REPORTS AND STUDIES

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GESAMP

Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection

SOURCES, FATE AND EFFECTS OF MICROPLASTICS IN THE MARINE ENVIRONMENT: PART 2 OF A GLOBAL ASSESSMENT

IMO FAO UNESCO IOC WMO UNIDO IAEA UN UNEP UNDP

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SOURCES, FATE AND EFFECTS OF MICROPLASTICS IN THE MARINE ENVIRONMENT: PART TWO OF A GLOBAL ASSESSMENT

A report to inform the Second United Nations Environment Assembly

GESAMP Working Group 40
2nd phase

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GLOSSARY

Organizations, techniques and other terms

Short form	Full name
ALDFG	Abandoned, Lost and otherwise Discarded Fishing Gear
FAO	Food and Agriculture Organization
FT-IR	Fourier transform infrared spectroscopy
UNEA	United Nations Environment Assembly

Common polymers

Short form	Full name	Short form	Full name
ABS	Acrylonitrile butadiene styrene	PGA	Poly(glycolic acid)
AC	Acrylic	PLA	Poly(lactide)
EP	Epoxy resin (thermoset)	PP	Polypropylene
PA	Polyamide 4,6, 11, 66	PS	Polystyrene
PCL	Polycaprolactone	EPS (PSE)	Expanded polystyrene
PE	Polyethylene	PU (PUR)	Polyurethane
PE-LD	Polyethylene low density	PVA	Polyvinyl alcohol
PE-LLD	Polyethylene linear low density	PVC	Polyvinyl chloride
PE-HD	Polyethylene high density	PU (PUR)	Polyurethane
PET	Polyethylene terephthalate	SBR	Styrene-butadiene rubber

Common chemical additives in plastics

Short form	Full name	Examples of function
BPA	Bisphenol A	a monomer used in the manufacture of polycarbonates and epoxy resins
DBP	dibutyl phthalate	anti-cracking agents in nail varnish
DEP	diethyl phthalate	skin softeners, colour and fragrance fixers
DEHP	di-(2-ethylhexyl)phthalate	plasticizer in PVC
HBCD	hexabromocyclododecane	flame retardant
NP	nonylphenol	stabilizer in food packaging and PVC
PBDEs	Polybrominated diphenyl ethers (penta, octa & deca forms)	flame retardants
	nonylphenol	stabilizer in PP, PS
phthalates	Phthalate esters	improve flexibility and durability

Common organic contaminants absorbed by plastics

Short form	Full name	Origin
DDT	dichlorodiphenyltrichloroethane	insecticide
PAHs	Polycyclic aromatic hydrocarbons	combustion products
PCBs	polychlorinated biphenyls	cooling and insulating fluids, e.g. in transformers

Contents

EXECUTIVE SUMMARY

This report provides an update and further assessment of the sources, fate and effects of microplastics in the marine environment, carried out by Working Group 40 (WG40) of GESAMP (The Joint Group of Experts on Scientific Aspects of Marine Protection). It follows publication of the first assessment report in this series in April 2015 (GESAMP 2015). The issue of marine plastic litter was raised during the inaugural meeting of the United Nations Environment Assembly (UNEA) in June 2014. Delegates from 160 countries adopted Resolution 1/6 on 'Marine plastic debris and microplastics' (Annex I). The resolution welcomed the work being undertaken by GESAMP on microplastics and requested the Executive Director of UNEP to carry out a study on marine plastics and microplastics. This was to be based on a combination of existing and new studies, including WG40. This provided the motivation for GESAMP to revise the original terms of reference to reflect both the request from UNEP to contribute to the UNEA study, and the key recommendations from the WG40 2015 report.

Each main section begins with key messages followed by a short summary of related findings from the first report. Each section ends with conclusions, knowledge gaps and research priorities. Greater effort has been made to describe the nature, distribution and magnitude of sources of macro- and microplastics. These are described by sea-based and land-based sectors, together with the main entry points to the ocean. Spatial (regional) and temporal differences in both sources and entry points are examined. One previously unrecognized source of secondary microplastics highlighted is debris from vehicle tyres.

The distribution of microplastics in the five main ocean compartments (sea surface, water column, shoreline, seabed and biota) are described, together with the transport mechanisms that regulate fluxes between compartments. Regional 'hot-spots' of sources, distribution and accumulation zones are reported, in response to the UNEA request.

The effects of microplastics on marine biota have been explored in greater detail.

Greater attention has been given to the interaction of microplastics with biota. A comprehensive literature review has been assembled with tables summarising the occurrence of microplastics in a wide variety of marine organisms and seabirds. There does appear to be an association between uptake of microplastics and changes in the physiological or biochemical response in some species, observed in laboratory experiments. It is not clear whether this will be significant at a population level with current observed microplastic numbers. The current understanding of the interaction of plastic-associated chemicals with biota is reviewed, using laboratory-based experiments, theoretical studies and field-based observations. It appears very likely that this interaction will be dependent on: i) the species; ii) the relative degree of contamination of the plastic, the biota concerned and the marine environment (sediment, water, foodstuff) in that region; iii) the size, shape and type of plastics; and iv) several time-related variables (e.g. environmental transport, gut transfer, absorption/desorption rates). This remains a contentious area of research. The occurrence of nano-sized plastics in the marine environment has yet to be established and we are dependent on drawing inferences from other fields of science and medicine when considering possible effects. Microplastics can act as vectors for both indigenous and non-indigenous species. Examples include pathogenic Vibrio bacteria, eggs of marine insects and the resting stages of several jellyfish species.

A new section considers the possible effect of microplastics on commercial fish and shellfish. Microplastics have been found in a variety of commercial fish and shellfish, including samples purchased from retail outlets. Generally the numbers of particles per organism are very small, even for filter-feeding bivalves in coastal areas bordered by high coastal populations. At these levels it is not considered likely that microplastics will influence the breeding/development success of fish stocks (food security) nor represent an objective risk to human health (food safety). However, data are rather scarce and this is an area that justifies further attention.

The economic aspects of microplastic contamination are considered in another new section. This relies heavily on studies looking at the effects of macrodebris on various sectors (e.g. fisheries, shipping, tourism, waste management), given the paucity of knowledge of direct economic effects of microplastics. Acting on macroplastics may be easier to justify, as the social, ecological and economic effects are easier to demonstrate. This in turn will reduce the quantities of secondary microplastics being generated in the ocean. One significant cost that may be incurred would be the provision of wastewater treatment capable of filtering out microplastics. Such systems are relatively common in some rich countries but absent in many developing nations. Clearly, there are many other reasons to introduce improved wastewater treatment (nutrient reduction, disease prevention), with reduction in microplastics being an additional benefit.

Social aspects are focused around factors influencing long-term behaviour change, including risk perceptions, perceived responsibility and the influence of demographics. This is key to implementing effective, acceptable measures.

A separate section summarizes good practice guidance on sampling and analysis at sea, in sediments and in biological samples. There are no global 'standards' but if these guidelines are followed then it will be easier to generate quality-assured data, in a cost-effective manner, and for datasets to be compared and combined with more confidence.

The final main section presents an initial risk assessment framework. Having described some basic principles about risk, likelihood and consequences the section provides a conceptual framework and two case examples (one real, one hypothetical) of how the framework can be utilized.

The report concludes with key conclusions and recommendations for further research.

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1 INTRODUCTION

1.1 Context

Concern about the quantity of plastic and microplastic debris in the ocean has grown rapidly in recent years. This has been evident in terms of the increased interest from governments, Intergovernmental Organizations (IGOs), regional seas organizations, the private sector, environmental NGOs, special interest groups, the media and the scientific community.

GESAMP recognized the importance of this topic within its Emerging Issues programme. It undertook a number of scoping activities that culminated in setting up a working group (Working Group 40, WG40) to undertake an initial assessment of: 'Sources, fate and effects of microplastics in the marine environment – a global assessment', published in April 2015 (GESAMP 2015[1]). The assessment included a number of recommendations for further investigation, to cover certain topics in greater depth or introduce new elements into the assessment.

The United Nations Environment Assembly (UNEA)[2] was created to help inform the development of the UN Sustainable Development Goals (SDGs), and to deliver the environmental dimension of the 2030 Agenda for Sustainable Development.[3] The issue of marine plastics and microplastics was raised during the inaugural meeting of the UNEA in June 2014. Delegates from 160 countries adopted Resolution 1/6 on 'Marine plastic debris and microplastics' (Annex I).

Paragraph 12 of Resolution 1/6 reads:

'[The United Nations Environment Assembly] … Welcomes the initiative by the Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection to produce an assessment report on microplastics, which is scheduled to be launched in November 2014 [GESAMP 2015]'.

Paragraph 14 of Resolution 1/6 included a request to the Executive Director of UNEP to carry out a study:

'… building on existing work and taking into account the most up-to-date studies and data, focusing on:

(a) Identification of the key sources of marine plastic debris and microplastics;

(b) Identification of possible measures and best available techniques and environmental practices to prevent the accumulation and minimize the level of microplastics in the marine environment;

(c) Recommendations for the most urgent actions;

(d) Specification of areas especially in need of more research, including key impacts on the environment and on human health;

(e) Any other relevant priority areas identified in the GESAMP assessment of the Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection;'

In addition to reviewing the extensive published literature on the topic, it was intended that the UNEA report should reflect the findings of several related but separate studies supported principally by UNEP:

a) core study focusing on strengthening the evidence base with regard to microplastics (this report);

b) study on the impact of microplastics on fisheries and aquaculture (FAO/UNEP);

c) compilation of Best Available Techniques (BATs) for solid waste management (undertaken by Tetra Tech);

d) modelling component, engaging wider modelling/oceanographic community (undertaken by CSIRO); and

e) socio-economic component, engaging researchers and universities to look at social aspects/welfare impacts and economic effects (undertaken by IEEP).

The UNEA report was published during the Second United Nations Environment Assembly (UNEA-2), 23 to 27 May 2016 (UNEP 2016).

1.2 GESAMP WG40 work programme and timeline

The new work programme has two main objectives:

1. to carry out a comprehensive assessment of the topic with input from a wide range of disciplines over a 3 to 4 year timeline;

2. to provide input to the UNEA-2 (23 to 27 May 2016) on topics of particular interest to UNEP and FAO.

Revised Terms of Reference (ToRs)

1. assess the main sources and categories of plastics and microplastics entering the ocean;

2. assess and utilize a range of physical and chemical models to simulate the behaviour of plastics and microplastics in the ocean in order to improve current assessment technologies;

3. assess the occurrence and effects of microplastics in commercial fish and shellfish species, including associated additive chemicals and contaminants in the edible fractions;

4. assess local, regional and global scales of accumulation of plastics and associated chemicals (additives and absorbed contaminants), including SIDS and regional hot-spots;

[1] http://www.gesamp.org/publications/publicationdisplaypages/reports-and-studies-no.-90
[2] http://unep.org/unea/
[3] Lee, G.E., 2014. UNEA 2014: Ground-Breaking Platform for Global Environmental Sustainability [Online]. Available at: http://climate-exchange.org/2014/07/02/unea-2014-ground-breaking-platform-for-global-environmental-sustainability/ [accessed 22 December 2015]

5. assess the effects of nano-sized plastics on marine organisms;

6. assess the risk of physical and chemical effects of ingested microplastics on marine organisms;

7. assess the significance of plastics and microplastics as a vector for organisms, facilitating the spread of non-indigenous (alien) species;

8. develop guidelines covering terminology and methodologies: i) size and shape definitions of particles; ii) sampling protocols for the whole spectrum of particle sizes in surface and sub-surface seawater, seabed sediments, shorelines and biota; and, iii) methodologies for physical and chemical identification and analysis of polymers and associated chemicals;

9. assess social and economic aspects influencing both the entry of plastics/microplastics into the ocean and the potential consequences from the resulting contamination; and

10. develop and utilize effective mechanisms for communicating the progress and conclusions of the working group to a wide audience (public and private sector).

Note: ToRs 1 to 7 cover specific areas of interest whereas ToRs 8 to 10 are cross-cutting.

Output

1. Report – Sources, fate and effects of microplastics in the marine environment – part two of a global assessment; and

2. Report – Sources, fate and effects of microplastics in the marine environment – part three of a global assessment. Due to be published in 2018 (content to be decided following publication of part two).

