.

External exposure like adsorption to the exterior of algae [6, 8] has been reported to have an effect on algae, and it is also a potential route for transfer to higher trophic levels. Uptake in the gills as reported for crab [12] and mussel [153], is considered a form of external exposure [48].

- Internal exposure through ingestion of the microplastics.

Ingestion of microplastics has been examined in many studies and was shown for many different types of organisms including crustacean, molluscs, worms, fish, birds and sea mammals [48]. Ingestion does however not indicate that microplastics accumulate in organisms, egestion has been shown in several cases and is further discussed in the next section on factors affecting exposure and toxicity.

- Secondary exposure by consumption of pray with microplastics in their intestines.

Secondary exposure can be expected for decomposers (like small crustacean and mussels) and consumers (like fish and crab). For decomposers it could consist of microplastics sorbed to the exterior of algae [8] consumed by filter feeders like mollusc or zooplankton species.

- Substrate for habitation.

The last kind of exposure to be mentioned is that the plastic particles can form a substrate for habitation. This considers the fact that microplastics will form a habitat for a specific kind of organisms that would otherwise not be present in such amounts at that specific location. Their increased presence as shown for insects [51] could endanger indigenous species. Microplastics are also mentioned as potential mode of transport for invasive species [94].

3.3.2 Factors affecting the exposure and toxicity

There are many factors that could influence the actual exposure of organisms to microplastics and potential toxic effects that might occur. These factors as given in the list below will be discussed in this section.

- size
- abundance
- primary or secondary particles
- ageing (sorbing and fouling)
- colour
- animal health / pre-exposure
- ingestion and egestion
- translocation
- density
- shape
- chemical composition

3.3.2.1 Size and abundance

With the size range in the current definition for microplastics "being solid particles that are smaller than 5 mm", it may be clear that it covers a wide variety of different kinds of particles. The size of the particles has been shown to affect the toxicity. For example, particles in the nano meter range (0.05 µm) posed a lower chronic toxicity to marine copepods than particles of 0.5 and 6 µm [73]. With increasing size the level of toxicity has been reduced. This study is also one of the few that showed a dose effect relation for microplastics of 0.05 µm and 0.5 µm, chronic EC50 values of 0.16 and 23.5 mg/L for survival of second generation copepods were derived. In acute tests, no effects were observed and for the larger particles of 6 µm no effects were observed in both the acute chronic tests.

In another study, for 385 gooseneck barnacles examined, it was reported that in the smaller species collected, in only one case ingested plastic particles larger than 300 µm were observed while for the larger animals these were observed in more than one third of the collected animals [50]. Similarly, no accumulation of plastic in the intestinal tract of wild caught fish was found [43]. It was stated that the particles sizes were

too small to accumulate in the intestinal tract. It was considered that effects are dependent on the particles size relative to the fish size and therefore effects at early life stages cannot be excluded.

Polystyrene particles with different sizes (0.05, 0.5 and 6 µm) and material properties in experiments on microalgae were found to have no effect on the photosynthesis, but growth was negatively affected (up to 45 %) by uncharged particles at high concentrations (250 mg/L) [125]. These adverse effects increased with decreasing particle size. However, another study showed that charged PS nano-sized plastics (0.02 µm) inhibited photosynthesis and reduced population growth and chlorophyll concentrations in the green algae *Scenedesmus obliquus* [49]. The update and tissue accumulation of polystyrene microplastics was tested in Zebrafish and revealed that smaller particles (5 µm) accumulated in the gills, in the liver and in the gut, while larger particles (20 µm) accumulated in the gills and in the gut. Both sizes caused inflammation and lipid accumulation in the liver [81].

These studies indicate that size is an important factor but also that the relation between the size of the particles and the size and / or age of the animal should be taken in to account.

3.3.2.2 Primary or secondary particles, ageing, colour and animal health

For primary and secondary particles it can also be expected that their toxic properties will be different from each other but currently no study could be retrieved examining the differences between these kinds of particles. Effects of ageing of microplastics has however been examined and these effects are expected to apply to secondary plastics as they are more affected by natural processes than primary particles. Although for the latter, ageing can also apply depending the time between release and exposure of potentially affected organisms. Aging can involve different processes. For example, algae can grow on the surface of microplastics (fouling) or, the other way round, microplastics can sorb to the exterior of algae. Polystyrene microplastics that were pre-exposed to algae caused mortality to *Daphnia magna* whereas this effect was not observed for fresh particles [6]. An explanation for the observed mortality is that the microplastics would adsorb to algae and as such increase the exposure of daphnids to microplastics through consumption of these algae.

In the same study the effect of the plastic particles on reproduction was enhanced by the fouling of the particles. Although the test organism was not a marine organism, the observed effects could potentially also occur in marine organisms. On the contrary, sea urchins have been reported to avoid ingestion of fouled particles while fresh particles were ingested [63]. Similar to fouling the colour of microplastics is also reported to affect their uptake by organisms. showed that white particles were more ingested by Common Goby fish (*Pomatoschistus microps*) than red and black particles when offered separately or in combination with algae [27]. However it should be noted that not only the colour of the particles differed but also the density as the red particles were slightly lighter than water (0.98 g/ml) and the white and black particles heavier (1.2 g/ml and 1.15 g/ml respectively). Also, the black particles were paramagnetic where the white and red particles were not. These factors could also have influenced the differences in uptake and were not investigated in this study. Nevertheless algae were ingested much more often than plastic particles when both were offered simultaneously. It was also observed that fish from a more contaminated area with a supposedly decreased health status, were not able to distinguish between algae and white particles while they could distinguish the red and black particles from the algae. The more healthy fish originating from a cleaner location could distinguish all plastic particles from the algae and ingested mainly algae.

3.3.2.3 Ingestion, egestion and translocation

Whether ingested microplastics will pose an effect to organisms after ingestion does also depend on whether the particles are egested again. When the latter occurs, effects like blockages or further uptake into organs are unlikely. No accumulation of plastic particles was found in wild fish and gooseneck barnacles [43, 50]. Both studies indicated that particle size is an important factor for ingestion and related effects of microplastics. Nevertheless ingestion by mussel [153] and crab [12] can result in uptake in the digestive gland (hepatopancreas) and even in the circulatory fluid of the mussel [14] indicating that egestion is not the case for every particle and translocation within the organism does occur. Although the latter study mentions accumulation of the plastic particles, the actual extend of this is unclear since ingestion and egestion has not been quantitatively monitored and it is uncertain which fraction of the ingested particles remains in the mussels.

As also indicated in paragraph 3.1, recent publications show different concentrations of particles in commercial mussels like those collected at contaminated sites ranging from 0.05 to 0.34 particle per gram wet weight [150] but also concentrations of 105 and 19 particle per gram dry weight [78]. Although the units differ, wet weight vs dry weight, the difference between the observed concentrations is large. This difference could be caused by the use of HNO_3 in the analysis, which could remove high amounts of plastic fibres from the extracts [20]. This suggests that accumulation of plastic fibres could occur since they might be missed in the analysis and blockage of intestines is not excluded. Even without accumulation, the translocation of particles to organs could be an important factor for the induction of toxic effects by the plastic particles. Also studies looking at ingestion by crustacean do not report accumulation of high numbers of plastic particles in the animals [22] though in neonate daphnids formation of a lump was observed indicating accumulation of the min the neonates [6, 22]. For marine isopods it is reported that concentrations of microplastics in the faeces was the same as in the food [57]. Also for worms, in general no accumulation is reported [7] but a longer passage time is indicating a longer time for interaction between the particles and organisms [159]. In general it can be concluded that accumulation of microplastics could occur but it is not always the case and quantitative figures are not available. Especially on the potential accumulation fibres more research is recommended.

Secondary exposure could occur because many marine organisms are continuously exposed to microplastics and will have microplastics in their intestines when they are consumed. Transfer of microplastics within the planktonic food web[122] and also from mussel to crab [42] were observed. It is unclear of this kind of exposure might cause different effects than direct ingestion. Microplastics attached to algae may be a pathway for microplastics from the water to marine benthic herbivores. *L.littorea* had no feeding preference for algae with or without microplastics attached to the surface. However, the occurrence of microplastics in the faecal pellets indicated that plastic particles may not rapidly accumulate in the intestines [56].

3.3.2.4 Density, shape and composition

Factors that may affect the toxic effects but which have been less examined are density, shape and composition of the polymers. Shape and composition of the polymers have to date been found to be examined in only one study each [2, 57]. In a higher number of studies, it has been shown that co-contaminants like additives or compounds sorbing to the microplastics are important for the toxic effects of microplastics [82, 96] but microplastics can also reduce the effect of other compounds [101]. The effects of co-

contaminants and additives will be discussed in more detail in the next section on potential effects of microplastics.

3.3.2.5 Conclusion on factors

Little is known of many factors potentially affecting the toxicity of microplastics. The most important pathway for exposure to microplastics is ingestion but this is depending of the relation of the particle size and size of the organism. Although particles are ingested and as such can be transferred between trophic levels, accumulation of high levels within organisms has rarely been reported because microplastics are in general being egested again with exception of a potential fraction that could remain in organs in the digestive tract and possibly accumulation of fibres. Although occurring in small amounts, translocation from the digestive tract to organs could be an important effect of microplastics.

3.3.3 Overview of potential effects of microplastics

It is generally acknowledged that not only the particles themselves can pose a (physical) effect on organisms but also contaminants adsorbed to the particles can pose a (chemical) effect. Likewise, additives or residues of monomers in the plastics could leach from the plastics and transfer to the organisms. Based on this, potential effects are discussed in the following groups of effects:

- particle related physical effects
- effects of additives
- effects of contaminants

3.3.3.1 Particle related physical effects

Blockage of the gastronomical tract is an effect often considered for (micro)plastics. This clearly is a physical effect and depends on actual accumulation of the ingested particles. As discussed in the previous section, actual accumulation of high number of microplastics in organisms is currently not reported and therefore it is considered likely that accumulation does not generally occur. Likewise, blockage of the intestinal tract is considered a rare event.

Nevertheless, effects of microplastics have been reported and a likely explanation is that they take the place of normal food and therefore reduce the total energy content of the food ingested. Marine worms exposed to fresh polystyrene and PVC particles showed reduced feeding activity and reduction in weight which was attributed to reduction in energy assimilation and / or due to the polystyrene particles present in the ingested sediment [7,159]. Similar concentrations of silica sand did not cause a reduction in feeding activity [159]. In both studies, significant effects were only observed starting at plastic concentrations in the sediment of 5% and 7% wet weight. Lugworms (*Arenicola marine*) produced less casts when exposed to microplastic pollution in the sediment they inhabit [54]. Metabolic rates were increased and microalgal biomass decreased. Responses differed with different types of plastic, indicating that different materials may have different effects. Similarly, for sea urchins reduced growth of larva was observed after ingestion of polyethylene particles [63] and this was also contributed to reduced feeding efficiencies associated with the ingestion of micro particles. In this case the effects were observed at a concentration (300 particles/ml) which was an excess of those currently recorded in marine habitats. Molluscs species exposed to microplastics (30 nm polystyrene nanosized particles (0.1, 0.2 and 0.3 g/L) showed an increase in pseudofaeces production and a decrease in filter-feeding activity [155].

Reproduction effects in daphnids were also contributed to particle related effects because reduced food availability or energy uptake causes changes in reproduction and in the offspring [6, 22]. The latter study also reported a change in feeding behaviour, as a shift in prey size was observed. In the presence of 20 µm

microplastics, the copepods shifted to catching smaller algae presumably trying to avoid ingesting microplastics. For copepods another study is available where a concentration effect relation has been shown [73]. In this study, the (chronic) effect observed on reproduction was also subscribed to insufficient nutrition. The chronic EC50 values determined for three different size of polystyrene particles are given in Table 23 below. From this table, it can also be seen that the offspring is more susceptible to effects of the particles than the parent.

Table 23: EC50 values determined for the effects of chronic exposure of marine copepods to polystyrene particles [73]

	EC50 determined	
Particle size	parent	offspring
0.05 µm	2.15 mg/L	0.16 mg/L
0.5 µm	>25 mg/L	23.5 mg/L
6 µm	>25 mg/L	>25 mg/L

For larger organisms like fish, particle related effects linked to food intake are less explicitly observed. In four out of five fish species, no significant relation between ingested plastic and physical condition was observed in wild caught fish in the North Sea [43]. Only for haddock a significant relation was observed, but the observed effect was only marginal. Therefore, it was concluded that the ingestion of plastic was unlikely to have affected the condition of the fish. Nevertheless, a reduced food intake by juvenile common goby was observed, suggesting that they would avoid ingestion of microplastics [27]. In the same study a decrease in the predatory performance of *P. microps* (common goby) after the exposure to microplastics was found. Weight loss, changes in metabolic performance and changes in feeding behaviour after feeding nano-sized polystyrene (1 to 100 nm; 0.01% w/v) were also reported [18].

Not all particle related effects of microplastics are related to a decrease in food intake; obstruction of the flagella was also mentioned as cause for the observed effects [6, 8]. Also reduction of photosynthesis and consequently reduced algal growth was mentioned as an indirect effect of microplastics[8], but this could not be confirmed by another study [6]. Damage to the cell walls was also mentioned as potential cause for the observed effects [8]. Experiments with the limnic zooplankton species *Daphnia magna* showed that short-term exposure (96 h) to 1- µm and 100- µm polyethylene particles (12.5 – 400 mg L^{-1}) led to immobilisation increasing with dose and time with an EC_{50} of 57 mg L^{-1} after 96h [113]. European sea bass (*Dicentrarchus labrax*) were chronically exposed to microplastics and their distal part of the intestine was most affected by pathological alterations [107]

For mussels and crabs, it was already mentioned that they ingest microplastics and that a fraction of the ingested particles accumulates in organs like the digestive gland [2, 12]. After uptake, effects at cell level, to tissues and to organs can be expected. Cellular effects are observed after the uptake by mussels of polyethylene and polystyrene particles contaminated with pyrene. Controls without pyrene were included [2]. Significant immunological effects, neurotoxic responses, cellular and genotoxic effects resulting from contaminated and uncontaminated microplastics were observed.