1.3 Structure and scope of the report

The current report takes the outcome of the first GESAMP assessment as a starting point. The sections on sources and fate have been expanded, and potential ecological impacts investigated in greater depth. A separate section is devoted to the potential impacts of microplastics on commercial fish and shellfish species. Greater effort has been directed at assessing social and economic aspects of microplastics, drawing on related literature as appropriate. A separate section discusses advances on sampling and analytical techniques, and the advantage of harmonized approaches, to allow greater data sharing and comparison. Risk assessment, to support decision making, is also given more prominence.

The intention is for the current report to provide a more robust evidence base to focus and support the development and implementation of potential solutions to reduce the impact of marine microplastics. It provides some examples, but does not advocate potential solutions and this would have been outside the scope of the ToRs. The report covers the Driver-Pressure-State-Impact components of the DPSIR conceptual framework for the adaptive management of environmental stressors (Figure 1.1). The Response component, devising possible microplastic reduction measures, is discussed in the UNEA report (UNEP 2016).

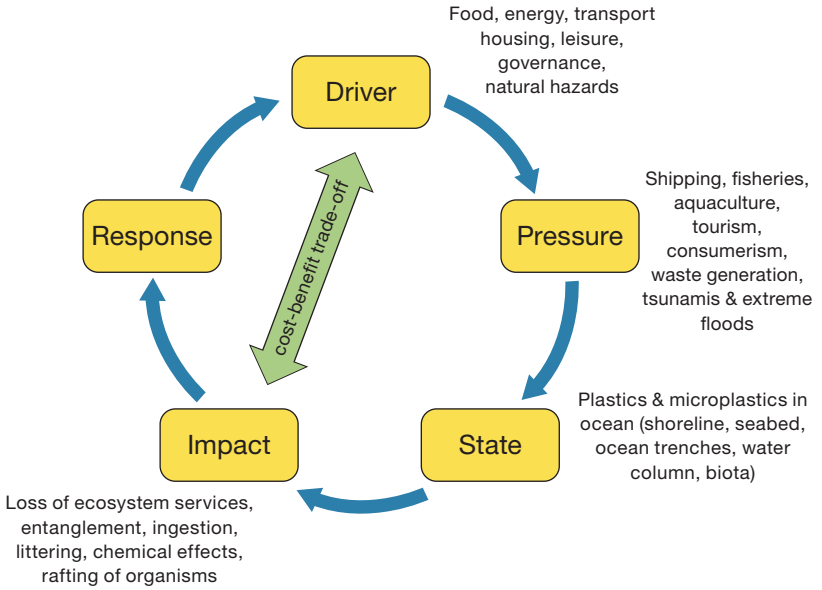


Figure 1.1 *DPSIR framework in relation to inputs and impacts of marine plastics and microplastics (UNEP 2016)*

2 SOURCES OF MICROPLASTICS

Key points

1. There are primary and secondary sources of microplastics. The distinction 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).
2. Fragmentation and degradation plays an essential role in the formation of secondary microplastics, but the processes are poorly understood.
3. There is evidence that microplastics are littered into the environment at all steps in the life cycle of a plastic product from producers to waste management.
4. Microplastics can enter the marine environment via riverine systems, coastlines, directly at sea from vessels and platforms or by wind-induced transport in the atmosphere.
5. Methods of defining microplastics, sampling and measurement vary considerably among studies, source sectors and geographical regions making it difficult to synthetize data across studies.

2.1 Lessons from the first assessment

Identification of sources is important to accurately assess the quantities of microplastics entering the marine environment, to provide an indication of regional or local 'hot-spots' of occurrence and accumulation, and to develop and monitor mitigation efforts and policies (GESAMP 2015). The identified challenges include uncertainties in the sources as well as the pathways by which microplastics arrive at a specific destination. There are two types of microplastics sources.

The primary sources are manufactured microplastics that were designed for particular applications. These primary particles may be released from discrete point sources such as plastic processing plants (production pellets or powders for injection moulding) or from more diffuse and regular source points such as populated places along rivers and coastlines (microbeads, industrial abrasives). There is a lack of quantitative data on inputs via small, but regular and persistent, losses of primary microplastics from multiple sources (GESAMP 2015). The secondary sources are microplastics created by fragmentation and degradation of macroplastics, including fibres from synthetic textiles. Estimating the source distribution of secondary microplastics inevitably relies on accurate assessment of the distribution and sources of macroplastics and on a good understanding of the degradation process.

The discussion on primary and secondary sources of microplastics is further developed in this chapter. Particularly, fragmentation and degradation of plastics in the environment is emphasized as it plays a major role in the release of secondary microplastics. Sources of microplastics are presented by sectors and characterized under producers/converters, sectoral consumers (land-based and sea-based), individual consumers and waste management. The main pathways from the source sectors into the marine environment are reviewed including riverine, coastal, marine-based and atmospheric inputs. The pathways are also categorized between primary and secondary sources of microplastics. Finally, this chapter provides a discussion on spatial and temporal variability of microplastic sources.

2.2 Primary and secondary sources of microplastics

2.2.1 *Overview of microplastic sources*

Marine debris has become a global environmental issue and a growing concern since the rise of the plastic industry in the mid-1950s. Annual global plastic production has increased steadily and reached 311 million tonnes in 2014. The majority is used to make items of packaging and for construction. Smaller proportions are used in a range of other applications, including the automotive industry, agriculture and for electrical and electronic components. See Figure 2.1 below for an example of plastic production data by sector in the EU.

The increase in plastic use has been accompanied by an increase in plastic litter in the ocean. The total number of macro- and microplastic objects has the potential to affect marine life with associated socio-economic consequences. As a result, it is important that we understand the sources and sinks of plastic debris into the ocean so that we can identify potential risks.

In addition to understanding sources and sinks, it is important that we come to an agreement on how to categorize the different types of debris. Microplastics have been attributed to several different size ranges which can sometimes be confusing and/or hinder data comparisons. It has become common to use the definition of any plastic particle $<$5 mm in diameter, which includes particles in the nano-size range (Table 2.1).

Table 2.1 Summary of size definitions of marine plastic litter and common sources

Size categories of marine plastic litter	Diameter			
	Micro <5 mm	**Meso <2.5 cm**	**Macro <1 m**	**Mega >1 m**
Source	Primary microplastics Secondary microplastics – fragmentation of larger plastic items	Direct and indirect: including fragmentation of larger plastic items	Direct: lost items from maritime activities or from rivers	Direct: abandoned gear, catastrophic events
Examples of marine litter	Primary: resin beads, microbeads from personal care products; Secondary: textile fibres, tyre dust	Bottle caps, fragments	Plastic bags, food and other packaging, fishing floats, buoys, balloons	Abandoned fishing nets and traps, rope, boat hulls, plastic films from agriculture

Primary microplastics include production pellets/powders and engineered plastic microbeads, used in cosmetic formulations, cleaning products and for industrial abrasives.

In contrast, secondary microplastics come from larger plastic items that are degraded and consequently fragmented, mostly due to weathering degradation, into microplastic particles.

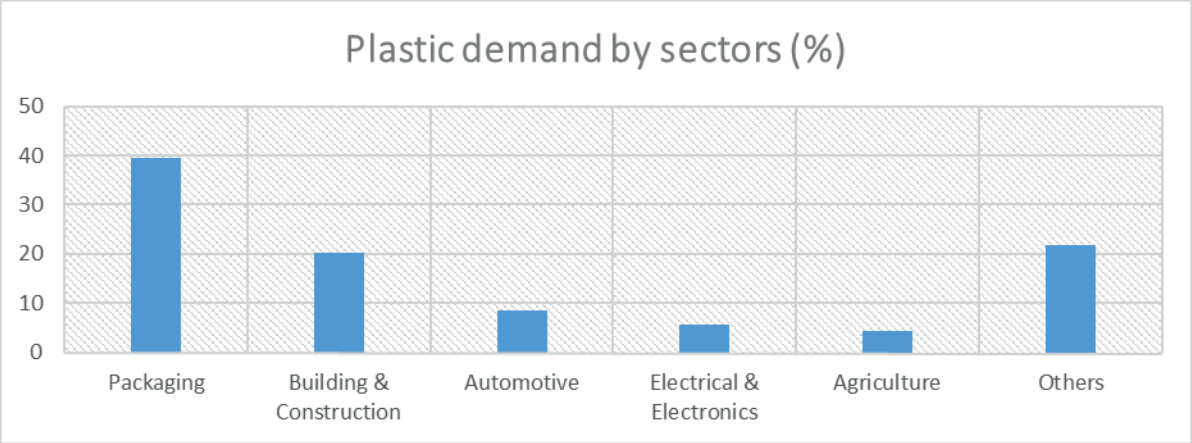


Figure 2.1 *European plastic demand by sectors 2013 (adapted from PlasticsEurope 2014)*

2.2.2 *Fragmentation and degradation*

The widespread degradation and fragmentation of plastic is one of the key factors causing microplastics to be ubiquitous in the marine environment. While there is extensive literature on the loss of mechanical integrity of plastics with weathering on both land and beach (O'Brine and Thompson 2010), as well as the ocean environments (Andrady 2011), studies on fragmentation as a consequence of weathering are sparse in the literature. This is partly due to lack of historical interest in the degradation process past the point where a product cannot be used in the intended application – durability is a key quality of most plastics. The time scale for which we can relate is also an important factor. Plastic was introduced in the 1950s, which means that observations can only be carried from that period of time, and by controlled laboratory experiments, and the long-term behaviour of plastics in the marine environment is essentially unknown. Weathering related degradation results in a progression of changes that include the loss in mechanical integrity, embrittlement, further degradation and fragmentation. Biodegradation of plastics occurs at a very slow rate; only 1% to 1.7% decrease in mass was observed in laboratory-accelerated degradation of PE over a 30-day duration by microorganisms isolated from marine waters (Harshvardhan and Jha 2013). Fragmentation, however, is most likely to occur at advanced stages of degradation well beyond embrittlement for most plastics, mainly due to exposure to solar UV radiation (Andrady 2011). As a result, not only are the fragmentation kinetics and processes very poorly understood, but there are no reliable estimates of the time to embrittlement of different types of plastics exposed to weathering either on land or at sea under a specified set of conditions.

The general methodology currently used in studies is to expose samples in the field followed by assessing their mechanical integrity in laboratory tests. A somewhat different approach to studying the degradation and fragmentation of plastics in the ocean was pursued in a recent Clean Sea project (Gerritse and Vethaak 2015). This involved the use of a marine

mesocosm exposed to fluorescent lighting. The degradation of various 'compostable' and 'durable' plastic polymer varieties was monitored using changes in their electrical resistance, weight loss and generation of microplastic particles.

Though not experimentally demonstrated as yet, model calculations suggest that smaller plastic particles degrade and split into smaller fragments at faster rates (Gerritse and Vethaak 2015). The larger specific surface area generated with fragmentation allows improved contact with water/sediment with faster leaching or sorption rates for chemicals and additional space for biofouling. In addition, it exposes a larger area for chemical, physical and biological degradation reactions (Gewert et al. 2015)

The degradation processes that result in converting macroplastics or meso-plastics debris into microplastics likely continues beyond this stage to nano-sized plastics (<100 nm). In fact, if the microplastic is exposed to solar UV radiation the increased surface area would accelerate such degradation. For example, the formation of nanoplastics (mainly in the 100 to 500 nm range) occurs during the degradation of natural rubber latex condoms in outdoor freshwater microcosms (Lambert et al. 2013). However, nanoplastics have not been detected as yet in the marine environment (mainly due to the logistics challenges in analytical procedures) and the range of marine organisms exposed to them are unknown (GESAMP 2015; Koelmans et al. 2015).

Two recent studies estimating microplastic abundance on the ocean surface observed a 100x reduction of small microplastics (<1 mm) when compared to larger microplastics (1 to 5 mm) (Cózar et al. 2014; Eriksen et al. 2014). While we may suppose nanoplastics are abundant in the marine environment (Andrady 2011; GESAMP 2015) we may not find them on the sea surface in large quantities if other mechanisms of chemical and biological degradation, current dynamics and buoyancy reduce their numbers.

Box 2.1 On bioplastics and biodegradability

Plastics from biomass feedstock

While a great majority of the plastics produced globally are based on non-renewable fossil fuel resources, plastic resins can be made from biomass feedstock as well. There are basically three categories of plastics from renewable biomass resources: a) Biopolymers or bioplastics; b) Bio-derived plastics; and, c) Bio-based plastics. The difference between these categories depends on the role played by the bio-resource in producing the resin.

With biopolymers such as cellulose, chitin or the bacterial copolymer poly hydroxyl butyrate valerate [PHBV], the polymer is created in the form it is available for human use by the plant or the microorganism. The production involves the mere extraction of the plastic from biomass. With bio-derived plastics such as rayon or chitosan, however, the polymer extracted from biomass is chemically converted into a modified polymer that has useful properties in practical applications. Cellulose that is partially acetylated into cellulose acetate for use in cigarette filters or regenerated as cellophane or rayon fibre and deamination of chitin from crab shells into chitosan are examples of such conversions. Bio-based polymers in contrast to the above are man-made polymers using monomers that are derived from biomass. For instance, plant carbohydrates might be fermented into alcohols that can be used to make *bio*-polyethylene. The polyethylene produced is similar in structure and properties to polyethylene made from fossil fuel feedstock and the prefix *bio*-merely indicates the origin of the monomer. With complex monomers, a part of the monomer might be bio-based (with the rest derived from fossil fuel) yielding a partially bio-based plastic.

Biodegradable plastics

Biodegradable polymers are able to undergo degradation into small molecules such as CO_2, CH_4 and H_2O due to the action of biota, usually microorganisms at a rate that is much faster than that for common plastics. The *bio*-prefix in *bio*-PE, *bio*-PET or *bio*-PA does not suggest that these polymers will therefore also be biodegradable. Some biopolymers, bio-derived plastics and bio-based plastics are indeed biodegradable. However, others in the same categories, such as the bio-based plastic *bio*-PE or the *bio*-derived plastic, fully acetylated cellulose, are not biodegradable. Knowing the nature of feedstock used to make the plastic does not allow a determination as to the biodegradability of the material.

Biodegradability and marine environment

The ocean (marine) environment is NOT a disposal environment like composting or anaerobic digestion which are sound end-of-life options for food and bio-waste components of the solid waste stream along with truly and completely biodegradable-compostable plastics. These compostable plastics meet the specification requirements of International Standards and are certified to these standards by independent third party organizations. Several polymer materials are being offered in the marketplace as "marine biodegradable" based on 30°C temperature laboratory scale experiments (ASTM D6691) demonstrating biodegradability. Another ASTM test method measures biodegradability in seawater sediment and the test temperature can be as high as 28°C. However, ocean temperatures drop precipitously as you go down in depth (4°C on reaching 2000 m) and the ocean environment can be much different and less active than the lab test environment. So these marine biodegradable plastics (which show complete biodegradability in a lab test method) could remain in ocean environments for very long periods of time and cause serious environmental impacts that have been recorded for ocean microplastics.

2.3 Sources by sector

2.3.1 Sources in brief

Macroplastics and microplastics entering the ocean come from a wide variety of land- and sea-based sources. Table 2.2 provides a summary of the main sectors identified, the types of plastic products or waste and the typical entry points to the ocean. The source sectors are categorized by producer/consumer responsibility during the lifetime of a plastic product following the flow of plastic through the economy. This is further detailed in Chapter 6 on socio-economic aspects (Figure 6.1). There are significant regional differences in the relative importance of sources and entry points.

Table 2.2 Sources of plastics and microplastics by usage sectors identified in this chapter

Category	Source sector	Description	Entry points	Knowledge
Producers/ Converters	Plastic Producers, Fabricators & Recyclers	Pellets & fragments	Rivers, Coastline, Atmosphere	High
Sectoral consumers	Agriculture	Greenhouse-sheets, pots, pipes, nutrient prills	Rivers, Coastline, Atmosphere	Low
	Fisheries	Fishing gear, packaging	Rivers, Coastline (e.g. ports), Marine	Medium
	Aquaculture	Buoys, lines, nets, PVC pipes	Rivers, Coastline, Marine	Medium
	Construction	EPS, packaging	Rivers, Coastline, Atmosphere	Low
	Terrestrial Transportation	Pellets, tyres, tyre dust	Rivers, Coastline, Atmosphere	Medium
	Shipping/ Offshore industry	Paints, pipes, clothes, miscellaneous, plastic-blasting, cargo	Rivers, Marine	Medium
	Tourism industry	Consumer goods, packaging, microbeads, textile fibres	Rivers, Coastline, Marine	High
	Textile industry	Fibres	Rivers, Coastline, Atmosphere	Low
	Sport	Synthetic turf	Rivers, Coastline, Atmosphere	Low
Individual consumers	Food & drink single-use packaging	Containers, plastic bags, bottles, caps, cups, plates, straws, spoons, etc.	Rivers, Coastline	High
	Cosmetics & personal care products	Microbeads, packaging, toothbrushes, etc.	Rivers, Coastline, Marine	Medium
	Textiles & clothing	Fibres	Rivers, Coastline, Atmosphere, Marine	Medium
Waste management	Solid waste	Unmanaged or poorly managed waste disposal	Rivers, Coastline, Atmosphere	Medium
	Water & wastewater	Microbeads, fragments, fibres	Rivers, Coastline	Medium

2.3.2 *Producers and converters*

Plastic pre-production resin pellets are manufactured and transported to a converting facility where the plastic is compounded and processed into useful products. Whenever transportation of resin pellets occurs there is a potential for accidental losses of pellets, on land and sea. Use of paved surfaces and catch trays for spillage during loading/unloading of rail cars or trucks, and the use of vacuum systems can often help reduce such losses. Once in the converting facility the best practices in processing and clean-up of equipment govern further potential resin loss. The use of storm-drain filters to contain the pellets and observing strict clean-up procedures are generally recommended to limit the loss of pellets at the fabrication facilities.

Although programmes exist to try to prevent loss, pellets are found in freshwater and marine habitats. For example, in sediment samples analysed from European rivers, 18% of the detected microplastic consisted of PS pellets (Karlsson 2015). These pellets showed visual and spectroscopic resemblance to primary pellets/ powder, which was potentially intended for use in polymer production. All samples were taken in rivers that flow nearby polymer plants.

In the manufacture of plastic products and packaging, fragments remain from trimming and tooling processes after typical injection moulding. Seams are ground down, producing microplastics. The tooling of solid blocks of plastic, using drills and milling tools also produce shavings. These fragments can be in all sizes, from the obvious microplastic to nano particles, lost as dust to the atmosphere.

2.3.3 *Land-based sectoral consumers*

Agriculture

There are many potential mechanisms whereby agriculture can be a source of microplastics. For example, plastics are used in agriculture for irrigation and as a mulch. They sit on the field for many months in the sun and when they are removed or disturbed by harvesting or watering can readily break down into microplastics. Runoff from agriculture can transport this material to the marine environment.

Agriculture occupies large areas around the world, but the areas with nutrient-poor soils require high levels of fertilization to maintain this industry. The financial costs or time expenditure associated with the use of fertilizers (nitrogen, phosphate and potassium) have been prohibitively high for some farming situations. One of the newest fertilization technologies, controlled-release fertilizers (CRFs), offers a method for reducing the quantity of fertilizer needed per unit area of cropland, as well as reducing time spent in fertilization efforts (Jacobs 2005). CRFs have advantages for agriculture in reducing cost and in reducing nutrient runoff levels into water systems (Landis et al. 2009), but are introducing a new environmental impact in the form of microplastics contamination.

CRFs encapsulate the N, P, and K nutrient combinations within a coating often composed of a polymer (e.g. polysulfone, polyacrylonitrile and cellulose acetate; Jarosiewicz and Tomaszewska 2003), called a nutrient pill (Landis et al. 2009). The fertilizer diffuses into the soil across this barrier (Gambash et al. 1990) over predetermined time periods (3 to 18 months), offering a continuous nutrient supply to the plant roots. The overall fertilization level required is reduced compared to traditional fertilizers because it reaches the plants as needed over time (Goertz 2000), eliminating the need for over-fertilization, a problem for the surrounding aquatic environment (Carpenter et al. 1998). This should lower the levels of nutrient runoff into water systems from those crops (Sharpley et al. 1994), thus lessening eutrophication that often occurs from these pollution sources (Vollenweider 1968; Vollenweider and Kerekes 1980).

This benefit is not without cost, however, because when nutrients are released, the remaining pill does not degrade. In addition, because the longer release periods are the more commonly desired, the thickness of this polymer layer must be increased proportionally to that intended release duration (Jacobs 2005). Surface runoff due to rainfall events washes soils from agricultural areas into aquatic systems. These plastics will be carried along with those soils and enter both river and estuarine systems along with surface soils. CRF fertilizers are applied either by being mixed into the soil or top dressed (Landis et al. 2009), depending on the particular crop being grown, with top dressed soils particularly at risk for microplastics runoff. Because the quantity of expended pills will increase within soils when these CRF fertilizers are reapplied every 1 to 2 years, this risk will increase over time. The volume of CRF in use in drainage basins and coastal regions, as well as the relative percentages of mixed and top-dressed usage, should be used to estimate the quantity of microplastics being released into aquatic ecosystems per year. Although there are no estimates available to date on the potential of CRFs to contribute to microplastics contamination in the ocean, there is an increasing trend associated with this risk due to the increasing use of fertilizers in agriculture (Heffner 2009).

Construction

Potential discharges related to construction should consider three phases used to describe the life cycle of infrastructure: i) construction, ii) life in service and iii) decommissioning / demolition. Although little information is available on the relative importance of the various entry points from the construction sector into the marine environment, it is clear that construction represents a major use of new plastics, contributing over 20% of annual production in Europe during 2013 (plus plastics used to package items in the construction industry) (PlasticsEurope 2014). This plastic will reach the end of its life and/or become fragmented if not adequately deposited or recycled. Hence there is a considerable reservoir of plastic items within existing constructions and depending on use and management this plastic may be released as microplastics.

Plastic products used in construction should have a long life-in-service in comparison to other applications where products such as single use carrier bags may have been designed to deteriorate on exposure to heat, light and oxygen. Still, there is the potential for emissions of microplastics during the construction phase associated with cleaning, abrasion or grinding. At any stage in the lifetime of a piece of infrastructure, shot blasting with microplastics can be used to clean paint from surfaces prior to further construction or maintenance. If the particles are not contained this could lead to a direct release into the environment either as airborne dust, or soil or water (natural and sewage) contamination.

Insulating foam, typically polyurethane, is often used in construction as a solid board or applied in liquid form inside walls and between ceiling joists. As the foam cures, it balloons out from between wall and ceiling timbers, which is usually trimmed manually with saws. This process produces tremendous amounts of microplastic residue, which are typically mediated by sweeping only.

During construction, components may arrive packaged in single use plastic film, pieces or granules, such as polystyrene. Some of these packaging materials may have been designed to have enhanced rates of degradation, and others may be made of conventional polymer. Unless these packaging items are contained on site and disposed of appropriately (e.g. via recycling)

there is potential for weathering-induced fragmentation of packaging leading to the release of secondary microplastics. The packaging is intended to be a product with a short lifetime, but in some instances may be in place for months or years depending on the duration of the building phase. In addition, the only available means of disposal may be a large container open to prevailing weather conditions. While this may be appropriate to contain heavy construction materials and debris it may not be effective in retaining lighter materials such as microplastics, particularly in windy conditions.

Decommissioning or demolition may also be an emission-source of plastic to the environment. Plastic components of all shapes, sizes, colours and polymers are likely to be distributed throughout a particular construction. Separation, sorting and recycling could therefore be problematic. Even during the recycling process, there is still potential for emissions of microplastics as the result of spillage if products are shredded into small particles. Plastic items and fragments may be released to the environment or become compacted into the substratum of the site.

An additional source of plastic comes from the use of materials in informal shelters and shanty towns. In some regions, such as West Africa, rubbish is used for land reclamation in areas when the local population are without land or conventional housing (UNEP 2016).