Filter-feeding Pacific oysters, exposed to polystyrene microspheres during reproductive cycles, showed modified feeding and reproductive disruption (a decrease in oocyte numbers, in the diameter, in sperm velocity and in the development of the offspring) [127]. Translocation of polystyrene with sizes between 2 and

16 µm to the circulatory fluid (haemolymph) of the mussel has been observed [14]. On effects of microplastics in the circulatory fluids of marine organisms little is known but the authors cited a study where polystyrene particles administered in the lungs of hamsters caused thrombosis in pulmonary blood vessels. For exposure through normal ingestion this effect has so far not been shown.

Exposure of mussels to high-density polyethylene particles with particles sizes smaller than 80 µm at a concentration of 2.5 g/L was followed by cellular uptake on epithelial cells of these microplastics [153]. There was also an increased formation of white blood cells (granulocytoma) containing vacuoles with accumulated microplastic particles. The formation of granulocytoma is mentioned in the publication as an inflammatory response often associated with environmental pollution. Furthermore, lysosomal membrane integrity was also reported to be disrupted that would indicate stress and toxicological responses.

External exposure can result in uptake of microplastics in gills as reported for mussel [2, 153] and crab [12]. Effects at enzyme levels in the gills after exposure was also reported [2]. This indicates that external exposure can also lead to effects at cell level.

A picture for the different modes of action potentially involved in the particle related effects of microplastics are shown in Figure 13. In this picture, it is postulated that all effects are related to energy balance in the organisms. It should be noted that not all effects will occur in every organism and not from all kinds of particles. Also, not every kind of effect will have an equal effect on the energy balance of an organism but with the current knowledge it cannot be concluded which kind of effect will have the highest impact.

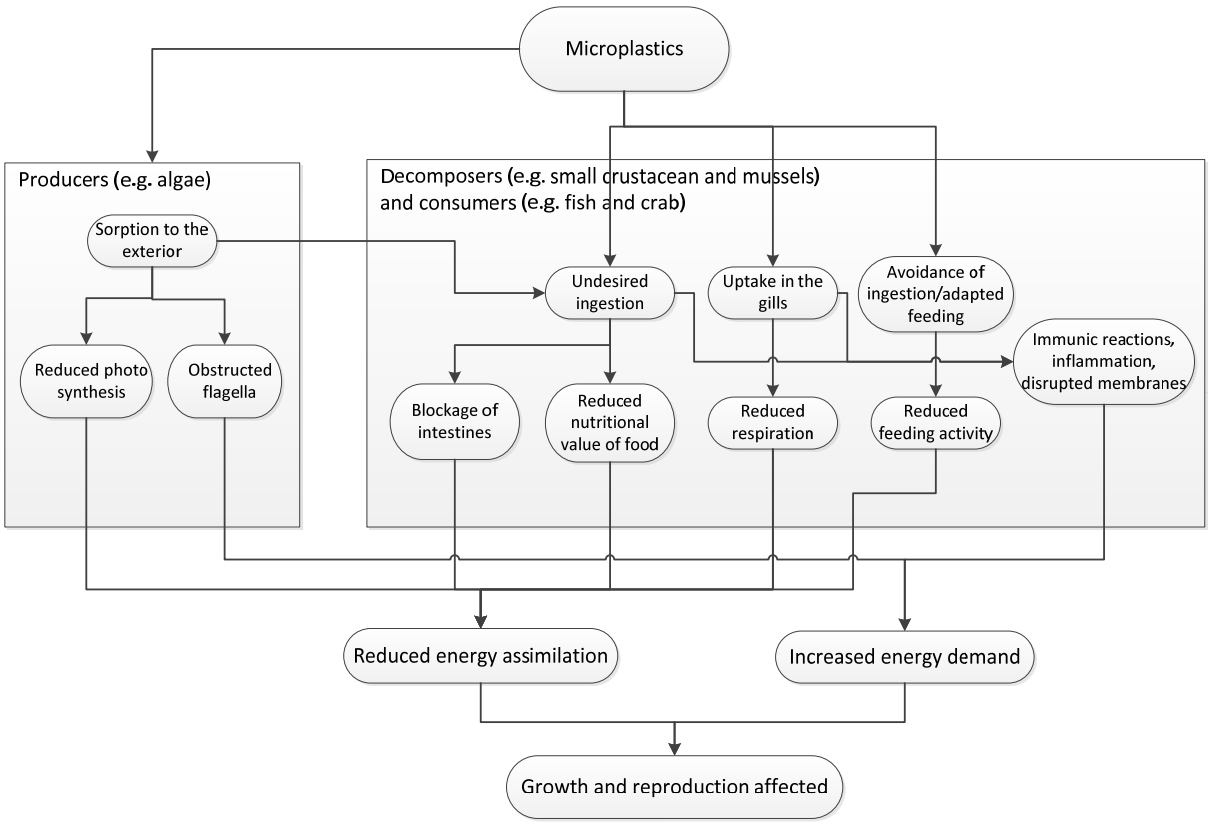


Figure 13: Overview of potential modes of physical action for toxic effects of microplastics in the marine environment

In Table 24, an overview is given of the exposure concentrations applied in the laboratory studies discussed above. The concentrations are not always comparable since the units do not overlap. However, it may be

clear that in most cases high concentrations are applied to obtain an effect. These concentrations should be compared to the current environmental concentrations to conclude whether the current environmental concentrations pose an ecological threat. It should also be noted that most laboratory studies performed their experiments with spherical particles. These particles are not representative for secondary microplastics that are unlikely to be as smoothly shaped as spherical particles. Furthermore, it may be noticed that many of the microplastics used in effect studies are in the nanometre range, whereas the monitoring studies are usually limited to the micrometer size (sizes >20 µm, which is considered the detection limit of FT-IR, or to >300 µm which is the mesh size of manta nets). Currently, the risk assessment suffers from a mismatch between type, size and concentration of particles measured in monitoring studies and type size of particles used in effect studies.

Table 24. Overview of test concentration and particles tested in laboratory toxicity experiments discussed in this section

Study	Organism examined	Exposure concentration	Particle tested*
Besseling et al. [7]	Lugworm	0.074 to 7.4% of feed (w/w)	PS, 400-1300 µm
Besseling et al. [6]	Algae	44-1100 mg/L	PS 0.07 µm
Besseling et al. [6]	Daphnid	0.22-150 mg/L	PS 0.07 µm
Bhattacharya et al. [8]	Algae	0.08-0.8 mg/L	PS, 0.02 µm
Brennecke et al. [12]	Crab	100-1000 mg/kg sediment	PS, shredded pellets
Browne et al. [14]	Mussel	500 mg/L	PS, 3 and 9.6 µm
Cole et al. [22]	Copepod	75 particles/ml	PS, 20 µm
De Sá et al. [27]	Fish	30 particles/fish	PE, 420-500 µm
Kaposi et al. [63]	Sea urchin	1-500 particles/ml	PE, 10-45 µm
Lee et al. [73]	Copepod	0.0125-25 mg/L	PS, 0.05, 0.5, 6 µm
von Moos et al. [153]	Mussel	2500 mg/L	PE, < 80µm
Wright et al. [159]	Lugworm	0-5% of the feed (w/w)	PVC, 130 µm
Avio et al. [2]	Mussel	0.050 mg/L	PE and PS, <100 µm
Hämer et al. [57]	Isopod	-12 or 120 particle /mg food -20 or 350 particle/mg food -0.3 mg/g food	-PS beads, 10 µm -PS fragments, <100 µm -PA fibres, <1000 µm

Plastic types are: PS = polystyrene, PE = polyethylene, PVC = polyvinyl chloride, PA = polyacryl

Conclusion on particle related effects

Particle related effects mostly observed are a reduction of energy intake as the ingested microplastics replace part of the ingested food or organisms spend time and energy avoiding microplastics. These effects were observed after exposure to high levels of microplastics, currently not observed in the environment. Nevertheless, microplastics are reported to be taken up in organs and immunological responses and effects at cell and enzyme level can occur. All these effects will potentially affect the energy balance and as such affect growth and reproduction.

3.3.3.2 Additives

Plastic can contain additives that are added in the production process to enhance the performance of the plastic. These additives generally consist of plasticisers (phthalates), dyes and flame retardants. For these kinds of additives, a lot of research is performed on the aquatic toxicity of the individual substances. However, their effect in combination with plastics has been explored in only a limited number of studies. In one case special attention is paid that the plastic used was free of additives [153] but this reveals no details on

the effect of additives. Also coloured particles were tested but here no attention is paid to potential chemical effects of the dyes within the particles [27].

One relevant study for the marine environment addressed co-contaminants in beach collected plastic pellets and virgin poly ethylene particles on sea urchins [96]. Sea urchins were not exposed directly to the particles but only to the water phase which is in contact with the particles. The water phases from both kinds of microplastics caused toxic effects to the sea urchin embryos but the toxic effects of the virgin particles were higher than of particles collected at beaches. This can well be explained by the fact that for example plasticisers like phthalates leached from the plastic and are unlikely to be present in high amounts in the beach collected particles. Persistent hydrophobic contaminants that might have sorbed to the particles in the environment are unlikely to find their way to the sea urchins tested as they will remain sorbed to the particles. Therefore, this experiment clearly shows that virgin particles can pose a kind of toxic effect that will not be posed by particles present in the environment. This fact makes it difficult to extrapolate effects of virgin particles observed in laboratory tests to the field, unless special attention is paid that additive free particles are used.

Polybrominated diphenyl ethers (PBDEs) are used as flame retardants and have been applied often in plastics. At least some PBDEs were transferred from the plastics to the seabirds that ingest the plastics [130]. The plastic content in several birds was compared with the presence of PBDEs in the body fat of the birds and with PBDEs in plastics in their pray (lantern fish). For the lower-brominated PBDEs their profile of relative concentration in the birds was comparable to that analysed in their pray. However, two higher brominated congeners were not detected in the pray while they were found in the plastics in the stomach of the bird where these congeners were also detected. The particles were near and over 5 mm in size; for smaller microplastics, it can be expected that these have a higher rate of transfer as these have a higher relative surface area. Health effects were not observed in this study.

More studies address the transfer of other hydrophobic persistent substances (for instance polycyclic aromatic hydrocarbons and polychlorinated biphenyls) from plastics to marine organisms but these are not added in the production process of the plastic but sorbed to the plastics during their presence in the environment. These studies will be discussed in the next section.

3.3.3.3 Contaminants

The contaminants generally considered for sorption to microplastics are polychlorinated biphenyls (PCBs), polycyclic aromatic hydrocarbons (PAHs) and PBDEs. Although the latter were also added to the plastics during production processes, they are also present in background concentrations as contaminant in the marine environment. Rochman et al. [116] showed an increase in the concentration of PBDEs in LDPE particles after three months in the marine environment. With these particles, the uptake of persistent substances from plastic particles in the food to body lipids was examined. These particles added to food fed to fish, caused a significant increase of PBDEs in the fish lipids. Other components in the food that also contained PBDEs could not be related to the observed effect. This uptake as observed for PBDEs in general (with exception of one congener) was not significantly observed for PAHs and PCBs in general. But for the individual congeners chrysene and PCB28 a significant uptake was observed, too. Toxic effects to the liver were also observed but since cod liver oil was part of the diet (including the control) and contained the same persistent substances as were monitored in the study, it cannot be concluded if this could be contributed to the substances transferred from the plastic to the fish lipids or from the plastic particles themselves. An interesting fact of this study is that the fish were exposed at levels of 10% microplastics in

the diet (w/w, comparable to 8 µg/L) for which it was claimed that it is comparable to concentrations observed in the marine environment.

Rainbow fish was exposed to microbeads from personal care products containing organic pollutants. Fish showed higher concentrations of polybrominated diphenyl ethers and resulted in increased accumulation after continuing the exposure [154].

Browne et al. [16] found that the combination of PVC with sorbed triclosan altered feeding behaviour and caused mortality in lugworms. When exposing lugworms (Arenicola marina) to sand with 5% microplastic that was presorbed with pollutants (nonylphenol and phenanthrene) and additive they found that the ability of coelomocytes to remove pathogenic bacteria was reduced by uptake of nonylpheno from PVC or sand, and that uptake of triclosan from PVC diminished the ability of worms to engineer sediments and caused mortality. Exposure to PVC particles alone made worms >30% more susceptible to oxidative stress.

Transfer of pyrene has also been shown from microplastics to mussel [2] and fish [101]. In the case for mussels, exposure to microplastics with pyrene sorbed to it, caused an increase of pyrene in the gills and digestive gland [2], this increase was not observed for the control and after exposure to virgin microplastics. Immunological responses were observed but these could not be contributed to the pyrene present and were supposedly mostly induced by physical effects of the microplastics. This study did not examine the uptake of pyrene without being sorbed to microplastics, so it can only be concluded that pyrene moves from the microplastics to the organs of the mussel but not that the microplastics actually mediated in the uptake as the uptake might also have occurred if the pyrene was present in the water in any other form. The study on the transfer from microplastics to fish [101] did look at the formation of bile metabolites of pyrene after exposure of pyrene alone or in combination with polyethylene particles to Common Goby. The combined exposure of pyrene with the microplastics resulted in a significant higher level of bile metabolites than when pyrene was exposed alone. This indicates that the microplastics tested increased the bioavailability or biotransformation of the pyrene. In this study was also shown that mortality caused by pyrene was delayed by the microplastics, the exact cause of this delay could not be clarified in this study.

Besseling et al. [7] looked at the bioaccumulation of PCBs by lugworms from sediment. For the lowest concentration of microplastics in sediment tested (0.074% w/w) an increase is bioaccumulation was observed but this was only significant for three PCB congeners. The actual extent of this observation is unclear since this effect was not observed for higher concentrations of microplastics in the sediment (0.74 and 7.4%). The highest concentration has affected the feeding rate and as such could have affected the bioaccumulation. A similar study was performed with lugworms exposed to sediment that contained 5% (w/w) of PVC particles that were contaminated with phenanthrene, a PBDE, triclosan and nonylphenol [132]. At the end of their trial the worms had higher concentrations of the contaminants than the sediment. This indicated a transfer of the contaminants to the worms. It should however not be ignored that the whole process is an equilibrium, contaminants can move from the plastics to the organisms but also the other way around. Likewise, the contaminants can move to the sediment and suspended organic matter. In modelling studies it was also considered that clean microplastics can reduce the concentration of contaminants in animals and also that different plastics have different sorbing capacities[68, 69, 132]. The model outputs confirmed that uptake of contaminants form plastic particles is possible but also that "clean" plastic particles can reduce the level of a contaminant in an organism [69]. The main conclusion from these studies however was that at the current environmental levels of microplastics and contaminants, the accumulation of contaminants from the microplastics plastics is marginal in comparison to the other accumulation processes occurring [68, 69].

Interaction between metals and microplastics is also possible, as was demonstrated for the effect of microplastics on the acute toxicity of chromium VI to juvenile fish [82]. The reducing effect of chromium (VI) on the predatory performance was increased by the presence of polyethylene microspheres. At combined exposure of the fish to the microplastics and Cr (VI) higher effects were observed on predatory performance and specific enzyme activity than for the contaminants alone. This would indicate toxicological interactions between the two kinds of contaminants. The results in this experiment were not always consistent since pre-exposed fish originating from a contaminated area reacted in some cases opposite than those from a less contaminated area. For the interpretation of the effects mixture toxicity should also be taken into account, because different contaminants can increase or reduce each other's toxic effect in an organisms (resp. synergism and antagonism)[128].

Conclusion on contaminants

Plastic particles can be as source and sink of chemicals and may impact the health of wildlife. However, interactions between microplastics and other contaminants have been reported but the actual extent of these effects in the marine environment is still unknown. In the marine ecosystem many processes are involved and many contaminants are presents. All these contaminants can interact with each other and can reduce or increase each other's effect. In this view, the effects of microplastics are one of the many influencing the health of the marine organisms. Therefore the effects of the microplastics should be considered in the view of the total pollution of the marine ecosystem.

3.4 Microplastics and human food safety

Ultimately, the risks microplastics pose to human health through consumption of contaminated food need to be considered, but these are still difficult to determine.

In general, we can say that the available data on the occurrence, toxicity and fate (what happens after digestion) of microplastics and nanoplastics in food are insufficient for a full risk assessment. Researchers are still attempting to answer these questions with focus on specific areas. There is also no legislation available for microplastics and nanoplastics as contaminants in food.

Data on the occurrence of microplastics and nanoplastics in food is limited. Methods for identification and quantification of microplastics in food, including seafood, have been reported in literature. However, in some of the studies, quality assurance to avoid contamination from the air and equipment is not described, and it is not always clear how a particle is identified as being a 'plastic'.

In the data available, we see concentrations of microplastics in fish but since microplastics are mostly present in the stomach and intestines, they are usually removed and consumers are not exposed to them when eating the fish. But in crustaceans and bivalve mollucs like oysters and mussels, the digestive tract is eaten so there is some exposure for the consumer. Besides seafood, emerging evidence shows that the microplastics, especially synthetic fibres, have been detected in a variety of foods, including drinking water, beer, honey, sugar, and table salt. Currently there are no data available on nanoplastics in food.

Toxicity and toxicokinetic data are lacking for both microplastics and nanoplastics for a human risk assessment. Based on a conservative estimate by EFSA [37] the presence of microplastics in seafood would have a small effect on the overall exposure to additives or contaminants. EFSA estimated the average intake for a portion of mussels (225 g) could contain 7 micrograms of microplastic. Even if this amount of material contained the highest ever measured concentrations of PCBs or BPA, it would make a small contribution to overall exposure to these substances: it would, in the worst-case scenario, increase PCB exposure by less than 0.01 percent or BPA exposure by less than two per cent.

Also, there is no available literature on the fate of microplastics during the processing of seafood. Microplastics are likely to originate from other sources than the food itself, e.g. processing aids, water, air or being release from machinery, equipment and textiles, although there is no available literature on this issue. It is therefore possible that the amount of microplastics increases during processing. The effect of other processes, e.g. cooking and baking, on the content of plastics is not known.

Whether the microplastics in food are harmful to consumers, it is too early to say, but it seems unlikely. The risk of chemical contaminants being transferred to humans depends on: i) the retention time of the particles in the food or seafood, ii) the rate and degree to which contaminants are released from the plastic, iii) the degree to which fine particles might be translocated from stomach of seafood to other tissues, and iv) the degree to which chemical contaminants can transfer from the consumed seafood to human body. The presence of microplastics in food could potentially increase direct exposure of plastic-associated chemicals to humans and may present an attributable risk to human health. However, on the basis of current evidence, the risk to human health appears to be no more significant than via other exposure routes [137].

4 General conclusions

In the rapidly developing field of microplastics research new information is generated at a high rate. The bulk of the information provided by this report was derived in 2015 by an exhaustive literature search in peer reviewed papers, supplemented with relevant publicly available grey literature published on behalf of national governments, the EU or non-governmental organizations. However, some additional data and information were included later up to spring 2017.

The presented monitoring data suffer by a lack of harmonization of sampling and analytical methods and are therefore difficult to compare. Not all the studies used procedures to minimize the possibility of contamination of samples or techniques to confirm the synthetic nature of the particles. This could have led to overestimations of microplastic counts in certain cases. On the other hand, when specific colouring techniques or harsh purification methods were used, certain types of plastics could have been missed, leading to an underestimation of microplastic counts.

The models used to calculate emissions and distribution of microplastics are a simplification of reality. This has led to several assumptions for inherently variable model parameters and for parameters with data gaps. The reported range in source emissions reflects these uncertainties. Assumptions are described in the chapters and could be refined if better data become available.

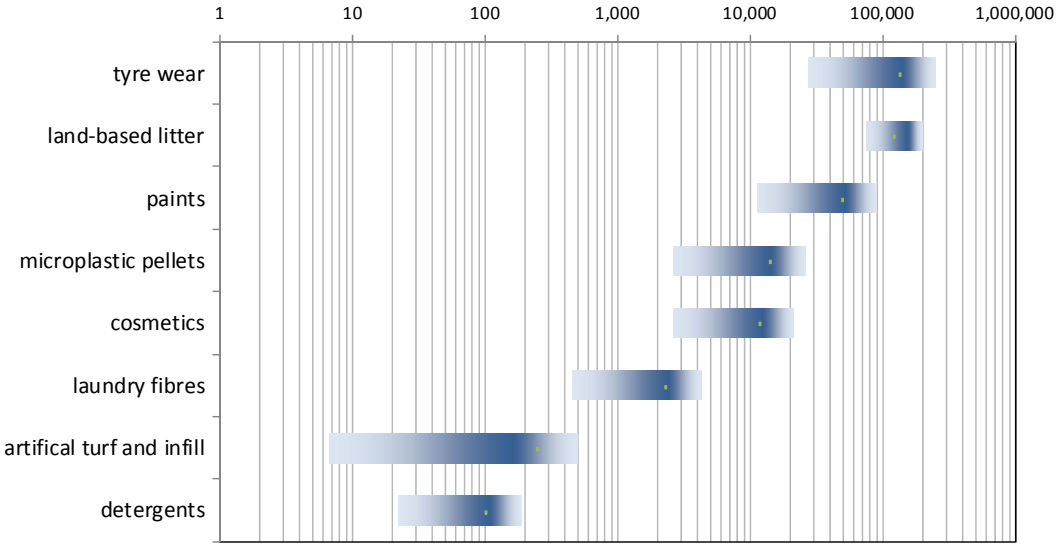


Figure 114: Estimated emissions of microplastics in OSPAR catchments (tonnes/year). The bars represent the uncertainty margins of the emission, white dots represent the midpoint.

Land-based sources included in the investigation were tyre wear, litter introduced from land-based sources, paint particles, microbeads in cosmetic and abrasive cleaning agents, laundry fibres, rubber infill and preproduction pellets. Considering the limitations and uncertainties as described above, the estimated amounts, as shown in Figure 14, were derived. Note that the estimated amounts have a high uncertainty band, which is caused by variations in local and regional habits, infrastructure, geography, weather, and by uncertainties about the actual loss of microplastics over the life-cycle and the distribution over environmental compartments. Despite these uncertainties, the approach gives an impression of the relative importance of different sources. The environmental impact not only depends on the mass of the microplastic release, but also of the size, shape and chemical composition of the particles.

After consumption, the particles potentially accumulate in the intestines, and could transfer to body fluids and could subsequently be taken up in cells. The actual extent of this process is uncertain but most particles are likely to be egested. Microplastics can have physical effects but also cause exposure to additives or interact with contaminants present in the ecosystem. The actual involvement of the different modes of action in the observed effects cannot be deducted from the knowledge currently available. It has been shown that microplastics interact with environmental contaminants and increase their exposure to marine organisms. However, the actual extent of this phenomenon in the marine environment is still unknown since many more parameters influence the health of marine ecosystems. The risk of microplastics consumption for human health at the current concentrations in seafood is low.

5 References

1. AISE, 2015, Pan-European consumer survey on sustainability and washing habits. Summary of findings 2014, 4 pages.
2. Avio, C.G., S. Gorbi, M. Milan, M. Benedetti, D. Fattorini, G. d'Errico, M. Pauletto, L. Bargelloni, and F. Regoli, Pollutants bioavailability and toxicological risk from microplastics to marine mussels. *Environmental Pollution*, **2015**. *198*(0): p. 211-222.
3. Baeyens, W., M. Leermakers, M. De Gieter, H.L. Nguyen, K. Parmentier, S. Panutrakul, and M. Elskens, Overview of trace metal contamination in the Scheldt estuary and effect of regulatory measures. *Hydrobiologia*, **2005**. *540*(1-3): p. 141-154.
4. Bakker, I.J.I., G.T. Klaver, S. Jansen, J. Joziasse, and E.S.v.d. Meulen, 2008, Variatie van de zwevend stof kwaliteit in het stroomgebied van de Maas, D.D. Cluster, 2008-U-R1081/A, 45 pages.
5. Besseling, E., E.M. Foekema, J.A. Van Franeker, M.F. Leopold, S. Kühn, E.L. Bravo Rebolledo, E. Heße, L. Mielke, J. Ijzer, P. Kamminga, and A.A. Koelmans, Microplastic in a macro filter feeder: Humpback whale *Megaptera novaeangliae. Marine Pollution Bulletin*, **2015**. *95*(1): p. 248-252.
6. Besseling, E., B. Wang, M. Lürling, and A.A. Koelmans, Nanoplastic Affects Growth of S. obliquus and Reproduction of D. magna. *Environmental Science & Technology*, **2014**. *48*(20): p. 12336-12343.
7. Besseling, E., A. Wegner, E.M. Foekema, M.J. van den Heuvel-Greve, and A.A. Koelmans, Effects of Microplastic on Fitness and PCB Bioaccumulation by the Lugworm Arenicola marina (L.). *Environmental Science & Technology*, **2013**. *47*(1): p. 593-600.
8. Bhattacharya, P., S. Lin, J.P. Turner, and P.C. Ke, Physical adsorption of charged plastic nanoparticles affects algal photosynthesis. *Journal of Physical Chemistry C*, **2010**. *114*(39): p. 16556-16561.
9. Blok, J., Environmental exposure of road border to zinc. *Science of the total environment*, **2005**. *248*: p. 173-190.
10. Boucher, J. and D. Friot, 2017, Primary microplastics in the oceans: A global evaluation of sources, International Union for Conservation of Nature and Natural Resources, 46 pages.
11. Brandsma, S.H., P. Nijssen, M.J.M. van Velzen, and H.A. Leslie, 2013, Microplastics in river suspended particulate matter and sewage treatment plants, 20 pages.
12. Brennecke, D., E.C. Ferreira, T.M.M. Costa, D. Appel, B.A.P. da Gama, and M. Lenz, Ingested microplastics (>100 µm) are translocated to organs of the tropical fiddler crab Uca rapax. *Marine Pollution Bulletin*, **2015**. *96*(1–2): p. 491-495.
13. Browne, M.A., P. Crump, S.J. Niven, E. Teuten, A. Tonkin, T. Galloway, and R. Thompson, Accumulation of Microplastic on Shorelines Woldwide: Sources and Sinks. *Environmental Science & Technology*, **2011**. *45*(21): p. 9175-9179.
14. Browne, M.A., A. Dissanayake, T.S. Galloway, D.M. Lowe, and R.C. Thompson, Ingested Microscopic Plastic Translocates to the Circulatory System of the Mussel, Mytilus edulis (L.). *Environmental Science & Technology*, **2008**. *42*(13): p. 5026-5031.
15. Browne, M.A., T.S. Galloway, and R.C. Thompson, Spatial Patterns of Plastic Debris along Estuarine Shorelines. *Environmental Science & Technology*, **2010**. *44*(9): p. 3404-3409.
16. Browne, Mark A., Stewart J. Niven, Tamara S. Galloway, Steve J. Rowland, and Richard C. Thompson, Microplastic Moves Pollutants and Additives to Worms, Reducing Functions Linked to Health and Biodiversity. *Current Biology*, **2013**. *23*(23): p. 2388-2392.
17. Carr, S.A., J. Liu, and A.G. Tesoro, Transport and fate of microplastic particles in wastewater treatment plants. *Water Research*, **2016**. *91*: p. 174-182.
18. Cedervall, T., L.-A. Hansson, M. Lard, and S. Linse, Food Chain Transport of Nanoparticles Affects Behaviour and Fat Metabolism in Fish. *PlosOne*, **2012**. *7*(2 e32254).
19. Claessens, M., S.D. Meester, L.V. Landuyt, K.D. Clerck, and C.R. Janssen, Occurrence and distribution of microplastics in marine sediments along the Belgian coast. *Marine Pollution Bulletin*, **2011**. *62*(10): p. 2199-2204.
20. Claessens, M., L. Van Cauwenberghe, M.B. Vandegehuchte, and C.R. Janssen, New techniques for the detection of microplastics in sediments and field collected organisms. *Marine Pollution Bulletin*, **2013**. *70*(1–2): p. 227-233.