Despite these diverse potential sources of microplastics either directly (primary) or indirectly (secondary microplastics) as a consequence of, and at all stages in, construction there are no published studies estimating microplastics generation from this source sector.

Transportation on land

Robust statistical analyses can help identify key loss points and simple, manageable responses to reduce loss. Analysis from a continent wide survey of the Australian coastline suggests that isolated areas may be important sources of plastic pollution through illegal dumping along road networks (Wilcox et al. 2014). Hence, in addition to focusing on major metropolitan areas, considering remote and regional sites is key to understanding loss rates and flows. This can help to target infrastructure and improve success of incentives and enforcement actions to reduce littering and improve packaging materials recovery.

The emission of rubber particle dust (mainly $<$80 micrometre) from tyre wear may be a major source of microparticles contamination to the sea (NEA 2014; Verschoor 2014). Part of the dust flies as particulate matter into the air, the rest lands directly on the road or adjoining land and from there a proportion will enter surface waters or drains. An unknown proportion will be carried to the sea. Car tyres are largely made of styrene-1.3-butadiene rubber (SBR) and recycled products made from tyre rubber. Every year, an estimated quantity of 17,000 tonnes of rubber tyre-wear is released into the Dutch environment (Verschoor et al. 2014). Annual emission estimates of tyre rubber dust for Norway, Sweden and Germany are 4,500, 10,000 and 110,000 tonnes respectively (NEA 2014). Average emissions of car tyre dust for the mentioned countries range between 1 and 1.4 kg/capita/year. Further detailed studies are needed to calculate emissions to the sea and to investigate the input from air transport and atmospheric deposition.

Tourism industry

Tourism is an important economic sector. The World Coast Conference (1993) identified tourism as the world's largest single industry, estimating that it constitutes 5% to 6% of the combined Gross National Product (GNP) of all nations. In addition, tourism has increased over recent years into a global industry, with the World Tourism Organization (WTO) estimating over one billion tourist arrivals across the globe.[4]

Since many popular tourist destinations are coastal (e.g. the Mediterranean is ranked the number one destination by the World Tourism Organization), it is reasonable to assume areas of high tourist activity are important to consider as proxy sources of marine debris. For example, it could be assumed that areas of high tourism are areas of high plastic input simply due to higher concentrations of people. It might be further argued that plastic input is exacerbated since tourists, while away from home, might be more likely to use disposable plastic (e.g. beverage bottles, food containers, etc.) compared to home where they have access to non-plastics. In addition, tourists may be less concerned about environmental impacts in places where they are not living. Conversely, it may be the case that some areas that rely on tourism as a major economic driver, particularly natural environments, are areas where clean-up efforts are more focused and numerous. Increasingly, tourism is spreading to less populated and more 'pristine' environments, where the infrastructure required to deal adequately with waste may be lacking. This is also the case for many Small Island Developing States (SIDS).

2.3.4 *Sea-based sectoral consumers*

Fisheries

Fishing gear may be lost at sea by accident, abandonment or deliberate disposal into the marine environment. Plastic debris resulting from fishing includes nets, traps, lines and ropes, floats, buoys, strapping bands, bait boxes and bags, strings for packaged baits, rubber gloves, galley wastes and household trash (Sheavly 2005). According to Brown et al. (2005), some of the causes related to the disposal of nets at sea are:

- conflict with other sectors, principally towed gear operators;
- working in deep water;
- poor weather conditions and/or on very hard ground;
- very long nets or fleets of nets; and
- using more gear than can be hauled regularly.

[4] http://www.e-unwto.org/doi/pdf/10.18111/9789284416899

The way in which fishing gears are handled may depend on several conditions: fishing area/region, type of fisheries, type and size of the vessel and crew members. Deliberate discarding of fishing gear is also associated with illegal, unreported and unregulated (IUU) fishing.

Artisanal fishers, also known as small scale fishers, have great diversity, and thus there is no single, agreed definition for this subsector (FAO 2015). They are particularly important in developing countries for their contributions to nutrition, food security, sustainable livelihoods and poverty alleviation (FAO 2014). In many populated regions worldwide, besides a poor management of plastic litter along the coast or inadequacy/unavailability of waste disposal/management systems, artisanal fishing may not be adequately regulated. This is either because there is no legislation or policy addressing these issues or the laws or regulations are not updated or enforced. Consequently, artisanal fisheries can be significant sources of ordinary and fishery-related plastics to the sea at local scales. Old structures and fishing gears are also of concern because they easily fragment generating microplastic particles.

According to a FAO report (FAO 2014), the total number of commercial fishing vessels in the world was estimated to be about 4.72 million in 2012. The fleet in Asia was the largest, consisting of 3.23 million vessels accounting for 68% of the global fleet, followed by Africa (16%), Latin America and the Caribbean (8%), North America (2.5%) and Europe (2.3%). From packaging items to food containers on fishing vessels, loss of plastic items overboard may occur. Although there are laws that support the management of plastic litter at sea from vessels (MARPOL Annex V and London Convention and Protocol), there are no known protocols or Standard Operating Procedures (SOPs) to cater to the day-to-day management of litter.

Fisheries management draws on fisheries science for the exploitation of the fishery at a sustainable level. Monitoring, Control and Surveillance (MCS), one of several tools of fisheries management, aims at managing the activities of fishers rather than fisheries. A strong MCS programme has fisheries observers and inspectors who collect data on the activities of vessels from catches and discards to garbage in an effort to support the implementation of regulations and policies to protect the marine environment (Sherif 2014).

Aquaculture

The role of aquaculture in supplying food from the sea and from inland waters is growing. World aquaculture production can be categorized into inland aquaculture and mariculture. Mariculture includes production operations in the sea and intertidal zones as well as those operated with land-based (onshore) production facilities and structures (FAO 2014). According to available statistical information, world food fish production by inland aquaculture and mariculture occupied 42.2% of the total 158 million tonnes of production (capture fisheries and aquacultures) in 2012, increasing from 13.4% in 1990 and 25.7% in 2000 (FAO 2014). See Chapter 5 for further information about this sector.

Studies on the environmental impact of mariculture activities largely focus on eutrophication effects and dissolved contaminants (Gallardi 2014) and rarely examine the types and quantities of lost culture gear. There are some studies reporting on lost or discarded mariculture gear and the resulting contamination of areas with extensive aquaculture (Andréfouët et al. 2014; Bendell 2015) but also areas farther afield (Fujieda and Sasaki 2005; Hinojosa and Thiel 2009; Gago et al. 2014). No quantitative estimates of plastic input from mariculture are available even though locally these inputs can be substantial.

Mariculture structures are either suspended from the sea surface (generally in waters of 10 m to 50 m depth) or in intertidal and shallow subtidal zones where they are placed directly on the bottom. The majority of mariculture activities use lines or cages suspended from buoyant structures, consisting of plastic buoys such as air-filled polypropylene and EPS (expanded polystyrene). These structures also require many lines (mostly non-buoyant plastics) and cages of various types (thin and thick filament net plastics, buoyant or non-buoyant). It is necessary to identify the types of plastics used in these activities and their potential to become sources of microplastic to the marine environment.

Aquaculture gear can be lost for the same reasons as capture fishing gears, e.g. wear and entanglement of structures. However, few studies have reported the cause and amount of loss or gear types (Fujieda and Sasaki 2005; Hinojosa and Thiel 2009; Heo et al. 2013; Liu et al. 2013; Hong et al. 2014; Rani et al. 2014; Al-Odaini et al. 2015). Major losses may be caused by storm events, due to detachment and breakage. In many cases, unused gear is also stored on the shore close to aquaculture centres, and as a result of weathering (e.g. of EPS) large quantities of microplastics may be generated and reach the sea via run-off or wave action, but this has not been quantified. Highly diverse species and consequent methods optimized for target species probably make it difficult to identify sources.

Aquaculture for oysters, mussels and other shellfish that uses EPS buoys has been considered a significant source in the Republic of Korea and Japan (Fujieda and Sasaki 2005; Lee et al. 2013; Jang et al. 2014; Lee et al. 2015). A single EPS buoy can fragment into many thousands of pieces. Most of the plastic used in aquaculture operations is polypropylene, which has a density of 0.9 g/cm^3 (Hidalgo-Ruz et al. 2012) and will float in seawater (assuming an average seawater density of 1.02 g/cm^3), which may mean that subtidal benthic organisms are not ingesting much of the plastic used in the aquaculture infrastructure. However, there is evidence that over time, low density polymers may become fouled and sink (Morét-Ferguson, Law et al. 2010) (Andrady 2011; Morét-Ferguson et al. 2010), in which case these less dense plastics may become available to benthic species. Organisms may also cause destruction of aquaculture structures resulting in fragmentation and generation of microplastics (Davidson 2012). The fraying of plastic-based ropes in close contact with growing mussels may influence the amount of microplastics released compared to other methods with fewer plastic structures (e.g. bottom or rack culture).

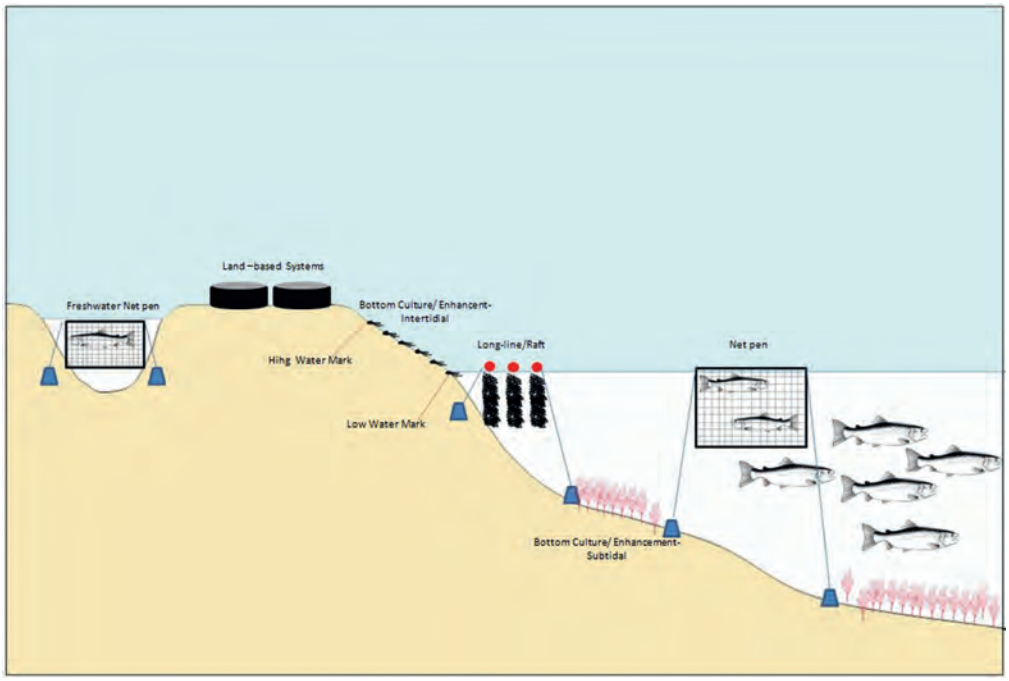


Figure 2.2 *Principal types of aquaculture structures (image courtesy of M. Thiel)*

Nets and cages fragment as result of wear due to fouling organisms, possibly also generating large amounts of debris. Currents and water movement may disperse microplastics and aquaculture gear which would mask the influence of localized microplastic sources and may affect ecosystems (Astudillo et al. 2009).

Efforts to manage and reduce marine debris originating from aquaculture gear have been reported in the Republic of Korea, Taiwan Province of China, Chile and other places (Hinojosa and Thiel 2009; Liu et al. 2013; Lee et al. 2015). However, mariculture methods, gear types, beach pollution, cause of gear loss, impacts, behaviour of gears in major countries in production and consumption should be targeted in the near future. Further research on managing and controlling this debris source is needed. See Chapter 5 for further information.

Shipping and offshore industry

Large shipping vessels with many crew members may carry supplies for several months. They generate solid wastes daily which may end up as marine debris if it is not secured and stored properly (Sheavly 2005). In accordance with amendments to MARPOL Annex V, as of 1 January 2013, all shippers have new responsibilities including the ability for crew to discharge residues and wastewater into the marine environment. Henceforth, shippers will determine whether disposals and wastewater are harmful to the marine environment.

Cargo waste from cargo holds (wire straps, packaging materials, i.e. plastic sheets, boxes) and sewage are among numerous waste items deposited into the marine environment from merchant ships and cruise liners. These items are most often disposed accidentally through bad handling or unfavourable weather conditions. However, waste disposals on many vessels may be handled inadequately either due to inadequate storage facilities on board or lack of reception facilities in ports.

The shipping industry is also regarded as a primary source of microplastics as routine cleaning of ship hulls using plastic abrasives results in high levels of microplastics being released directly into the ocean (Song et al. 2015). Mishandling of cargo or accidental spills are considered to be the main reason why high levels of microplastics have been found in some harbour sediments, particularly resin pellets. Chemical carriers carry the raw materials for plastics manufacture, such as in the form of polymers in solution or as stabilized dimers (a pair of monomers), and it is considered that these could form microplastics following operational or accidental discharge, although there is a lack of data to quantify this source.

Similarly, activities on oil and gas platforms may generate items which are deliberately or accidentally released into the marine environment including hard hats, gloves, storage drums, survey materials and personal waste (Allsopp et al. 2006). Undersea exploration and resource extraction also contribute to marine debris (Sheavly 2005).

Single-use plastics are also used by environmental scientists. Applications include Expendable Bathythermographs (XBTs) for measuring the vertical temperature of the upper ocean, meteorological balloons for measuring the structure of the atmosphere, and passive drifters for measuring water currents.

2.3.5 *Individual consumers*

Food and drink packaging

Around 40% of all plastic production is used for packaging (Figure 2.1). A substantial proportion of this is used to package food and drinks and is abundant as macro-debris in the marine environment, as evidenced in many coastal surveys by regional seas organizations, NGOs and other groups (Table 2.3). Food and drink packaging is widely used for convenience and long-term storage. Because fast-food consumption is often away from home and hence away from domestic waste management, items of fast-food packaging are commonly found as litter. These items of macro-debris are fragmenting in the marine environment and likely a major source of microplastics. Hence managing end of life packaging is of fundamental importance when considering the environment.

Table 2.3 List of top 10 items found by the international coastal clean-up initiative, a programme involving nearly 650,000 volunteers in 92 countries and over 5,500 sites (adapted from Ocean-Conservancy 2013)

Items	**Number of items**
Cigarettes / cigarette filters	2 117 931
Food wrappers / containers	1 140 222
Beverage bottles (plastic)	1 065 171
Plastic bags	1 019 902
Caps / lids	958 893
Cups, plates, forks, knives, spoons	692 767
Straws, stirrers	611 048
Beverage bottles (glass)	521 730
Beverage cans	339 875
Paper bags	298 332

Cosmetics and personal care products Microplastic particles are widely used as abrasive agents and fillers in a wide range of cosmetic products such as facial scrubs and shower gels. These particles will inevitably be released to wastewater systems upon washing or directly into aquatic environments via recreational bathing (Fendall and Sewell 2009). The total quantities of these microplastics (or microbeads as they are known commercially) can be substantial. Napper et al. 2015 estimated that one use of facial exfoliants per day by the UK population could emit to the environment 16 to 86 tonnes of PE microbeads to the environment per year. Since there is no effective way for users to dispose of these plastic particles via solid waste management, most will pass directly into wastewater and potentially the environment. Many of these particles will be captured by sewage treatment facilities. Estimates of the likely capture rate vary but it is considered inevitable that substantial quantities of microbeads pass through sewage treatment into the environment. In many developing regions there is no provision of wastewater treatment (UNEP 2016). Use of microplastics in cosmetics therefore represents a significant direct source of microplastics to the environment. The total quantity of microplastics by weight may be small in relation to macroplastic debris and possibly also in relation to other direct sources of microplastics such as release from car tyres (see section on terrestrial transportation above). However, the use of microplastics in PCPs is potentially avoidable since particles other than plastic can be used as alternatives. The issue has attracted considerable attention from NGOs (e.g. Beat the microbead campaign[5] or Fauna and Flora good scrub guide[6]). Some manufacturers have announced that they will voluntarily phase microbeads out of their products and some regions have introduced legislation to prohibit the use of microbeads in products sold within their jurisdiction. The US passed a federal law in 2015 to ban microbeads in rinse-off personal care products by 2018.

Textile and clothing

Release of fibres from textiles is recognized as a potential large source of microplastic-sized particles. A recent Dutch study found a total average of 2.09×10^8 fibres/m^3 in washing machine effluent and 62% were synthetic fibres (Karlsson 2015). Browne et al. (2011) found that an estimated 1,900 synthetic microfibres were rinsed out of a single piece of clothing. Industrial laundering facilities and Laundromats likely expel microfibres to the atmosphere in unknown quantities. Similar to microbeads from cosmetics, fibres will be carried via wastewater to sewage treatment facilities where a proportion will be removed. However, in many parts of the world, particularly developing countries, the great majority of communities have no sewage treatment capability and microplastic contaminated wastewater is directly discharged in surface waters (Corcoran et al. 2009). The relatively conspicuous nature of fibres compared to other natural particulates might bias their detection in sediment. However, it is still clear that substantial quantities of fibres are accumulating in the environment.

2.3.6 *Waste management*

Solid waste

Unless end-of-life items are managed within a waste stream, it is inevitable that they will contaminate the environment. Waste management options can range from open tips or dumps to landfills, varying levels of incineration, waste to energy and/or recycling. Still, within a waste stream, some material escapes to the environment. For example, when discarded in poorly managed dumps or land fill sites, waste will likely be transported away by winds, and may subsequently enter rivers or the sea. In addition, in some countries, there are coastal dumps where waste is deposited directly on the shoreline and then carried away by the sea (UNEP 1999).

[5] http://www.beatthemicrobead.org
[6] http://www.fauna-flora.org/initiatives/the-good-scrub-guide/

Increasing the extent and improving the quality of waste management is recognized as being one of the most important immediate steps toward reducing inputs of debris to the ocean, particularly in developing countries. This is dependent on having good waste collection systems and infrastructure. Broadly speaking, steps to reduce the amount of waste escaping the waste stream require increasing investment. As a consequence, leaky waste streams are more likely in emerging economies. Recycling is widely regarded as a preferred treatment option within the waste hierarchy (Hopewell et al. 2009). This will enable end of life items to have new value rather than becoming waste, however this will require sophisticated and expensive separation infrastructure. But, it is far preferable to reduce the quantity of plastic entering the waste stream by improved design, reducing useage (especially of single-use packaging) and re-using more durable items where practical. These concepts are further developed in UNEP (2016).

Water and wastewater

Wastewater provides a pathway for solid particles to be transported into aquatic habitats. This includes macroplastics and microplastics. Large, solid items enter the wastewater system with sewage via toilets and can include nappies/diapers, tampons, contraceptives and cotton buds (Tudor et al. 2002). Theoretically these should be removed by primary sewage treatment preventing their entry to the environment. However, during periods of heavy rainfall, the volume of water passing through sewage systems can overwhelm them allowing material to escape into the environment (Williams and Simmons 1997). As a consequence, sewage-related debris is commonly reported in marine litter surveys. Once in the environment, these items of macro-debris have the potential to fragment into smaller pieces and ultimately into microplastics. Reducing sanitary-related plastics requires a combination of education, re-design and infrastructure development (UNEP 2016).

2.4 Entry points to the ocean

2.4.1 Rivers

Microplastics in freshwater ecosystems are increasingly reported, with some available studies suggesting large contamination worldwide. Elucidating sources and pathways of microplastics in freshwater ecosystems will be a major challenge for future research as this information will be the basis for management strategies and reduction measures. Reliable data on concentrations, fluxes and polymer types in continental aquatic environments, including urban water systems, are still needed as freshwater ecosystems have received far less attention despite the fact that the majority of plastic litter is being produced onshore and introduced into marine environments by rivers.

Some studies report not only the presence of microplastics in freshwater ecosystems, but show that contamination is as severe as in the oceans (Dris et al. 2015). In these continental waters, microplastics have been observed in both sediments (predominantly lakeshores but also riverbanks) and water samples (predominantly surface water of lakes and rivers).

Both primary and secondary microplastics can enter the continental aquatic environment through several pathways. The debris enters aquatic systems directly by water run-off or via storm water and wastewater treatment plant (WWTP) outlets. For example, granulated polyethylene (PE), polypropylene (PP) or polystyrene (PS) particles, used for example in skin cleaners, can be introduced into wastewater (Gregory 1996). Furthermore, laundry washing machines discharge a large amount of plastic fibres into wastewater (Browne 2015; Karlsson 2015). Industrial or agricultural (Rillig 2012) activities also contribute to the amount of microplastics in freshwater/aquatic ecosystems. High amounts of microplastic particles and fibres have also been detected in the vicinity of industrial plants involved in paper production (Dubaish and Liebezeit 2013). Primary microplastics and synthetic fibres are also known to contaminate sewage sludge (Zubris and Richards 2005). These can runoff with storm water and enter freshwater habitats. Generally, studies indicate spatial associations between the types of microplastics found and human activities (Eerkes-Medrano et al. 2015).

The nature, composition or relative abundance of the microplastic material can sometimes aid in its identification. For example, raw plastic (pellets and flakes) was found in the Danube, a river that has plastic production sites adjacent to it (Lechner et al. 2014). Moreover, resin pellets and microbeads were most abundant in the industrial region of Lake Huron and the densely populated and industrial Lake Erie (Eriksen et al. 2013). The lack of primary pellets but an abundance of secondary fragments on the shores of the sparsely populated mountain lakes (Garda and Hovsgol) suggested an origin from the breakdown of household items (Eerkes-Medrano et al. 2015). Finally, McCormick et al. (2014) demonstrated increases in concentrations of primary microplastics, up to 9.2 to 17.93 times, downstream from a wastewater treatment plant.

Initial freshwater studies are finding that similar physical, chemical and biological factors to those suggested for marine systems contribute to microplastic transport and dispersal, including flow velocity, water depth, substrate type, bottom topography, and seasonal variability of water flows (Eerkes-Medrano et al. 2015). Factors that may have a temporal aspect include storms, floods, or anthropogenic activity. In estuaries, however, microplastic abundance has also received attention (Browne et al. 2010; Sadri and Thompson 2014; Zhao et al. 2014), but given the strong influence of salinity gradients and tidal movements in these systems, it remains difficult to understand local partitioning, the role played by the freshwater inputs and the degree to which estuaries may represent ‘hot-spots’ of accumulation. As rivers have shown to be a significant pathway of microplastics to the ocean, these relationships are important to further understand source pathways and potential remediation that can be taken to avoid release of microplastics.