21. Cole, G. and C. Sherrington, 2016, Study to Quantify Pellet Emissions in the UK. Report to FIDRA, 45 pages.
22. Cole, M., P. Lindeque, E. Fileman, C. Halsband, and T.S. Galloway, The Impact of Polystyrene Microplastics on Feeding, Function and Fecundity in the Marine Copepod Calanus helgolandicus. *Environmental Science & Technology*, **2015**. *49*(2): p. 1130-1137.
23. Cole, M., H. Webb, P.K. Lindeque, E.S. Fileman, C. Halsband, and T.S. Galloway, Isolation of microplastics in biota-rich seawater samples and marine organisms. *Sci. Rep.*, **2014**. *4*.
24. Cózar, A., E. Martí, C.M. Duarte, J. García-de-Lomas, E. van Sebille, T.J. Ballatore, V.M. Eguíluz, J.I. González-Gordillo, M.L. Pedrotti, F. Echevarría, R. Troublè, and X. Irigoien, The Arctic Ocean as a dead end for floating plastics in the North Atlantic branch of the Thermohaline Circulation. *Science Advances*, **2017**. *3*(4).
25. D. Hoornweg and P. Bhada-Tata, 2012, What a waste: A global review of solid waste management. *https://openknowledge.worldbank.org/handle/10986/17388*, W. bank, 161 pages.
26. Dannis, M., Rubber dust from the normal wear of tyres. *Rubber Chemistry and technology*, **1974**. *47*: p. 1011-1037.
27. De Sá, L.C., L.G. Luís, and L. Guilhermino, Effects of microplastics on juveniles of the common goby (Pomatoschistus microps): Confusion with prey, reduction of the predatory performance and efficiency, and possible influence of developmental conditions. *Environmental Pollution*, **2015**. *196*(0): p. 359-362.
28. Dekiff, J.H., D. Remy, J. Klasmeier, and E. Fries, Occurrence and spatial distribution of microplastics in sediments from Norderney. *Environmental Pollution*, **2014**. *186*(0): p. 248-256.
29. Devriese, L.I., M.D. van der Meulen, T. Maes, K. Bekaert, I. Paul-Pont, L. Frère, J. Robbens, and A.D. Vethaak, Microplastic contamination in brown shrimp (Crangon crangon, Linnaeus 1758) from coastal waters of the Southern North Sea and Channel area. *Marine Pollution Bulletin*, **2015**.
30. Dris, R., J. Gasperi, C. Mirande, C. Mandin, M. Guerrouache, V. Langlois, and B. Tassin, A first overview of textile fibers, including microplastics, in indoor and outdoor environments. *Environmental Pollution*, **2017**. *221*: p. 453-458.
31. Dris, R., J. Gasperi, M. Saad, C. Mirande, and B. Tassin, Synthetic fibers in atmospheric fallout: A source of microplastics in the environment? *Marine Pollution Bulletin*, **2015**.
32. Dubaish, F. and G. Liebezeit, Suspended Microplastics and Black Carbon Particles in the Jade System, Southern North Sea. *Water, Air, & Soil Pollution*, **2013**. *224*(2): p. 1-8.
33. EC, DIRECTIVE 2008/56/EC OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL of 17 June 2008 establishing a framework for community action in the field of marine environmental policy (Marine Strategy Framework Directive). *Official Journal of the European Union*, **2008**. *L164*: p. 19-40.
34. EC, 2017, Roadmap. Strategy on Plastics in a Circular Economy, 4 pages.
35. ECHA, 2012, Guidance for monomers and polymers. Guidance for the implementation of REACH. Version 2.0, ECHA-12-G-02-EN, 25 pages.
36. ECHA, November 2012, Guidance on information requirements and chemical safety assessment. Chapter R.11: PBT assessment, ECHA-12-G-24-EN, 99 pages.
37. EFSA Panel on Contaminants in Food [CONTAM], Presence of microplastics and nanoplastics in food, with particular focus on seafood. *EFSA Journal*, **2016**. *14*(6): p. 4501.
38. Ellen MacArthur Foundation, 2016, The new plastic economy. Rethinking the future of plastics, 118 pages.
39. Environment Agency, 2005, Environmental Risk Evaluation Report: 4-tert-Octylphenol, E.A.f.E.a. Wales, pages.
40. Eriksen, M., L.C.M. Lebreton, H.S. Carson, M. Thiel, C.J. Moore, J.C. Borerro, F. Galgani, P.G. Ryan, and J. Reisser, Plastic Pollution in the World's Oceans: More than 5 Trillion Plastic Pieces Weighing over 250,000 Tons Afloat at Sea. *PlosOne*, **2014**. *9*(12).
41. Essel, R., R.H. Ahrens, L. Engel, and M. Carus, 2014, Sources of microplastic relevant to marine protection, Nova Institute für Oekologie und Innovation GmbH, 31969, 45 pages.
42. Farrell, P. and K. Nelson, Trophic level transfer of microplastic: Mytilus edulis (L.) to Carcinus maenas (L.). *Environmental Pollution*, **2013**. *177*: p. 1-3.

43. Foekema, E.M., C. De Gruijter, M.T. Mergia, J.A. van Franeker, A.J. Murk, and A.A. Koelmans, Plastic in North Sea Fish. *Environmental Science & Technology*, **2013**. *47*(15): p. 8818-8824.
44. Frias, J.P.G.L., V. Otero, and P. Sobral, Evidence of microplastics in samples of zooplankton from Portuguese coastal waters. *Marine Environmental Research*, **2014**. *95*(0): p. 89-95.
45. Galgani, F., D. Fleet, J. van Franeker, S. Katsanevakis, T. Maes, J. Mouat, L. Oosterbaan, I. Poitou, G. hanke, R. Thompson, E. Amata, A. birkun, and C. Janssen, 2010, Marine Strategy Framework Directive. Task Group 10 Report. Marine Litter, JRC, EUR 24340, 57 pages.
46. Gallagher, A., A. Rees, R. Rowe, J. Stevens, and P. Wright, Microplastics in the Solent estuarine complex, UK: An initial assessment. *Marine Pollution Bulletin*, **2015**(0).
47. Gallagher, A., A. Rees, R. Rowe, J. Stevens, and P. Wright, Microplastics in the Solent estuarine complex, UK: An initial assessment. *Marine Pollution Bulletin*, **2016**. *102*(2): p. 243-249.
48. GESAMP, 2015, Sources, fate and effects of microplastics in the marine environment: a global assessment, I.F.U.-I.U.W.I.U.U.U.J.G.o.E. on and the Scientific Aspects of Marine Environmental Protection, GESAMP No. 90, 96 pages.
49. GESAMP, 2016, Sources, fate and effects of microplastics in the marine environment: Part two of a global assessment (Kershaw, P.J. and Rochman, C. eds.), IMO/FAO/UNESCO-IOC/UNIDO/WMO/IAEA/UN/UNEO/UNDP Joint Group of Experts in the Scientific Aspects of Marine Environmental Protection, GESAMP No. 93,, 222 pages.
50. Goldstein, M.C. and D.S. Goodwin, Gooseneck barnacles (Lepas spp.) ingest microplastic debris in the North Pacific Subtropical Gyre. *PeerJ*, **2013**. *1*: p. e184.
51. Goldstein, M.C., M. Rosenberg, and L. Cheng, Increased oceanic microplastic debris enhances oviposition in an endemic pelagic insect. *Biology Letters*, **2012**. *8*(5): p. 817-820.
52. Gouin, T., J. Avavalos, I. Brunning, K. Brzuska, I. de Graaf, J. Kaumanns, T. Koning, M. Meyberg, K. TRettinger, H. Schlatter, J. Thomas, R. van Welie, and T. Wolf, Use of micro-plastic beads in cosmetic products in Europe and their estimate emissions to the North Sea environment. *SOFW Journal International Journal for Applied Science*, **2015**. *141*(3): p. 40-46.
53. Gouin, T., N. Roche, R. Lohmann, and G. Hodges, A Thermodynamic Approach for Assessing the Environmental Exposure of Chemicals Absorbed to Microplastic. *Environmental Science & Technology*, **2011**. *45*(4): p. 1466-1472.
54. Green, D.S., B. Boots, J. Sigwart, S. Jiang, and C. Rocha, Effects of conventional and biodegradable microplastics on a marine ecosystem engineer (Arenicola marina) and sediment nutrient cycling. *Environmental Pollution*, **2016**. *208*: p. 426-434.
55. Gustavsson, M., Icke-avgasrelaterade partiklar i vägmiljön.VTI meddelande 910. **2001**.
56. Gutow, L., A. Eckerlebe, L. Giménez, and R. Saborowski, Experimental Evaluation of Seaweeds as a Vector for Microplastics into Marine Food Webs. *Environmental Science and Technology*, **2016**. *50*(2): p. 915-923.
57. Hämer, J., L. Gutow, A. Köhler, and R. Saborowski, Fate of microplastics in the marine isopod idotea emarginata. *Environmental Science and Technology*, **2014**. *48*(22): p. 13451-13458.
58. HELCOM, 2014, BASE project 2012-2014: Preliminary study on synthetic microfibers and particles at a municipal waste water treatment plant, 17 pages.
59. Hillenbrand, T., D. Toussaint, E. Böhm, S. Fuchs, U. Scherer, A. Rudolphi, M. Hoffmann, J. Kreissig, and C. Kotz, 2005, Einträge von Kupfer, Zink und Blei in Gewässer und Böden - Analyse der Emissionspfade und möglicher Emissionsminderungsmaßnahmen, 329 pages.
60. Hohn, D., Moby-Duck: True Story of 28,800 Bath Toys Lost at Sea & of the Beachcombers, Oceanographers, Environmentalists & Fools Including the Author Who Went in Search of Them. **2011**.
61. ICC, 2015, International Ocean Cleanup. Trash free seas. Every piece, every person., O. Conservancy, 14 pages.
62. Jambeck, J.R., R. Geyer, C. Wilcox, T.R. Siegler, M. Perryman, A. Andrady, R. Narayan, and K. Lavender Law, Plastic waste inputs from land into the ocean. *Science*, **2015**. *347*(6223): p. 768-771.
63. Kaposi, K.L., B. Mos, B.P. Kelaher, and S.A. Dworjanyn, Ingestion of Microplastic Has Limited Impact on a Marine Larva. *Environmental Science & Technology*, **2014**. *48*(3): p. 1638-1645.