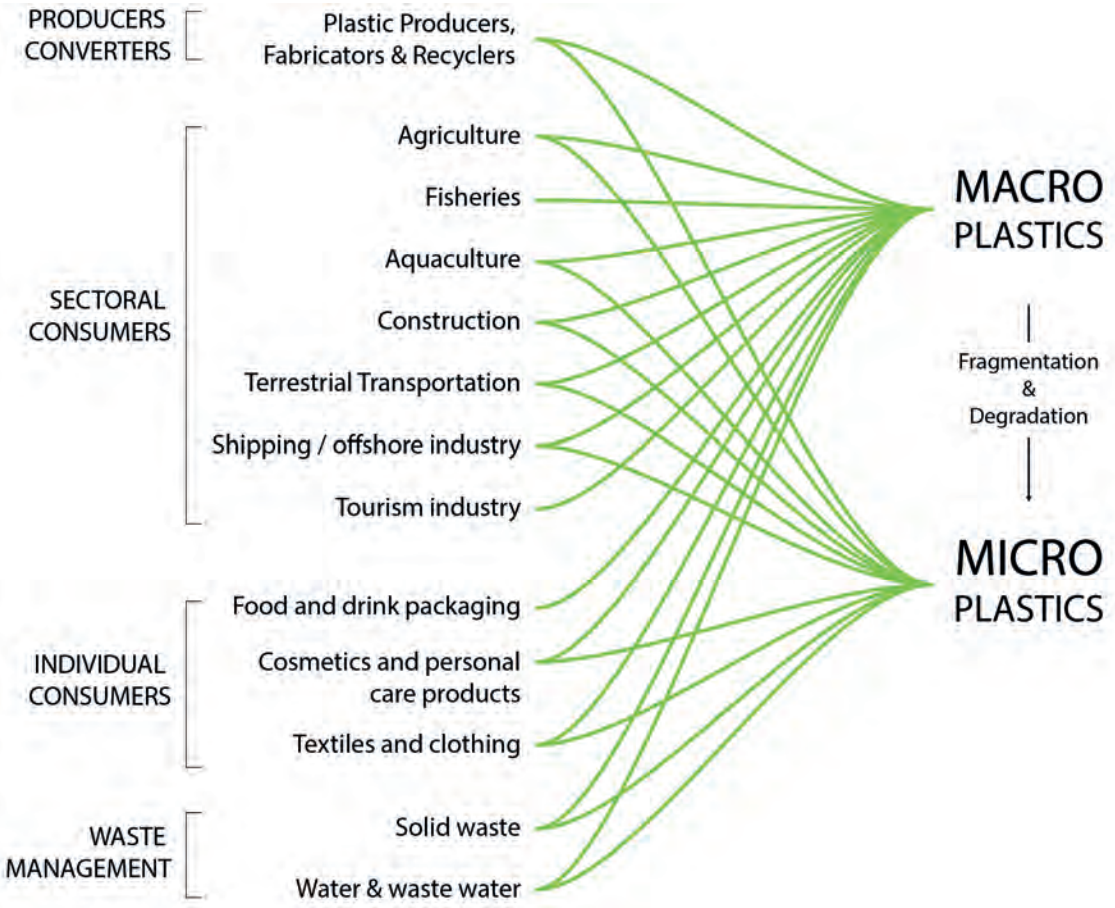


Figure 2.3 *Potential sources of microplastics to the marine environment entering via rivers*

Concentrations of microplastics reported for rivers (Table 2.4), are highly variable (up to a factor of 10^9; Dris et al. 2015), likely due to the different methodologies used but also because of converging currents, proximity to sources and the downstream location from cities. In addition each set of measurements represents a 'snap-shot' and the total flux of particles averaged over a representative time period is very difficult to estimate by measurements alone.

Lebreton et al. (2012) used an ocean circulation model coupled to a Lagangian particle-tracking model to simulate the input, transport and accumulation of marine debris over a 30-year period. A total of 9.6 million particles were released with inputs dependent on three proxies: coastal population density, impervious surface layer and shipping density, using the data layers estimated by Halpern et al. (2008). A modelled particle release distribution from riverine inputs is compared to the surface water data (Table 2.3) in Figure 2.4 (when reported in particles per cubic metres and when using similar measuring methods from 333 µm to ~1 mm neuston net's mesh size). The modelled riverine input represents urban development pressure on rivers. Best fit between measured microplastic particles in the surface water per day and modelled particle release rate was found for $y = 1.8207e^{0.1135x}$ ($R^2 = 0.75$). Using this relation, we estimate a discharge of >60 billion particles entering the ocean from rivers every day. Clearly, there is significant uncertainty in such estimates but they can be useful to indicate the relative importance of different sources and help to direct further research and possible mitigation measures.

While this assessment shows how proxies can be a useful tool in providing global estimates for a given input scenario, caution must be taken with this particular estimate as it is based on a handful of rivers (n=10), mainly in Europe and South America. Also, only surface water data was taken into account and suspended particles are therefore omitted. In that sense, this estimate is rather conservative.

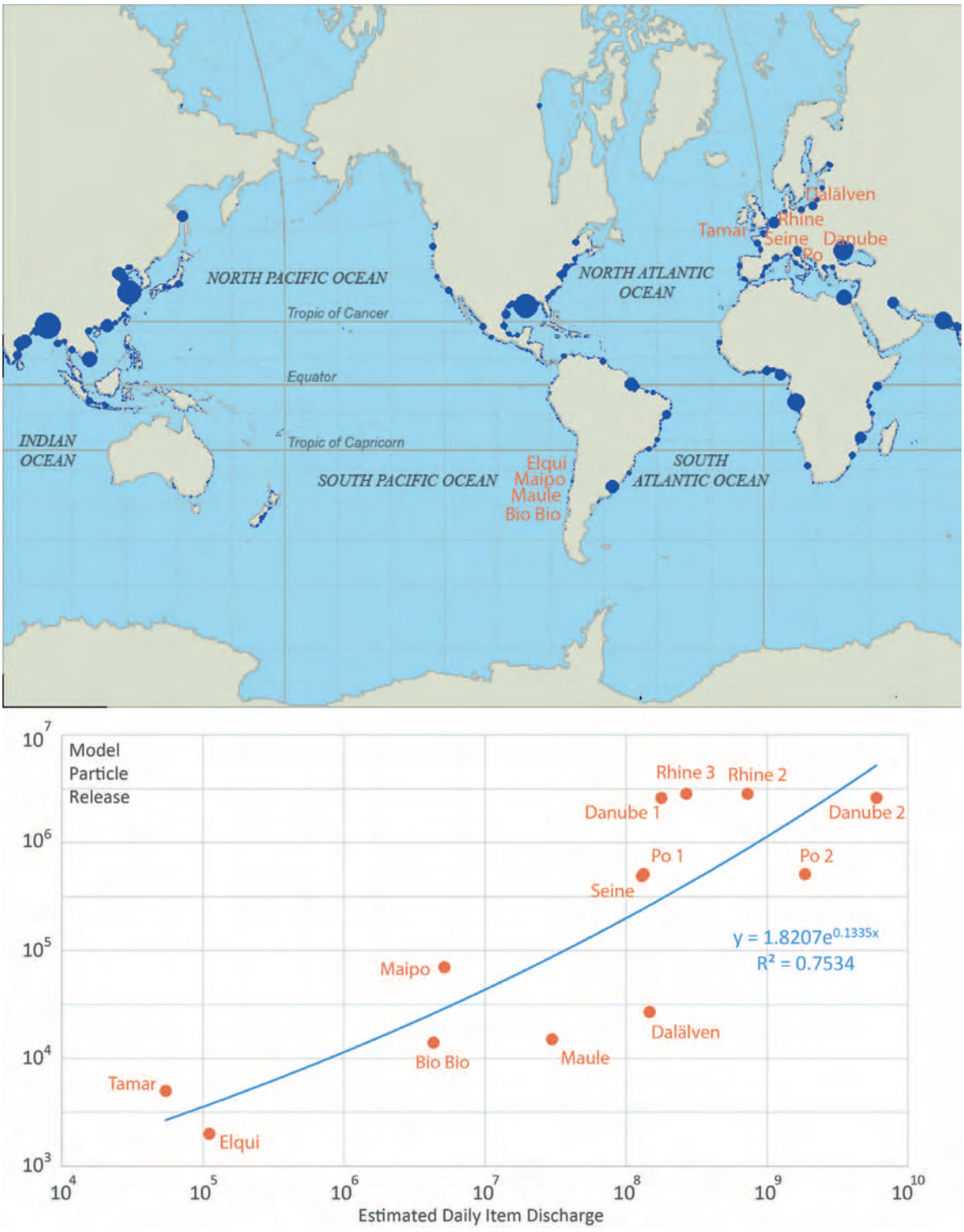


Figure 2.4 *Upper panel – comparison between estimated (measured) microplastic densities at the surface waters of rivers in Europe and South America and modelled global riverine input distribution (Lebreton et al. 2012). The distribution (bottom) is adapted from proxy data on urban runoff computed from total impervious surface area per watershed (Halpern et al. 2008)*

Table 2.4 Measured microplastic contamination on the surface waters and sediment of various rivers

Location	Compartment	Sampling	Abundance (densities)	References
Danube River, Austria, Europe	Surface waters	Size classes: <2 mm, 2–20 mm Sampling mesh: 500 mm	Max: 141 647.7 items/1000 m^3, Mean: 316.8 (±4664.6) items/1000 m^3 73.9% represent spherules (~3 mm)	Lechner et al. 2014
Solent, Hamble, Itchen and Test Rivers, UK, Europe	Surface waters	1235 (total of 4 samples) sampled in each estuary. 0.3 mm mesh	Itchen 1,55 mp/m^2 Test 5,86/m^2 Hamble 0,4 mp/m^2 Total all estuaries: 3,72/m^2 (Southampton water 1,29/m^2)	Gallagher et al. 2015
Tamar estuary, UK, Europe	Surface waters	Size classes: <1 mm, 1e3 mm, 3e5 mm, >5 mm Sampling mesh: 300 mm	Max: 204 pieces of suspected plastic Mean: 0.028 items/m^3 Abundances include all plastic particles, of which 82% represents size <5 mm	Sadri and Thompson 2014
Los Angeles River, San Gabriel River, Coyote Creek, USA, North America	Surface, mid- and near-bottom water	Size classes: >¼1.0 and <4.75 mm, >¼4.75 mm Sampling mesh: 333, 500, and 800 mm	Max: 12 932 items/m^3 Mean 24-h particle counts on date of greatest abundance: Coyote creek: 4999.71 items/m^3 San Gabriel river: 51 603.00 items/m^3 Los Angeles River: 1 146 418.36 items/m^3 Item size class: 1.0–4.75 mm	Moore et al. 2011
Elqui, Maipo, Maule and BioBio Rivers, Chile, South America	Surface waters	neuston net with a mesh size of 1 mm and an opening area of 27 * 10.5 cm^2.	Elqui Mouth: 0,12875/m^3 Maipo: 0,647/m^3 Maule: 0,74 /m^3 BioBio: 0,05/m^3	Rech et al. 2015
Po River, Italy, Europe	Surface waters	neuston net (330µm), Monthly	1 (Spring) to 12.2 items/m^3 (winter)	Vianello et al. 2013
Seine River, France, Italy	Surface waters	a plankton net (80-mm mesh), and a manta trawl (330-mm mesh)	(i) plankton net: 3 to 108 particles/m^3. (ii) manta trawl : 0.28–0.47particles/m^3	Dris et al. 2015
Nakdong River / Jinhae Bay, South Korea, Asia	Surface waters	Trapping of surface water, 2mm mesh screen, 100 times, 3.14 m^2 /2.2–2.8 L. samples/station	120 000 particles/m^3 (10% paints), 187 000 ±207 000 particles/m^3 after heavy rain	Song et al. 2015
North Shore Channel, Chicago, USA, North America	Surface waters	two neuston nets (0.92 × 0.42 m and 0.36 × 0.41 m) of 333-µm mesh	Upstream waters: 1.94± 0.81 particles/m^3 Downstream waters: 17.93± 11.05 particles/m^3.	McCormick et al. 2014
Elbe, Mosel, Neckar and Rhine Rivers, Germany, Europe	Sediment	Size classes: <5 mm	Max: 64 items/kg dry weight, Mean: not indicated	Wagner et al. 2014
St Lawrence River, Canada/USA, North America	Sediment	Size classes: not indicated. Items size range: 0.4 to 2.16 mm	Mean: 13 759 (±13 685) items/m^2, Max: 136 926 (±83 947) items/m^2.	Castañeda et al. 2014
Rhine, Germany, Europe	Sediment	63–5000 µm Three size classes: 630–5000, 200–630, and 63–200 µm	Range: 228–3 763 particles/kg	Klein et al. 2015
Yangtze Estuary, China, Asia	Surface	pumping/filtration (32-µm steel sieve)	4 137.3 ± 2 461.5 particles /m^3	Zhao et al. 2014