64. Karlsson, T.M., 2014, Can microlitter in sediment and biota be quantified. Method development and analysis of microlitter in field-collected biota and sediment. MSc thesis University of Gothenburg., 39 pages.
65. Klein, J., G. Geilenkirchen, J. Hulskotte, N. Ligterink, P. Fortuin, and H. Molnar-in 't Veld, 2014, Methods for calculating the emissions of transport in the Netherlands, plus Excel-file, Task Force on transportation of the Dutch Pollutant Release and Transfer Register, 96 pages.
66. Klein, J., H. Molnar-in 't Veld, G. Geilenkirchen, J. Hulskotte, N. Ligterink, G. Kadijk, and R. de Boer, 2015, Methods for calculating the emissions of transport in the Netherlands, Task Force on transportation of the Dutch Pollutant Release and Transfer Register, 74 pages.
67. Klein, S., E. Worch, and T.P. Knepper, Occurrence and Spatial Distribution of Microplastics in River Shore Sediments of the Rhine-Main Area in Germany. *Environmental Science & Technology*, **2015**. *49*(10): p. 6070-6076.
68. Koelmans, A.A., E. Besseling, and E.M. Foekema, Leaching of plastic additives to marine organisms. *Environmental Pollution*, **2014**. *187*: p. 49-54.
69. Koelmans, A.A., E. Besseling, A. Wegner, and E.M. Foekema, Plastic as a Carrier of POPs to Aquatic Organisms: A Model Analysis. *Environmental Science & Technology*, **2013**. *47*(14): p. 7812-7820.
70. Kreider, M.L., J.M. Panko, B.L. McAtee, L.I. Sweet, and B.L. Finley, Physical and chemical characterization of tyre-related particles: Comparison of particles generated using different methodologies. *Science of the Total Environment*, **2010**. *408*: p. 652-659.
71. Lassen, C., S.F. Hansen, K. Magnusson, F. Norén, N.I.B. Hartmann, P.R. Jensen, T.G. Nielsen, and A. Brinch, 2015, Microplastics - Occurrence, effects and sources of releases to the environment in Denmark, D. EPA, 205 pages.
72. Lebreton, L., J. van der Zwer, J.-W. Damsteeg, B. Slat, A. Andrady, and J. Reisser, River plastic emissions to the world's oceans. *Nature Communications*, **2017**. *8*(15611): p. 1-10.
73. Lee, K.-W., W.J. Shim, O.Y. Kwon, and J.-H. Kang, Size-Dependent Effects of Micro Polystyrene Particles in the Marine Copepod Tigriopus japonicus. *Environmental Science & Technology*, **2013**. *47*(19): p. 11278-11283.
74. LEITAT, 2016, Mitigation of microplastics impact caused by textile washing processes - LCA report (Public). Associated action C2, LIFE13 ENV/IT/001069, 32 pages.
75. LeMaitre, O., M. Süssner, and C. Zarak, 1998, Evaluation of tyre wear performance. SAE technical paper series 980256, I.c.a. Exposition, pages.
76. Leslie, H., 2012, Microplastic in Noordzee zwevend stof en cosmetica, IVM, W-12/01, 19 pages.
77. Leslie, H., M. Moester, M. de Kreuk, and D. Vethaak, Verkennende studie naar lozing van microplastics door rwzi's. *H_2O*, **2012**. *14/15*: p. 45-47.
78. Leslie, H., M.J.M. van Velzen, and A.D. Vethaak, 2013, Microplastic survey of the Dutch environment. Novel data set of microplastics in North Sea sediments, treated wastewater effluents and marine biota., IVM Institute for Environmental Studies, R-13/11, 30 pages.
79. Leslie, H.A., S.H. Brandsma, M.J.M. van Velzen, and A.D. Vethaak, Microplastics en route: Field measurements in the Dutch river delta and Amsterdam canals, wastewater treatment plants, North Sea sediments and biota. *Environment International*, **2017**. *101*: p. 133-142.
80. Liebezeit, G. and F. Dubaish, Microplastics in Beaches of the East Frisian Islands Spiekeroog and Kachelotplate. *Bulletin of Environmental Contamination and Toxicology*, **2012**. *89*(1): p. 213-217.
81. Lu, Y., Y. Zhang, Y. Deng, W. Jiang, Y. Zhao, J. Geng, L. Ding, and H. Ren, Uptake and Accumulation of Polystyrene Microplastics in Zebrafish (Danio rerio) and Toxic Effects in Liver. *Environmental Science and Technology*, **2016**. *50*(7): p. 4054-4060.
82. Luís, L.G., P. Ferreira, E. Fonte, M. Oliveira, and L. Guilhermino, Does the presence of microplastics influence the acute toxicity of chromium(VI) to early juveniles of the common goby (Pomatoschistus microps)? A study with juveniles from two wild estuarine populations. *Aquatic Toxicology*, **2015**. *164*: p. 163-174.
83. Lusher, A.L., A. Burke, I. O'Connor, and R. Officer, Microplastic pollution in the Northeast Atlantic Ocean: Validated and opportunistic sampling. *Marine Pollution Bulletin*, **2014**. *88*(1–2): p. 325-333.

84. Lusher, A.L., G. Hernandez-Milian, J. O'Brien, S. Berrow, I. O'Connor, and R. Officer, Microplastic and macroplastic ingestion by a deep diving, oceanic cetacean: The True's beaked whale Mesoplodon mirus. *Environmental Pollution*, **2015**. *199*(0): p. 185-191.
85. Lusher, A.L., M. McHugh, and R.C. Thompson, Occurrence of microplastics in the gastrointestinal tract of pelagic and demersal fish from the English Channel. *Marine Pollution Bulletin*, **2013**. *67*(1–2): p. 94-99.
86. Magnusson, K., K. Eliasson, A. Fråne, K. Haikonen, J. Hultén, M. Olshammar, J. Stadmark, and A. Voisin, 2016, Swedish sources and pathways for microplastics to the marine environment. A review of existing data. Revised version March 2017, IVL Svenska Miljöinstitutet, C 183, 89 pages.
87. Mani, T., A. Hauk, U. Walter, and P. Burkhardt-Holm, Microplastics profile along the Rhine River. *Scientific Reports*, **2015**. *5*: p. 17988.
88. Martins, J. and P. Sobral, Plastic marine debris on the Portuguese coastline: A matter of size? *Marine Pollution Bulletin*, **2011**. *62*(12): p. 2649-2653.
89. Mermaids, 2016, Report on the influence of commercial laundry agents and washing conditions on microplastics release reduction by use of commercial detergents, additives and changing laundering conditions. Action A2, LIFE13 ENV/IT/001069 project: Mitigation of microplastics impact caused by textile washing processes, 170 pages.
90. Minifibers Inc. *Choosing The Proper Short Cut Fiber for Your Nonwoven Web* accessed: 12-4-2017], Available from: *http://www.minifibers.com/documents/Choosing-the-Proper-Short-Cut-Fiber.pdf*.
91. Mintenig, S., I. Int-Veen, M. Löder, and G. Gerdts, 2014, Mikroplastik in ausgewählten Kläranlagen des Oldenburgisch- Ostfriesischen Wasserverbandes (OOWV) in Niedersachsen. Probenanalyse mittels Mikro-FTIR Spektroskopie, 50 pages.
92. Napper, I.E., A. Bakir, S.J. Rowland, and R.C. Thompson, Characterisation, quantity and sorptive properties of microplastics extracted from cosmetics. *Marine Pollution Bulletin*, **2015**. *99*(1–2): p. 178-185.
93. Napper, I.E. and R.C. Thompson, Release of synthetic microplastic plastic fibres from domestic washing machines: Effects of fabric type and washing conditions. *Marine Pollution Bulletin*, **2016**. *112*(1–2): p. 39-45.
94. Nerland, I.I., C. Halsband, I. Allan, and K.V. Thomas, 2014, Microplastics in the marine environment: Occurence, distribution and effects, NIVA Norwegian Institute for Water Reseach, 71 pages.
95. NOAA and UNEP. *The Honololu Strategy, a global framework for prevention and management of marine debris* accessed, Available from: *https://5imdc.files.wordpress.com/2011/03/honolulustrategy.pdf*.
96. Nobre, C.R., M.F.M. Santana, A. Maluf, F.S. Cortez, A. Cesar, C.D.S. Pereira, and A. Turra, Assessment of microplastic toxicity to embryonic development of the sea urchin Lytechinus variegatus (Echinodermata: Echinoidea). *Marine Pollution Bulletin*, **2015**. *92*(1–2): p. 99-104.
97. Norén, F., 2007, Small plastic particles in coastal Swedish waters, KIMO Sweden, 11 pages.
98. Norén, F., 2011, Survey of microscopic anthropogenic particles in Skagerrak. Pilot study October-November 2010, S. Institute of Marine Research, 22 pages.
99. OECD, 1996, OECD Guideline for testing of chemicals. Technical Guideline 118: Determination of the Number-Average Molecular Weight and the Molecular Weight Distribution of Polymers using Gel Permeation Chromatography., OECD, 9 pages.
100. OECD, Emission scenario document on coating industry (paints, lacquers and varnishes). *OECD Health and Safety Publications, Series on Emission Scenario Documents*, **2009**. *22*: p. 201.
101. Oliveira, M., A. Ribeiro, K. Hylland, and L. Guilhermino, Single and combined effects of microplastics and pyrene on juveniles (0+ group) of the common goby Pomatoschistus microps (Teleostei, Gobiidae). *Ecological Indicators*, **2013**. *34*(0): p. 641-647.
102. OSPAR, 2000, Quality Status Report 2000, 108 pages.
103. OSPAR. *The North-East Atlantic Environment Strategy. Strategy of the OSPAR Commission for the Protection of the Marine Environment of the North-East Atlantic 2010–2020, (OSPAR Agreement 2010-3)* accessed, Available from: *http://www.ospar.org/site/assets/files/1200/strategy.pdf*.
104. OSPAR. *Regional Action Plan for Prevention and Management of Marine Litter in the North-East Atlantic* accessed, Available from: *www.ospar.org/documents?d=32986*.

105. Pakula, C. and R. Stamminger, Electricity and water consumption for laundry washing by washing machine worldwide. *Energy Efficiency*, **2010**. *3*(4): p. 365-382
106. Pant, P. and R.M. Harrison, Estimation of the contribution of road traffic emissions to particulate matter concentrations from field measurements: A review. *Atmospheric Environment*, **2013**. *77*: p. 78-97.
107. Pedà, C., L. Caccamo, M.C. Fossi, F. Gai, F. Andaloro, L. Genovese, A. Perdichizzi, T. Romeo, and G. Maricchiolo, Intestinal alterations in European sea bass Dicentrarchus labrax (Linnaeus, 1758) exposed to microplastics: Preliminary results. *Environmental Pollution*, **2016**. *212*: p. 251-256.
108. Pham, C.K., E. Ramirez-Llodra, C.H.S. Alt, T. Amaro, M. Bergmann, M. Canals, J.B. Company, J. Davies, G. Duineveld, F. Galgani, K.L. Howell, V.A.I. Huvenne, E. Isidro, D.O.B. Jomes, G. Lastras, T. Morato, J.N. Gomes-Pereira, A. Purser, H. Stewart, I. Tojeira, X. Tubau, D. van Rooij, and P.A. Tyler, Marine Litter Distribution and Density in European Seas, from the Shelves to Deep Basins. *Plos One*, **2014**. *9*(4): p. e95839.
109. Pilz, H., B. Brandt, and R. Fehringer, 2010, The impact of plastics on life cycle energy consumption and greenhouse gas emissions in Europe. Summary report, Denkstatt GmbH, 45 pages.
110. Plastics Europe, 2013, Plastics - The facts 2013. An analysis of European latest plastics production, demand and waste data, 40 pages.
111. Plastics Europe, 2015, Plastics - The facts 2014-2015. An analysis of European plastics production, demand and waste data, Plastics Europe Association of Plastic Manufacturers, 34 pages.
112. Plastics Europe, 2016, Plastics - the Facts 2016. An analysis of European plastics production, demand and waste data, 38 pages.
113. Rehse, S., W. Kloas, and C. Zarfl, Short-term exposure with high concentrations of pristine microplastic particles leads to immobilisation of Daphnia magna. *Chemosphere*, **2016**. *153*: p. 91-99.
114. Rijkswaterstaat, 2014, Emissieschattingen diffuse bronnen. Emissieregistratie. Effluenten RWZI's, regenwaterriolen, niet aangesloten riolen, overstorten en IBA's, 29 pages.
115. Rijkswaterstaat, 2014, Emissieschattingen difusse bronnen. Emissieregistratie. Huishoudelijk afvalwater scheepvaart, 10 pages.
116. Rochman, C.M., E. Hoh, T. Kurobe, and S.J. Teh, Ingested plastic transfers hazardous chemicals to fish and induces hepatic stress. *Scientific Reports*, **2013**. *3*.
117. Rummel, C.D., M.G.J. Löder, N.F. Fricke, T. Lang, E.M. Griebeler, M. Janke, and G. Gerdts, Plastic ingestion by pelagic and demersal fish from the North Sea and Baltic Sea. *Marine Pollution Bulletin*, **2016**. *102*(1): p. 134-141.
118. Russia, 2013, Informal Document GRPE-65-20. Particulate matter emissions by tyres. 65th GRPE, 15-18 January 2013, agenda item 16, 4 pages.
119. Salvador Cesa, F., A. Turra, and J. Baruque-Ramos, Synthetic fibers as microplastics in the marine environment: A review from textile perspective with a focus on domestic washings. *Science of the Total Environment*, **2017**. *598*: p. 1116-1129.
120. Sanchez, W., C. Bender, and J.-M. Porcher, Wild gudgeons (Gobio gobio) from French rivers are contaminated by microplastics: Preliminary study and first evidence. *Environmental Research*, **2014**. *128*: p. 98-100.
121. Seas at Risk, 2016, Microplastic Pellet Loss: Preventing Pollution Through The EU Plastics Strategy. A Briefing Note by the European Coalition to End Plastic Pellet Loss, 25 pages.
122. Setälä, O., V. Fleming-Lehtinen, and M. Lehtiniemi, Ingestion and transfer of microplastics in the planktonic food web. *Environmental Pollution*, **2014**. *185*: p. 77-83.
123. Sherrington, C., C. Darrah, S. Hann, G. Cole, and M. Corbin, 2016, Study to support the development of measures to combat a range of marine litter sources. Report for European Commission DG Environment, Eunomia, 432 pages.
124. Shui, S. and A. Plastina, 2013, World apparel fiber consumption survey, F.F.a.A.o.o.t.U.N.a.I.C.A. Committee, 27 pages.
125. Sjollema, S.B., P. Redondo-Hasselerharm, H.A. Leslie, M.H.S. Kraak, and A.D. Vethaak, Do plastic particles affect microalgal photosynthesis and growth? *Aquatic Toxicology*, **2016**. *170*: p. 259-261.