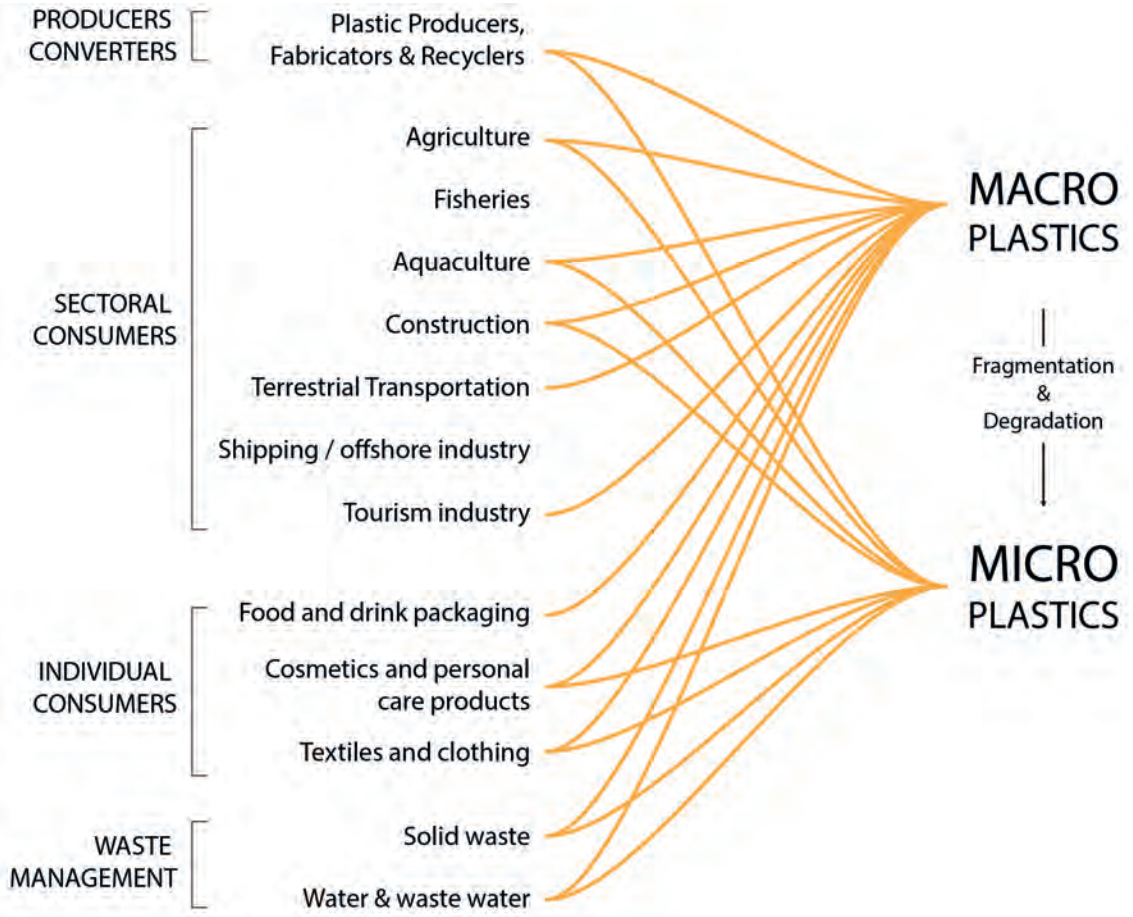


Figure 2.5 *Potential sources of microplastics to the marine environment entering via coastlines*

2.4.2 *Coastline*

Early estimates from the US National Academy of Science claim that a total of 5.8 million tonnes (6.4 million short tons) of waste are released into the ocean every year and of this 0.7% is plastic, roughly 41,000 metric tons (NAS 1975). A careful reading of this reference suggests that this number is based on an extrapolation of values from estimates of wastes produced by individual households. These inferences may not be entirely accurate. More recently, a study calculating the amount of mismanaged plastic waste generated by coastal populations worldwide estimated that 4.8 to 12.7 million tonnes can potentially enter the ocean as marine debris (Jambeck et al. 2015). A summary of common sources of macro- and microplastics from coastlines is provided in Figure 2.5. An additional source that is regionally important is shipbreaking directly on the shoreline, such as in India and Bangladesh (Reddy et al. 2006).

The framework proposed by Jambeck et al. (2015) integrates data on solid waste, population density and economic status for 192 coastal countries. The annual amount of mismanaged plastic waste generated by populations living within 50 km of the coast was estimated at 31.9 million metric tons per year. Mismanaged waste was defined as 'material that is either littered or inadequately disposed. Inadequately disposed is not formally managed and includes disposal in dumps or open, uncontrolled landfills, where it is not properly contained' (Jambeck et al. 2015). It should be noted that informal waste picking appears to be included in the mismanaged waste category. However, waste picking forms an extremely important social and economic role in India[7] (Sharholy et al. 2008) and parts of Asia and undoubtedly reduces the quantities of plastic from reaching the ocean. The study predicts an order of magnitude increase in marine littering from coastal population pressure by 2025 if no improvements are made on waste management infrastructure. The work also suggests that 83% of the global mismanaged plastic waste in coastal regions for 2010 was generated by 20 countries, a list dominated by Asian countries (11 countries in the top 20) with China ranking first (1.32 to 3.53 million metric tons of annual plastic debris input) and Indonesia second (0.48 to 1.29 million metric tons). An unquantified proportion of the plastic waste encountered in waste in countries in Asia and west Africa originates from more developed countries, especially in North America and western Europe. This is as a result of both the legal and illegal trade in packaging/construction plastics as well as plastics associated with electronics goods.

It should be noted, however, that the estimate in Jambeck et al. (2015) relies on a conversion rate of 15% to 40% from mismanaged plastic waste on land to potential plastic marine pollution. The conversion

[7] http://www.theguardian.com/global-development-professionals-network/2014/jul/01/india-waste-picking-women-waste-cities-urban

rate is based on municipal water quality data from the San Francisco watershed in California. In order to refine global input estimates from coastal locations, the conversion rate from mismanaged plastic on land to floating marine debris should ideally take into account regional/local social and economic factors as well as site-specific coastal environment and contextual data such as land use, coastal morphology, shoreline substrate, precipitation rates, wind, wave or tidal circulation. It is for instance unknown from the initial estimate of total input to the ocean, what percentage is actually washing ashore soon after leaving land. There is a need for refining the general understanding of coastal dynamics for marine debris, particularly episodes of stranding and release.

Extreme events such as storms, storm surges and tsunamis are also a significant immediate source of land-based plastic debris (Thiel and Haye 2006). A well-documented example is the pulse of debris washed into the North Pacific by the 2011 Tohoku tsunami (Lebreton and Borrero 2013; Maximenko and Hafner 2014).

2.4.3 *Marine*

Plastic litter originating from marine sources is generated from all types of boats, ships and offshore platforms by accidental loss, indiscriminate littering or illegal disposals (Allsopp et al. 2006). The occurrence of compounded plastics in the open ocean is most probably due to the routine solid waste disposals by individual ships (Colton et al. 1974). Fishing and aquaculture activities may also add large amounts of plastics into the ocean. The recent increase in population along the coast globally and the accessibility of nylon netting, monofilament fragments for fishing and other purposes, have substantially become a cause of plastic litter generation (Bourne 1977).

Numerical modelling assessment of marine debris dispersal originating from shipping activity is reviewed in Lebreton et al. (2012). The framework uses global shipping line frequency as a proxy for model particle release distribution (Figure 2.7). No numerical modelling studies investigating the contribution from fishing and aquaculture industries to the marine debris issue on a global scale have been proposed to date. Estimated distribution of fishing effort derived from catch statistics and fleets location (Watson et al. 2013) could be used for particle model source distribution. The study on global fishing effort shows that international fleets now fish all of the world's oceans and have increased in power by an average of 10-fold (25-fold for Asia) since the 1950s. In regard to fish and shellfish farming, however, while aquaculture and mariculture production for individual countries is well documented no quantitative or qualitative distribution of aquaculture activity on a global scale has been proposed to date.

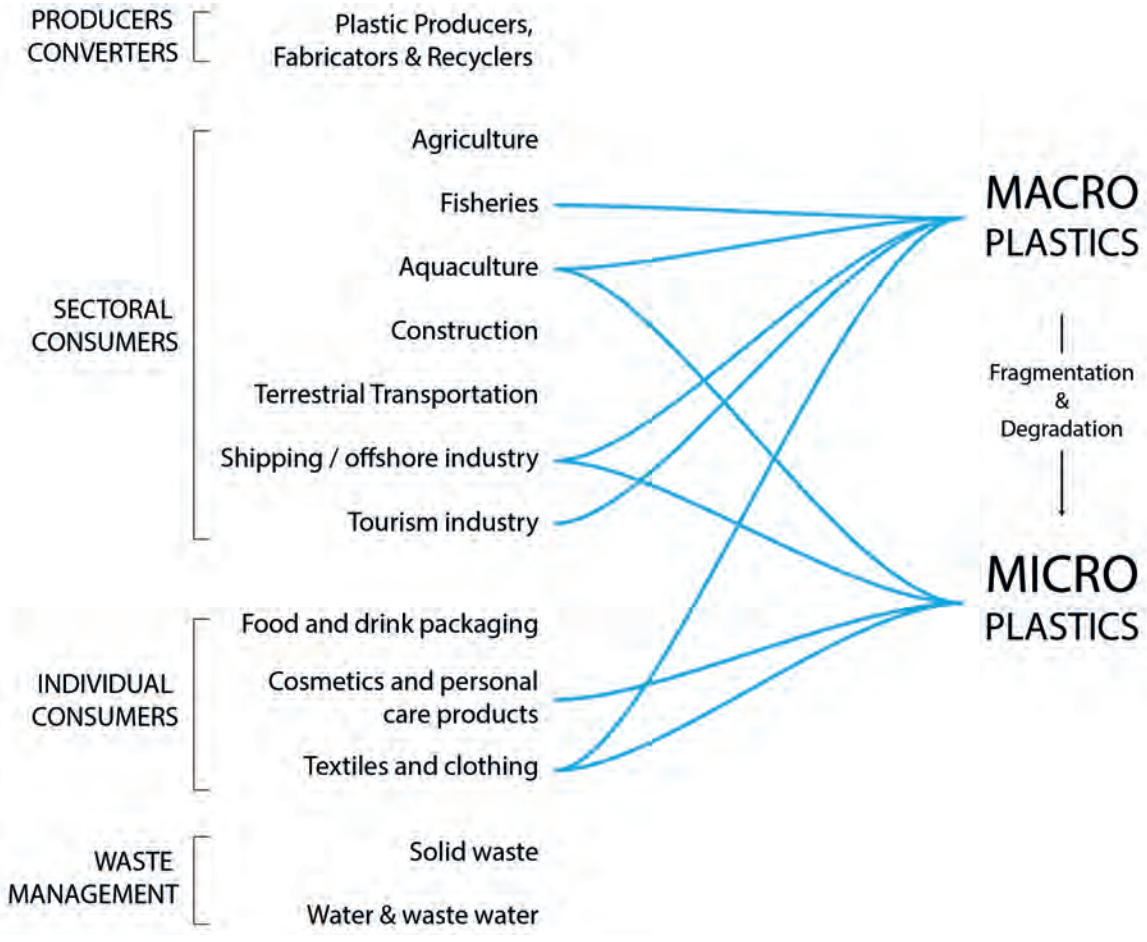


Figure 2.6 *Potential sources of microplastics directly to the marine environment*