126. Sundt, P., P.E. Schulze, and F. Syversen, 2014, Sources of microplastics-pollution to the marine environment, Norwegian Environment Agency, 86 pages.
127. Sussarellu, R., M. Suquet, Y. Thomas, C. Lambert, C. Fabioux, M.E.J. Pernet, N.L. Goïc, V. Quillien, C. Mingant, Y. Epelboin, C. Corporeau, J. Guyomarch, J. Robbens, I. Paul-Pont, P. Soudant, and A. Huvet, Oyster reproduction is affected by exposure to polystyrene microplastics. *Proceedings of the National Academy of Sciences of the United States of America*, **2016**. *113*(9): p. 2430-2435.
128. Syberg, K., F.R. Khan, H. Selck, A. Palmqvist, G.T. Banta, J. Daley, L. Sano, and M.B. Duhaime, Microplastics: Addressing ecological risk through lessons learned. *Environmental Toxicology and Chemistry*, **2015**. *34*(5): p. 945-953.
129. Talvitie, J., A. Mikola, O. Setälä, M. Heinonen, and A. Koistinen, How well is microlitter purified from wastewater? – A detailed study on the stepwise removal of microlitter in a tertiary level wastewater treatment plant. *Water Research*, **2017**. *109*: p. 164-172.
130. Tanaka, K., H. Takada, R. Yamashita, K. Mizukawa, M.A. Fukuwaka, and Y. Watanuki, Accumulation of plastic-derived chemicals in tissues of seabirds ingesting marine plastics. *Marine Pollution Bulletin*, **2013**. *69*(1-2): p. 219-222.
131. ten Broeke, H., J. Hulskotte, and H. Denier van der Gon, 2008, Emission estimates for diffuse sources. Netherlands Emission inventory. Road traffic tyre wear, TNO, 27 pages.
132. Teuten, E.L., J.M. Saquing, D.R.U. Knappe, M.A. Barlaz, S. Jonsson, A. Björn, S.J. Rowland, R.C. Thompson, T.S. Galloway, R. Yamashita, D. Ochi, Y. Watanuki, C. Moore, P.H. Viet, T.S. Tana, M. Prudente, R. Boonyatumanond, M.P. Zakaria, K. Akkhavong, Y. Ogata, H. Hirai, S. Iwasa, K. Mizukawa, Y. Hagino, A. Imamura, M. Saha, and H. Takada, Transport and release of chemicals from plastics to the environment and to wildlife. *Philosophical Transactions of the Royal Society B: Biological Sciences*, **2009**. *364*(1526): p. 2027-2045.
133. UNECE. *Globally Harmonised System of Classification and Labelling of Chemicals* accessed: 6-11-2014], Available from: *www.unece.org/trans/danger/publi/ghs/ghs_rev05/05files_e.html*.
134. UNEP, 2005, Marine litter. An analytical overview, 58 pages.
135. UNEP, 2009, Marine litter: A global challenge, United Nations Environment Programme, 234 pages.
136. UNEP, 2014, Valuing plastics: The business case for measuring, managing and disclosing plastic use in the consumer goods industry, United Nations Environment Programme, 116 pages.
137. UNEP, 2016, Marine plastic debris and microplastics – Global lessons and research to inspire action and guide policy change., 274 pages.
138. United Nations. *Probabilistic Population Projections based on the World Population Prospects: The 2015 Revision. Population Division, DESA. http://esa.un.org/unpd/ppp/2* accessed: 18-4-2017].
139. University of Minnesota, Plastics Manufacturer Reduces Waste through Good Housekeeping and Recycling, *http://infohouse.p2ric.org/ref/17/16189.htm*, Accessed 21-4-2017. **1992**.
140. Urgert, W., 2015, Microplastics in the rivers Meuse and Rhine. Developing guidance for a possible future monitoring program. MSc Thesis Open University., 106 pages.
141. Van Cauwenberghe, L., M. Claessens, M.B. Vandegehuchte, and C.R. Janssen, Microplastics are taken up by mussels (Mytilus edulis) and lugworms (Arenicola marina) living in natural habitats. *Environmental Pollution*, **2015**. *199*(0): p. 10-17.
142. Van Cauwenberghe, L., M. Claessens, M.B. Vandegehuchte, and C.R. Janssen, Microplastics are taken up by mussels (Mytilus edulis) and lugworms (Arenicola marina) living in natural habitats. *Environmental Pollution*, **2015**. *199*: p. 10-17.
143. Van Cauwenberghe, L., M. Claessens, M.B. Vandegehuchte, J. Mees, and C.R. Janssen, Assessment of marine debris on the Belgian Continental Shelf. *Marine Pollution Bulletin*, **2013**. *73*(1): p. 161-169.
144. Van Cauwenberghe, L., L. Devriese, F. Galgani, J. Robbens, and C.R. Janssen, Microplastics in sediments: A review of techniques, occurrence and effects. *Marine Environmental Research*, **2015**.
145. Van Cauwenberghe, L. and C.R. Janssen, Microplastics in bivalves cultured for human consumption. *Environmental Pollution*, **2014**. *193*: p. 65-70.
146. Van Cauwenberghe, L., W. Van Echelpoel, K. De Gussem, G. De Gueldre, M. Vandegehuchte, and C. Janssen, Unraveling the sources of marine microplastics: your daily contribution?,in: Mees, J. et al. (Ed.) (2015). Book of abstracts - VLIZ Young Scientists' Day. *VLIZ Special Publication*, **2015**. *71*: p. 136.

147. Van Cauwenberghe, L., A. Vanreusel, J. Mees, and C.R. Janssen, Microplastic pollution in deep-sea sediments. *Environmental Pollution*, **2013**. *182*: p. 495-499.
148. van der Wal, M., M. van der Meulen, G. Tweehuizen, M. Peterlin, A. Palatinus, M. Kovač Viršek, L. Coscia, and A. Kržan, 2015, SFRA0025: Identification and Assessment of Riverine Input of (Marine) Litter. Final Report for the European Commission DG Environment under Framework Contract No ENV.D.2/FRA/2012/0025, Eunomia, 208 pages.
149. Van Sebille, E., C. Wilcox, L. Lebreton, N. Maximenko, B.D. Hardesty, J.A. Van Franeker, M. Eriksen, D. Siegel, F. Galgani, and K.L. Law, A global inventory of small floating plastic debris. *Environmental Research Letters*, **2015**. *10*(12).
150. Vandermeersch, G., L. Van Cauwenberghe, C.R. Janssen, A. Marques, K. Granby, G. Fait, M.J.J. Kotterman, J. Diogène, K. Bekaert, J. Robbens, and L. Devriese, A critical view on microplastic quantification in aquatic organisms. *Environmental Research*, **2015**.
151. Verschoor, A., 2015, Towards a definition of microplastics. Considerations for the specifiction of physico-chemical properties, RIVM, 2015-0116, 27 pages.
152. Verschoor, A.J., L.R.M. de Poorter, E. de Valk, R. Dröge, and J. Kuenen, 2016, Emission of microplastics and potential mitigation measures. Abrasive cleaning agents, paints and tyre wear, RIVM/TNO, 2016-0026, 76 pages.
153. von Moos, N., P. Burkhardt-Holm, and A. Köhler, Uptake and Effects of Microplastics on Cells and Tissue of the Blue Mussel Mytilus edulis L. after an Experimental Exposure. *Environmental Science & Technology*, **2012**. *46*(20): p. 11327-11335.
154. Wardrop, P., J. Shimeta, D. Nugegoda, P.D. Morrison, A. Miranda, M. Tang, and B.O. Clarke, Chemical Pollutants Sorbed to Ingested Microbeads from Personal Care Products Accumulate in Fish. *Environmental Science and Technology*, **2016**. *50*(7): p. 4037-4044.
155. Wegner, A., E. Besseling, E.M. Foekema, P. Kamermans, and A.A. Koelmans, Effects of nanopolystyrene on the feeding behavior of the blue mussel (Mytilus edulis L.). *Environmental Toxicology and Chemistry*, **2012**. *31*(11): p. 2490-2497.
156. Weijer, A., J. Knol, and U. Hofstra, 2017, Verspreiding van infill en indicatieve massabalans. Rapport i.o.v. BSCN i.s.m. gemeenten Rotterdam, Utrecht, Amsterdam en Den Haag, I. SWECO, 48 pages.
157. Werner, S., A. Budziak, J. van Franeker, F. Galgani, G. Hanke, T. Maes, M. Matiddi, P. Nilsson, L. Oosterbaan, E. Priestland, R. Thompson, J. Veiga, and T. Vlachogianni, 2016, Harm caused by Marine Litter. MSFD GES TG Marine Litter - Thematic Report,; JRC Technical report; EUR 28317 EN; doi:10.2788/690366, JRC, 92 pages.
158. Woodall, L.C., A. Sanchez-Vidal, M. Canals, G.L.J. Paterson, R. Coppock, V. Sleight, A.M. Calafat, A.D. Rogers, B.E. Narayanaswamy, and R.C. Thompson, The deep sea is a major sink for microplastic debris. *Royal Society Open Science*, **2014**. *1*(140317).
159. Wright, S.L., D. Rowe, R.C. Thompson, and T.S. Galloway, Microplastic ingestion decreases energy reserves in marine worms. *Current Biology*, **2013**. *23*(23): p. R1031-R1033.

Appendices

Appendix 1 Further options for specification of the term microplastics

The provisions for polymers and monomers under REACH are described in a separate guidance document [35]. This document defines a polymer as a substance consisting of molecules characterised by the sequence of one or more types of monomer unit. Such molecules must be distributed over a range of molecular weights. Differences in the molecular weight are primarily attributable to differences in the number of monomer units.

In accordance with REACH (Article 3(5)), a polymer is defined as a substance meeting the following criteria:

a) Over 50 percent of the weight for that substance consists of polymer molecules (see definition below); and,
b) The amount of polymer molecules presenting the same molecular weight must be less than 50 weight percent of the substance.

The definition is in line with definitions recommended earlier by the OECD [99].

Physical state of microplastics

The Globally Harmonized System for Classification and Labelling of substances and mixtures [133] defines a solid as *a substance or mixture which does not meet the definitions of liquid or gas*.[2] It employs the melting temperature and vapour pressure as threshold values. The complete GHS definition is shown in the textbox Definitions of solid, liquid and gas according to the Globally Harmonized System for Classification and Labelling of substances and mixtures (GHS).

A solid is a substance or mixture which does not meet the definitions of liquid or gas

- Liquid means a substance or mixture which at 50 °C has a vapor pressure of not more than 300 kPa (3 bar), which is not completely gaseous at 20 °C and at a standard pressure of 101.3 kPa, and which has a melting point or initial melting point of 20 °C or less at a standard pressure of 101.3 kPa.
- Gas means a substance which (i) at 50 °C has a vapour pressure greater than 300 kPa; or (ii) is completely gaseous at 20 °C at a standard pressure

[2] GHS is an internationally agreed-upon system, created by the United Nations. It is designed to replace the various classification and labeling standards used in different countries by using consistent criteria for classification and labeling on a global level.

http://ec.europa.eu/enterprise/sectors/chemicals/files/clp-ghs-glossary_en.pdf

Solubility

The solubility of polymers depends on their polarity, molecular weight, branching, crosslinking degree, and crystallinity. Polar macromolecules like polyethyleneglycol (PEG), polyacrylic acid, polyacrylamide and polyvinyl alcohol (PVA) among others, are soluble in water. Conversely, nonpolar polymers or polymers showing a low polarity such as polystyrene, polymethyl methacrylate, polyvinyl chloride, and polyisobutilene are poorly soluble in water.

In the implementation of REACH, a substance is considered poorly water soluble when it has a water solubility **below 1mg/L** at 20 °C or below the detection limit of the analytical method of the tested substance [36]. It is proposed by the authors to adopt this interpretation as presented in the REACH guidance document on information requirements and chemical safety assessment.

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Appendix 2 Estimation of population living in OSPAR catchment.

Reference year is 2000 unless stated otherwise.

	Population in OSPAR catchment (x million) [102]	Total national population in reference year (x million) [138]	Percentage population in OSPAR catchment	
			Calculated	Used in this report
Region I				
Faroe Islands	0.04	0.046	86%	100%[1]
Finland (1997)	0.009	5,140	0.2%	0.2%
Greenland (1995)	0.004	0.055	7.2%	7.2%
Iceland (1996)	0.27	0.27	100%	100%
Norway (1996)	1.1	4.4	25%	25%[3]
Russia (1989)	1.2	146	0.8%	0.8%
Region II				
Belgium	10.1	10.2	99%	100%[1]
Czech Republic (1996)	10.3	10.3	100%	100%
Denmark	2.2	5.3	41%	41%
France	25.3	59.4	43%	43%[3]
Germany	72.5	81.9	89%	89%
Liechtenstein (1996)	0.031	0.03	99%	100%[1]
Luxembourg (1996)	0.4	0.4	97%	100%[1]
Netherlands	15.6	15.9	98%	100%[1]
Norway	3.3	4.5	73%	75%[3]
Sweden	2.4	8.9	27%	27%
Switzerland	5.7	7.2	80%	80%
UK	36.4	58.8	62%	62%[3]
Region III				
Ireland	3.6	3.8	94%	100%[1]
UK	22.3	5.9	38%	38%[3]
Region IV	47.2			
France	16.7	59.4	28%	28%[3]
Portugal	9.5	10.3	92%	100%[1]
Spain	21.1	40.7	52%	52%
Region V				
Azores – Portugal	0.24	10.3	2%	2%[3]

[1] Percentage should be 100% because the country discharges only to the OSPAR Maritime Area. Minor deviation are probably caused by a potential mismatch of the reference year. If reference year was not given it was taken as default reference year unless otherwise stated.

[2] Sum of Region I and II for Norway should be 100%, therefore percentage of Region 2 is adjusted from 773% to 75%.

[3] Some countries discharge in two OSPAR regions. Norway: Region I + II: 25 + 75 = 100%; France: Region II + IV = 43% + 28% = 71%: UK: Region II + II = 62 + 38% = 100%; Portugal: Region IV + V = 100 + 2 ==102%. Portugal is adjusted to 100%.

Appendix 3 Estimation of population living in OSPAR coastal zones

Reference year is 2000 unless stated otherwise.

Country	Total national Population (2010) [138]	Coastal population (2010) [62]	Calculated % coastal population per country	% population in OSPAR catchment (See appendix 1)	Calculated coastal population in OSPAR catchment (%)
Belgium	10,929,978	4,747,957	43%	100%	43%
Denmark	5,550,959	5,376,386	97%	41%	40%
Finland	5,367,693	2,927,674	55%	0.20%	0.11%
France	62,961,136	17,287,280	27%	71%	19%
Germany	80,435,307	8,837,035	11%	89%	10%
Iceland	318,042	292,708	92%	100%	92%
Ireland	4,617,334	3,749,576	81%	100%	81%
Luxembourg	567,110	0	0%	100%	0%
Norway	4,891,251	4,131,679	84%	73%	61%
Portugal	10,584,837	8,507,951	80%	100%	80%
Spain	46,601,492	22,771,488	49%	52%	25%
Sweden	9,382,297	6,202,234	66%	27%	18%
Switzerland	8,298,660	0	0%	80%	0%
The Netherlands	16,631,571	8,971,770	54%	100%	54%
UK	62,716,684	43,258,889	69%	100%	69%
Greenland	57,068	57,068	100%	7%	7.00%
Russian Federation	143,158,099	10,812,537	8%	0.80%	0.80%
Liechtenstein		0	0%	100%	0.00%
Czech Republic		0	0%	63%	0.00%
Total					

Appendix 4 Production and use of plastics

The world production of plastic materials in 2015 was estimated to be 322 Mtonne. This figure includes thermosets and polyurethane, and other plastics (thermosets, coatings, adhesives, sealants and non PP-fibres. Europe (EU27 + CH+NO) is responsible for 18% of the world production: 58 Mtonnes [111].

The plastics industry includes polymer producers and plastic converters. Plastic recyclers are a separate group of industry.

a) There are more than 50 large plastics producing companies in Europe. Most major plastics producing companies by production volume are represented by the European plastics producers association PlasticsEurope. A member list is given on their website (*www.plasticseurope.org*). Plastics producers manufacture their material mainly in the form of pellets and to smaller extend as powder or even other shapes/states. It is used as raw material for further processing steps by converters.
b) The plastics converters (also called processors) buy in raw material in granular or powder or liquid etc., subject it to a process involving pressure, heat and/or chemistry and apply design expertise to manufacture their products. They often undertake additional finishing operations such as printing and assembly work to add further value to their activities. The European Association of Plastic Converters (EuPC) now totals about 51 plastics converting national and European industry associations, it represents approximately 50,000 companies. A list of associations is given at their website (*www.plasticsconverters.eu*)

Polymer producers manufacture polymers in the form of powders, or pellets. There are approximately 50 polymer producing companies in Europe, represented by Plastics Europe, an Association of Plastic manufacturers. A member list is given on their website (*www.plasticseurope.org*). European polymer producers manufacture approximately 57 Mtonnes of polymers in the form of powders or pellets. These pellets are generally considered as primary microplastics. This is raw material that is used for further processing.

The plastic converters (also called processors) buy in raw material in granular or powder form, subject it to a process involving pressure, heat and/or chemistry and apply design expertise to manufacture their products. They often undertake additional finishing operations such as printing and assembly work to add further value to their activities. The Association of Plastic Converters (EuPC) now totals about 51 European Plastics Converting national and European industry associations, it represents approximately 50,000 companies. A list of associations is given at their website (www.plasticsconverters.eu)

Which part of the total European polymer production is converted to engineered microplastics for use in products is unknown.

Appendix 5 Estimated pellet loss in the OSPAR Maritime Area

Country	number of companies [121]	Plastic demand Mton [112]	Calculated Pellet loss per company [tonnes/year]	Calculated Total pellet loss per country [tonnes/year]	Calculated Total pellet loss in OSPAR catchments
Contracting parties					
Belgium	936	2.0	0.22 - 2.2	204 - 2041	204 - 2041
Denmark	562	0.4	0.14 - 1.4	40 - 400	16 - 164
Finland	620	0.4	0.15 - 1.5	40 - 400	0.1 - 1
France	4,836	4.6	0.10 - 1.0	460 - 4600	327 - 3266
Germany	7,437	12.0	0.16 - 1.6	1200 - 12000	1068 - 10680
Iceland	-	0.0	0.00 - 0.0	0 - 0	0 - 0
Ireland	237	0.2	0.08 - 0.8	20 - 200	20 - 200
Luxembourg	27	0.1	0.00 - 0.0	6 - 59	6 - 59
Norway	385	0.1	0.03 - 0.3	10 - 100	7 - 73
Portugal	1,137	0.7	0.10 - 1.0	70 - 700	70 - 700
Spain	5,076	3.8	0.07 - 0.7	376 - 3760	196 - 1955
Sweden	1,601	0.7	0.07 - 0.7	70 - 700	19 - 189
Switzerland	591	0.7	0.17 - 1.7	70 - 700	56 - 560
The Netherlands	1,308	1.9	0.15 - 1.5	190 - 1900	190 - 1900
United Kingdom	5,995	3.6	0.06 - 0.6	360 - 3600	360 - 3600
TOTAL	30,748	31.2	0.11 1.1	**3116** **31160**	
Other countries within OSPAR catchment					
Greenland	unknow	0[1]	unknown	0 - 0	0 - 0
Russia	unknown	8	unknown	800 - 8000	6 - 64
Liechtenstein	unknown	0[1]	unknown	0 - 0	0 - 0
Czech Republic	4406	1.1	0.02 - 0.2	110 - 1100	69 - 693
TOTAL					**2615 - 26145**

[1] Due to the size of the population it is assumed that no plastic industries are present in these countries.

Ranges are a result of low (0.01%) and high (0.1%) estimates for the emission factor.

Appendix 6 Overview of calculated fibre emissions

Reference year 2015

OSPAR Contracting parties					fibres in laundry effluent [tonnes]		fibres directly to surface water [tonnes]		fibres in STP effluent [tonnes]		total laundry fibre emission to water [tonnes]		fibres in sewage sludge [tonnes]	
	population 2015	Washing frequency ppp month (AISE 2015)	not connected to STP [Eurostat]		[min]	[max]	[min]	[max]	[min]	[max]	[min]	[max]	[min]	[max]
Belgium	11,299,190	5.8	16%	(2013)	51	231	8.2	36.9	12.1	54.3	20.3	91.2	31	140
Denmark	5,669,080	5.5	9%	(2015)	24	110	2.2	9.9	6.2	28.0	8.4	37.9	16	72
Finland	5,503,460	5.5	17%	(2013)	24	107	4.0	18.1	5.5	24.8	9.5	42.9	14	64
France	64,395,350	5.8	19%	(2013)	292	1316	54.1	243.4	67	300.3	121	543.7	172	772
Germany	80,688,540	5.8	3%	(2013)	366	1649	11.0	49.5	100	447.8	110	497.2	256	1151
Iceland	329,430	5.5	34%	(2010)	1	6	0.5	2.2	0.3	1.2	0.7	3.3	1	3
Ireland	4,688,470	7.5	35%	(2014)	28	124	9.6	43.4	5.0	22.5	14.6	65.9	13	58
Luxembourg	605,110	5.8	1%	(2014)	3	12	0.0	0.1	0.8	3.4	0.8	3.6	2	9
Norway	5,210,970	5.5	18%	(2013)	22	101	4.1	18.3	5.1	23.2	9.2	41.4	13	60
Portugal	10,349,800	6.3	29%	(2009)	51	230	15.0	67.5	10.1	45.4	25.1	112.9	26	117
Spain	46,121,700	6.3	22%	(2012)	227	1024	50.0	225.2	50	223.6	100	448.8	128	575
Sweden	9,779,430	5.5	13%	(2013)	42	189	5.5	24.6	10.3	46.2	15.7	70.8	26	119
Switzerland	8,654,270	5.8	2%	(2013)	39	177	0.8	3.5	10.8	48.5	11.6	52.1	28	125
The Netherlands	16,924,930	5.8	1%	(2013)	77	346	0.5	2.1	21.4	96.2	21.8	98.3	55	247
UK	65,015,480	7.5	0%	(2014)	382	1718	0.0	0.0	107	481.0	107	481.0	275	1237
Total	**335,235,210**				**1631**	**7339**	**165**	**745**	**410**	**1846**	**576**	**2591**	**1055**	**4748**
Average					4485		455		1128		1583		2901	

Appendix 6 continued….

OSPAR catchment														
	population in catchment 2015	Washing frequency ppp month (AISE 2015)	not connected to STP [Eurostat]	reference year	[min]	[max]	[min]	[max]	[min]	[max]	[min]	[max]	[min]	[max]
Belgium	11,299,190	5.8	16%	(2013)	51	231	8.2	36.9	12.1	54.3	20.3	91.2	31	140
Denmark	2,324,323	5.5	9%	(2015)	10	45	0.9	4.1	2.6	11.5	3.5	15.5	7	30
Finland	11,007	5.5	17%	(2013)	0	0	0.0	0.0	0.0	0.0	0.0	0.1	0	0
France	45,720,699	5.8	19%	(2013)	208	934	38.4	172.8	47	213.2	86	386.0	122	548
Germany	71,812,801	5.8	3%	(2013)	326	1467	9.8	44.0	89	398.5	98	442.6	228	1025
Iceland	329,430	5.5	34%	(2010)	1	6	0.5	2.2	0.3	1.2	0.7	3.3	1	3
Ireland	4,688,470	7.5	35%	(2014)	28	124	9.6	43.4	5.0	22.5	14.6	65.9	13	58
Luxembourg	605,110	5.8	1%	(2014)	3	12	0.0	0.1	0.8	3.4	0.8	3.6	2	9
Norway	3,804,008	5.5	18%	(2013)	16	74	3.0	13.3	3.8	16.9	6.7	30.2	10	43
Portugal	10,349,800	6.3	29%	(2009)	51	230	15.0	67.5	10.1	45.4	25.1	112.9	26	117
Spain	23,983,284	6.3	22%	(2012)	118	532	26.0	117.1	26	116.3	52	233.4	66	299
Sweden	2,640,446	5.5	13%	(2013)	11	51	1.5	6.7	2.8	12.5	4.2	19.1	7	32
Switzerland	6,923,416	5.8	2%	(2013)	31	141	0.6	2.8	8.6	38.8	9.3	41.6	22	100
The Netherlands	16,924,930	5.8	1%	(2013)	77	346	0.5	2.1	21.4	96.2	21.8	98.3	55	247
UK	65,015,480	7.5	0%	(2014)	382	1718	0.0	0.0	107	481.0	107	481.0	275	1237
Greenland	738,023	5.5	17%[1]		3	14	0.5	2.4	1	3.3	1	5.8	2	9
Liechtenstein	37,531	5.8	2%[3]		0	1	0.0	0.0	0	0.2	0	0.2	0	1
Russian Federation	1,147,655	6.1	15%[2]		5	25	0.8	3.8	1	5.8	2	9.6	3	15
Czech Republic	6,642,207	6.1	15%	(2013)	32	143	4.9	21.8	8	33.9	12	55.7	19	87
TOTAL	**274,997,810**				**1354**	**6095**	**120**	**541**	**346**	**1555**	**466**	**2096**	**889**	**3999**
Average					3725		331		950		1281		2444	

[1] no data, data for Finland were assumed to be representative for Greenland
[2] no data, data for Czech Republic were adopted
[3] no data, data for Switzerland were adopted

Appendix 7 Background information on emission factors of tyre wear and their environmental distribution

The information is copied from [131]; table numbers are adjusted to fit in the present OSPAR document.

1. Emission factors for urban driving, rural driving and highway driving used in The Netherlands.

There is usually more acceleration and braking within urban areas than within rural areas or on highways. There are also more corners and bends, the road network is more dynamic and the relative differences in speed are greater. Although it is known that tyre wear per km driven within urban areas is therefore higher than outside these areas, there is little data available to properly substantiate this. The limited information available is summarized in Table 25.

Table 25 Effects of driving style and environment on emission factors

Condition	Value	Unit	Source
Highway – 120 km/h Taking bends – 50 km/h	24 490	mg/km/tyre	Dannis [26]
Road bends in urban areas Driving gently Driving professionally[1]	30 12 70	mg/km/tyre	LeMaitre et al. [75])
Dry conditions Winter compared to summer	150 140	%	

[1)] "Driving professionally": Accelerating quickly, keeping to the maximum speed as much as possible and braking little.

Data published by Dannis (1974) and LeMaitre et al. (1998) confirms that bendy roads cause greater wear. The extremely high value recorded by Dannis (1974) is no longer considered to be realistic. Thanks to advances in technology, the properties of tyres (such as wear resistance and grip) have significantly improved in the period since Dannis conducted his study (early 1970s).

The data set out in table 9 clearly shows that wear per kilometer is greater when there are lots of bends and when during acceleration, as in urban traffic, but does not provide an exact ratio. **As an initial estimate, it is assumed that emission factors within urban areas are a factor 2** higher per kilometer driven than on motorways and secondary roads. This is based on the ratio between values measured by Le Maitre et al., but only "driving gently" is considered to be an underestimation of current motorway traffic. Table 10 sets out the definitive differentiated emission factors.

2. Distribution of tyre wear emissions to various environmental compartments

A 100% distribution of emissions within urban areas to the sewers is not probable. A GIS overlay of the land use database of the Netherlands with sewer areas in the emissions registration scheme reveals that exactly 50% of the surface area of sewer areas comprises paved ground (see Table 26).

Table 26 Results GIS overlay / sewer areas in urban areas in The Netherlands

Aggregate name	Ground area [ha]	Sewer area [ha]
Ground - paved	433,649	341,061
Ground - unpaved	2,893,753	336,033
Ground - semi-paved	50,136	16,016
Total	3,377,538	693,109

The fact that 50% of the sewer area is paved does not necessarily mean that 50% of the emissions from the deposition of coarse particulates will fall on the sewers. In the sewer environment, runoff coefficients are used to determine the amount of rainwater falling in the sewer area that goes into the sewers. A runoff coefficient of 50% is frequently applied. In a recent study carried out by the Netherlands Organization for Applied Scientific Research (TNO) in the service areas of two municipal wastewater treatment plants in North Brabant in 2005, runoff coefficients of 50% and 90% were measured for Den Bosch and Asten respectively.

Because water acts as the medium for transporting the contamination, an obvious starting point for calculations would be the distribution of water among the compartments.

However, there are various factors leading to deviations upwards and downwards:

- An amount of water evaporates, whereby the running-off surface area is greater than the surface area that can be derived from precipitation and water going into the sewers. If the deposition of precipitating particulates is distributed reasonably homogeneously, the running-off surface is a good measurement for the quantity of emission that runs off. However, on unpaved ground, part of the contamination is filtered off, whereby the quantity of pollution per quantity of water (concentration) in the unpaved ground reduces (decreasing contribution to sewers)
- Coarse particulate matter is deposited closer to the source. The source is the surfaced road. The deposition will therefore be more concentrated there on the road and as a result emissions will be greater in ratio to the quantity of water (concentration) (increasing contribution to sewers)
- A proportion of the contamination on the roads will be collected via street refuse. This proportion does not end up in the sewers. The majority of street refuse is coarse sand that contains relatively little contamination (decreasing contribution to sewers)
- A part of the soil material in unpaved ground will leach into the sewers. Pollution will have accumulated in this soil material over the course of time, originating from tyre wear among other sources. This means that a small part of the pollution originally accumulated in the soil will still end up in the sewers (increasing contribution to sewers).

These counteracting processes make it difficult to reach a quantitatively well-founded decision. It can definitely be stated that the quantity of emissions going into the sewers is below 100%. It is hereby proposed (until better measuring data is available) to assume that 50% goes into the sewers within urban areas instead of 100%. However, because a part of the pollutant loading from the soil still leaches into the sewers, it is necessary to take a

slightly higher input to the sewers into consideration. An input of 60% into the sewers is therefore selected as the provisional value.

Distribution for rural driving

The distribution of emissions of coarse fraction of tyre particulates within rural areas is more complex than within urban areas. Blok conducted a fairly extensive study in this respect [9]. According to Blok, approximately 70% of the total quantity of material largely ends up in the soil of the verges via the mechanism of run-off. According to Blok, the remaining 30% is distributed via the mechanism of drift. In this respect, we assume that the finest fraction of this 30% (approximately 5% fine particulate matter) will be transported further away via atmospheric transport. According to Blok [9], the largest proportion (25 of the 30%) distributed via drift does not travel further than 4.5 m (other roads) to 6 m (motorways) from the edge of the road. It is not known what share of the surface area situated at 4.5 to 6 meters alongside roads is made up of ditches, but the value will be less than 50% of the surface area between 4.5 and 6 meters from roads. As an initial estimate, we assume a value of half of 25%, rounding off downwards, which gives an estimate of 10% direct emissions to surface water.

Table 27 provides the division of tyre particulates to the various environmental compartments.

Table 27 Proposed distribution percentages for tyre particulates to compartments

	Atmosphere	Soil	Surface water	Sewers
Fine particulates (incl. metals)				
Urban driving	100	0	0	0
Rural driving	100	0	0	0
Highway driving	100	0	0	0
Coarse particulates (incl. metals)				
Urban driving	0	40	0	60
Rural driving	0	90	10	0
Highway driving	0	90	10	0

Appendix 8 Overview of monitoring data in marine water samples.

(Based on publications until end of 2015)

Location	Type	Size	Number of samples	Amount (n/m^3)	Ref
Region III UK, western English Channel	fibres (61 %) fragments (36%) beads (3%)	Fibres: widths varying between 6 and 175 µm and lengths > 250 µm fragments and beads: predominantly < 250 µm	North Sea	0.27	23
Region II Germany Jade Bay	granules, fibres, black carbon particles	fibres between <100 µm and about 1 mm most granular material < 100 µm, occasionally up to 300 µm		0.064 granular particles 0.088 fibres 0.030 black carbon particles In total 0.182 particles	32
Region II The Netherlands	75% blue fibres	< 300 µm	3 with a total volume of 278 m^3 water	average 0.9 (range 0.2 - 1.6)	76
Region II French, Belgian and Dutch North Sea coast	not reported	30 - 300 µm	12	Seawater: average 0.4 ± 0.3 particles/L (range: 0.0 - 0.8) recalculated to 0.004 particles/m^3	141
Regions II, III, IV and V Ireland	fibres (95.9%), fragments, bead and foam most common colour blue (37.7%)	range 0.2 mm to 43.2 mm, with the most common size class between 1.25 and 2.5 mm (median 2.4 mm) 89% of total plastic identified is < 5 mm	470 (2000 L samples)	2.46 ± 2.43 (median: 2, range)	83
Region IV Portugal				0.02–0.036	44

Appendix 9 Overview of monitoring data in marine sediment samples

(Based on publications until end of 2015)

Location	Type	Size	Number of samples	Amount (n/kg sediment)	Ref.
North Sea					
Region II Belgium	fibres (59%) (all sampling locations) granules(25%) (all sampling locations) plastic films (4%) (all sampling locations, with some exceptions) spherules (12%) (exclusively in harbour sediment)	38 µm - 1 mm	3 harbours, sampling 3-4 locations per harbour. 2 coastal zones, 3 locations per site 3 beach sites, 2-3 locations per site	166.7 ± 92.1 (range 126.1 - 213.4) particles/kg dw (harbours) 97.2 ± 18.6 (range 91.9 - 105.2) particles/ kg dw (coastal zone)	19
Region II NL, harbours, coastal zones, offshore up to 175 km from coast.	Spheres, fibres and fragments	two categories: 1 - 300 µm and 300 - 5000 µm. North Sea: > 300 µm (54% particles), < 300 µm (46% particles) Wadden Sea: > 300 µm (31% particles), < 300 µm (69% particles) Harbours: > 300 µm (20% particles), < 300 µm (80% particles)	3 harbours, 3-4 locations per harbour. 2 coastal zones, 3 locations per site	Total mean 840/kg dw (median 500/kg dw, range 100 - 3600) North Sea: 440/kg dw (SD 160) Wadden Sea: 770/kg dw Harbours: 3300/kg dw (SD 420)	78
Region II Germany, East Frisian islands	Granules and fibres (fragments and pellets absent!)	The majority of granular material was < 100 µm in size	15 locations, 5 individual samples per location. North sea: 12 locations Wadden Sea: 1 location Harbours: 2 locations.	Kachelotplate range: 0 - 49600 granules/kg 100-1400 fibres/kg. Spiekeroog: 1500 - 5800 (mean 3800) particles/kg	80
Region II Sea coast: NL, BE, FR	not specified, only: particles.	3 - 1175 µm	29	range: 1.5 to 23.4 particles kg 1 dry sediment average 6.0 (SD 5.7) parts/kg dry sediment	141
Region IV Porcupine Abyssal Plain, deep sea sediment		Dimensions of microplastics found (length x width): 161 - 137 µm 83 - 44 µm 125 - 76 µm	3	1 particle per 25 cm^3 Recalculated to 31 particles/kg sediment (assuming a sediment weight of 1.3 g/cm^3)	147
Region II, Sweden	Milkwhite/transparent spheres	0.5 – 1 mm		332 particles per 100 ml (industrial harbour Stenungsund) 2 – 34 per 100 ml (Tjuvkols harbour, Stenungsund outside the harbour)	97
Region III, UK Beach		1.6 – 1.0 µm	30	<1 – 8 items/50 mL	15
Region III, Portugal, beach		1.2 µm – 5 mm		133.3 items/m^2	88

Location	Type	Size	Number of samples	Amount (n/kg sediment)	Ref.
Region II, Germany beach		1 – 15 mm		5000 – 7000 items/m^3 (urban beach) 150 – 700 items/m^3 (rural beach)	[144]
Region II Germany East Frisian islands		< 1mm	12	1.3 – 2.3 items/kg dw	[28]

Appendix 10 Overview of microplastics in marine biota

(Based on publications until end of 2015)

Location	Species	Size	Number of samples	Amount (n)	Ref
North Sea					
Region II North Sea, NL	*Megaptera novaeangliae* (baleen whale)	n=9, (3.3-12) x (0.3–8.2) x (0.04–0.2) mm n=4, (2.3–170) x (0.1–1.5) x (0.1–0.4) mm n=1, 3.6 x 2.4 x 0.1 mm n=1, 5.8 x 3.3 x 0.01 mm n=1, 1.1 x 0.8 x 0.2 mm	1/5th to 1/10th of the intestines was sampled	Total number of particles in intestine samples. Size (mm): n=9, (3.3-12) x (0.3–8.2) x (0.04–0.2) n=4, (2.3–170) x (0.1–1.5) x (0.1–0.4) n=1, 3.6 x 2.4 x 0.1 n=1, 5.8 x 3.3 x 0.01 n=1, 1.1 x 0.8 x 0.2 for the total animal the number is 5 to 10 times higher.	[5]
Region II, Ireland	*Mesoplodon mirus* (female beaked whales): one calf, two adults			mean nr of 2.95 particles per section of intestine (± 2.09 range 0 - 7) mean nr of 7.25 particles per stomach compartment (± 2.63, range 5 - 11) total in digestive tract: 88 particles	[84]
Region II and III North Sea; BE, NL, UK and FR	*Crangon crangon* (L.), 45–55 mm carapace length (eye socket to tail)	200 µm up to 1000 µm	165 individuals	average microplastic content of 0.68 ± 0.55 microplastics/g ww or 1.23 ± 0.99 microplastics/shrimp based on an average weight of 1.8 g/shrimp. An average value of 1.03 fibres/g ww was established for *C. crangon* in the shallow water habitats	[29]
Region II North Sea; northern and southern part	herring, gray gurnard, whiting, horse mackerel, haddock, atlantic mackerel, and cod	< 5 mm, median size 0.8 mm	33 fish sampled	6 individuals contained more than 1 particle. The maximum numbers of particles discovered in one fish was 4.	[43]
Region II North Sea, NL coast	Littorina littorea (Periwinkle) Gammarus sp.(Amphipod) Crassostrea gigas(Pacific oyster) Mytilus edulis(Blue mussel) Carcinus maenas(Sand crab)	1 - 300 µm 300-5000 µm	7	Periwinkle: 20 particles/g dw Amphipod: 11 particles/g dw Pacific oyster (oosterschelde): 87 particles/g dw Blue mussel (oosterschelde): 105 particles/g dw Sand crab: 0 particles/g dw Pacific oyster (Botlek harbour): 30 particles/g dw Blue mussel (Ter Heide North Sea) 19 particles/g dw	[78]

Location	Species	Size	Number of samples	Amount (n)	Ref
Region III Ireland, west-coast	Merlangius merlangus (whiting) Micromesistius poutassou (blue whiting) Trachurus trachurus (Atlantic horse mackerel) Trisopterus minutus (poor cod) Zeus faber (John Dory) Aspitrigla cuculus (red gurnard) Callionymus lyra (Dragonet) Cepola macrophthalma (redband fish) Buglossisium luteum (solenette) Microchirus variegates (thickback sole)	The size (length at the longest point) of plastics range: 0.13 mm - 14.3 mm most common size class: 1.0–2.0 mm. 92.4% of the sample < 5mm	504 fish (184 ingested plastics)	1.90 ± 0.10 particles per fish (average)	85
Region II North Sea coast; NL, BE, FR	*Arenicola marina,* size: 7 x 11 cm (lugworm, deposit feeder) *Mytillus edulis*, size: 4 x 4.5 cm (blue mussel, filter feeder)	particle size range: *M. edulis* tissue: 20 - 90 µm *M. edulis* faeces: 15 - 500 µm *A. marina* tissue: 15 - 100 µm *A. marina* faeces: 35 - 1000 µm	not reported	*M. edulis*: 0.2 (SD 0.3) parts/g tissue *M. edulis*: 0.1 (SD 0.2) parts/g faeces *A. marina*: 1.2 (SD 2.8) parts/g tissue *A. marina*: 0.3 (SD 0.6) parts/g faeces	142
Regions II and III North Sea, GE Atlantic Ocean, GB, FR	*Mytilus edulis* adult size (5.2 ± 0.4 cm) *Crassostrea gigas*average shell length of 9.0 ± 0.5 cm. both cultured for consumption	after gut depuration: *M. edulis* 5 - 10 µm (50.0%) *C. gigas*: 11 - 15 µm (29.6%), 16 - 20 µm (33.3%) The depuration period resulted in the removal of all (in *M.*	*M. edulis*: n = 36 *C. gigas*: n = 10	*M. edulis*: 0.36 ± 0.07 particles g-1 ww without depuration *M. edulis* : 0.24 ± 0.07 particles g-1 ww after three days depuration *C. gigas*: 0.47 ± 0.16 particles g-1 ww *C. gigas*: 0.35 ± 0.05 particles per gram soft tissue (ww) after depuration	145

Location	Species	Size	Number of samples	Amount (n)	Ref
		edulis) or the majority (in *C. gigas*) of the largest microplastics (i.e. particles > 25 µm in length).			
Regions II, III and IV Tagus estuary, Portugal Ebro estuary, Spain Po estuary, Italy Mussels farms: France, Italy, Denmark, Spain and the Netherlands	*Mytilus galloprovincialis* (from estuary and mussel farm) *Mytilus edulis*(from mussel farm)		Acid mix method: 5 replicates (each one mussel) (both estuary and farm mussels) Nictric acid method: 12 replicates (each three mussels) (only estuary mussels: *M. galloprovinciali*)	acid mix method: commercial mussels (*M. galloprovincialis* and *M. edulis*): 0.13 ± 0.14 total microplastics/g ww estuary mussels (*M. galloprovincialis*): 0.18 ± 0.15 total microplastics/g ww nitric acid method: estuary mussels (*M. galloprovincialis*): 0.12 ± 0.04 total microplastics/g ww	150

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