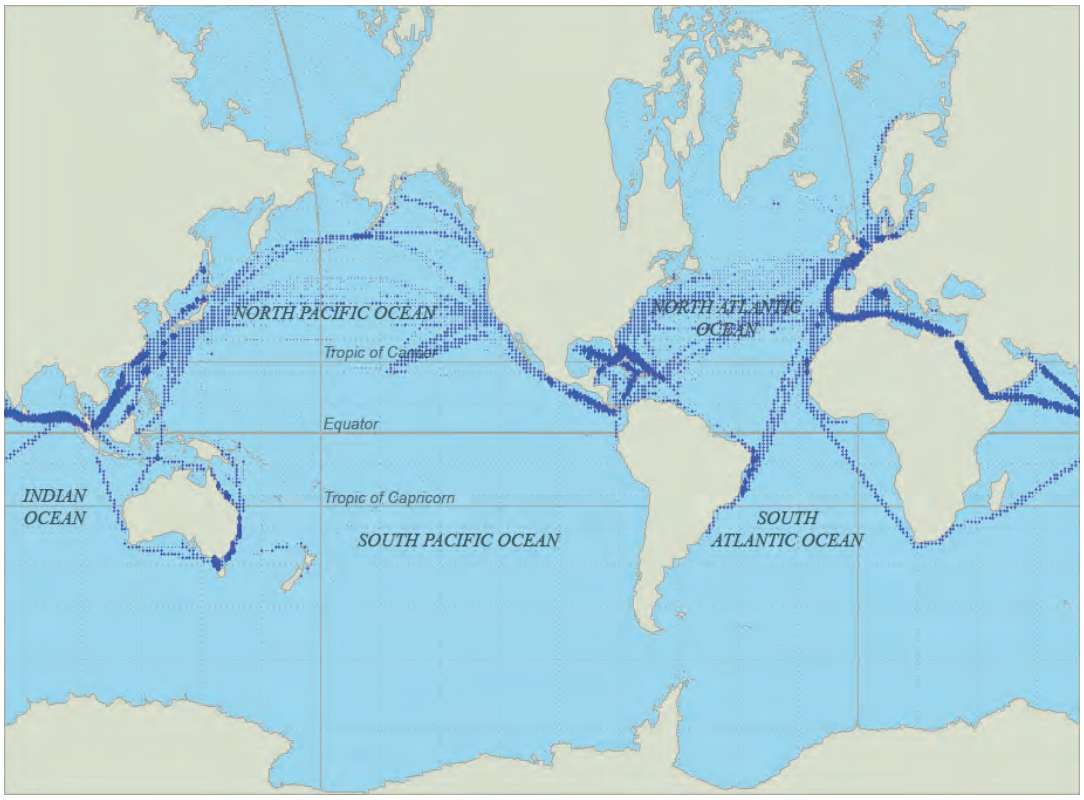


Figure 2.7 *Model source distribution for maritime traffic scenario based on major shipping lanes (image courtesy of L. Lebreton)*

2.4.4 *Atmosphere*

Atmospheric aerosol particles, defined as natural and anthropogenic solid or liquid droplets suspended in the atmosphere, may have sizes ranging from a few nanometres in diameter to several tens of micrometres (Pryor et al. 2015) and include primary anthropogenic aerosol particles derived principally from fuel combustion and industrial processes, as well as synthetic fibres (Dris et al. 2015).

Since plastic fragments are transported by the wind, this must be also the case for microplastics, and atmospheric inputs of microplastics cannot be ignored. In Lake Hovsgol, a remote mountain lake in Mongolia, an average density of 20,264 particles/km^2 (997 to 44,435 particles/km^2, min-max values) was observed (Free et al. 2014), indicating a significant contamination for a remote non-densely populated area attributed to aerial transfer from distant urban sources.

In a recent experiment (Dris et al. 2015, 2016), total atmospheric fallout (wet and dry) was collected through a funnel during a 3-month period, at various frequencies, to better understand fluxes of microplastics to the watershed of the Seine river in Paris (France). Microplastics were observed with fibres being 90% of the total number. Half of the fibres were longer than 1000 µm. Microplastic fallout ranged from 29 to 280 (average 118) particles/m^2/day. The lowest fallout was measured during dry periods and the highest fluxes were measured during periods of daily rainfall.

Micro-particles in the ocean surface can be scavenged by bubbles and re-suspended in the atmosphere when the bubbles burst. This allows transport in the atmosphere before being redeposited in the sea.

The atmosphere is an important pathway by which many natural and anthropogenic materials are transported from the continent to the ocean and also because of the low density of some polymers; fallout of plastic particles directly from the atmosphere or indirectly through rivers and watersheds seems to be far from negligible.

2.5 Scale variability

2.5.1 *Time-scale-dependency*

Most current models and estimates of litter quantities and distributions in the oceans consider continuous input of litter into the ocean and meso- to large-scale oceanographic models distributing plastic litter within the oceans. However, both entry and dispersal of litter in the oceans can be highly variable on temporal and spatial scales, which is important to keep in mind when evaluating ecological and economic risks and when designing preventative measures.

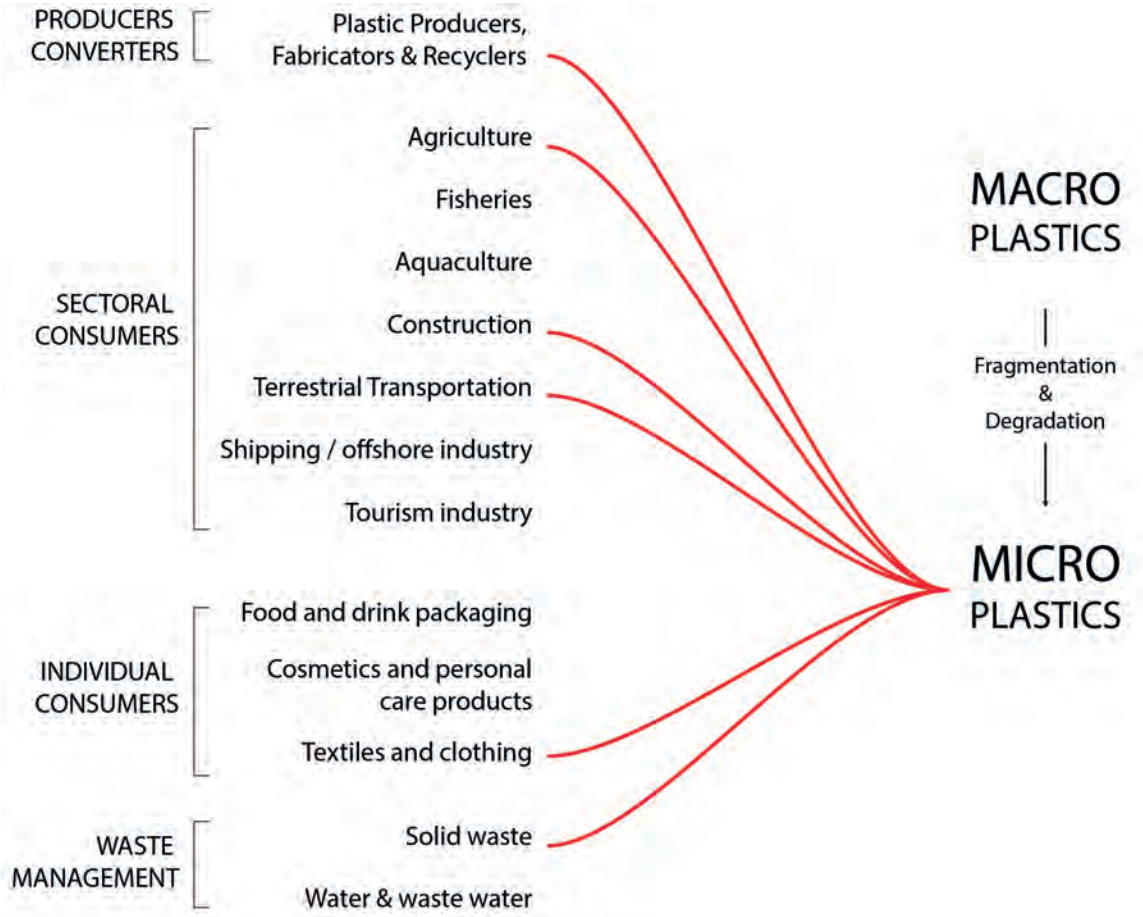


Figure 2.8 *Potential sources of microplastics to the marine environment entering via the atmosphere*

Plastic litter from many sources reaches the ocean on a near-continuous basis. For example, domestic and riverine litter can be considered chronic sources (Jambeck et al. 2015), similar to litter from shipping or fishing (see above). But, in many regions this is an over-simplification. Riverine flows are highly episodic even in temperate zones, varying over hours, days, seasonally and multi-year, and can deliver substantial quantities of litter to the ocean during high flow events (Moore et al. 2011; Carson et al. 2013; Rech et al. 2014, 2015). In tropical and sub-tropical regions, seasonal monsoons can flush out otherwise stagnant waterways. In subarctic regions river flows will be highest during the spring thaw. However, there is limited information to quantify the impact of these variable inputs. During major episodic (catastrophic) events, such as large-scale river basin or coastal flooding, major storms (hurricanes, cyclones, typhoons) and tsunamis, very large amounts of litter may be delivered to the oceans in a short period (Thiel and Haye 2006). The recent 2011 tsunami in Japan is the first catastrophic event that has spurred systematic research efforts in quantifying and tracking plastic (and other) litter introduced to the oceans (Bagulayan et al. 2012; Lebreton and Borrero 2013; Calder et al. 2014).

Currently, little information is available about the proportions of plastic litter that enter the oceans via chronic and catastrophic sources. The frequency, quantification and impact of litter introduced to the oceans by these catastrophic events (flood events, cyclones, tsunamis) deserve more research attention. As a first step, it would be valuable to map regions in the world that are subject to catastrophic events (e.g. tsunamis along the Pacific Rim, cyclones at subtropical latitudes).

2.5.2 *Regional scale dependency*

It is important to realize that the quantities of macro- and microplastics entering the oceans from the sources described above may vary considerably from location to location. This may be due to the relative importance of different sectors, the adequacy of waste collection and management, and a whole series of environmental, social and economic factors. These differences persist despite increasing globalization of trade and movement of people.

For example, although coastal tourism is now a global phenomenon there are regions and countries where it represents a relatively larger contributor to the local economy and social welfare compared with other sectors. This can result in an increase in both the pressure from marine litter and the local socio-economic impact. Such areas include SIDS and developing countries where a lack of adequate waste collection and management can exacerbate the problem.

On smaller spatial scales, sources of microplastic can vary from country to country and even municipality to municipality. As industry footprints and waste management differs, so will the amount and contributions of different types of microplastic litter. This means that efforts to mitigate microplastic waste entry into the

ocean will need to be tailored to local circumstances, while recognizing that there will be commonalities in the types of waste being produced.

Seasonal environmental factors can affect both the generation of waste (e.g. seasonal fisheries, coastal tourism) and the transport of plastic to the ocean (seasonal storms and flooding events). These variables also have a spatial element.

2.6 Conclusions, knowledge gaps and priorities

2.6.1 *Conclusions*

This chapter documents a series of studies assessing the magnitude of microplastic contamination in the marine environment by identified source sectors. Evidence of the generation of primary or secondary microplastics has been reported at every level of a plastic product's lifecycle, from both diffuse and point sources. Plastics enter the ocean via freshwater systems, wastewater run-offs and littering around the coastline, losses or discards at sea and atmospheric transport.

While source sectors can be identified, there is a considerable lack of data quantifying the scale of the issue. Quantified results are reported for a relatively small number of case studies across sectors worldwide with highly varying methods and contexts. Accurately quantifying the various sources of microplastics represents an important challenge for future research, as it would require internationally coordinated monitoring campaigns on identified sectors. Understanding the variability of microplastics inputs into the ocean over space and time is an additional challenge. Unfortunately, methods of defining microplastics, sampling, and interpreting patterns in space or time vary considerably among studies, yet if data could be synthesized across studies, a global picture of the problem may be available (see Chapter 7). A thorough mapping effort quantifying the loss of macro- and microplastics would help in predicting source estimates at regional or global scale and would assist in implementing policies and regulations. Models are very useful at augmenting gaps in observations and in running scenarios, although at some stage they need to be validated and tested against observations. Further model development would be helpful.

2.6.2 *Knowledge gaps*

On land, despite several identified sources of microplastics in the construction sector, there are no published studies estimating microplastics generation from construction sites. In agriculture, no estimates are available on the potential of CRFs to contribute to microplastics contamination in the ocean. More detailed studies are required to calculate emissions from terrestrial transportation to the sea. There are very few large-scale programmes measuring occurrence of debris around coastline and most peer-reviewed scientific studies describe local patterns.

At sea, studies on the environmental impact of mariculture activities largely focus on eutrophication effects and dissolved contaminants and rarely examine the types and quantities of lost culture gear. No quantitative estimates of plastic input from fishing and mariculture are available even though locally these inputs can be substantial.

Reliable data on concentrations, fluxes and polymer types in continental aquatic environments, including urban water systems, are still needed as freshwater ecosystems have received far less attention despite the majority of plastic litter being produced onshore and introduced into marine environments by rivers. This needs to include the total load and not simply that floating on the surface.

Currently, little information is available about the proportions of plastic litter that enter the oceans via chronic and catastrophic sources, in particular quantitative estimates about catastrophic events are lacking.

There is a need to further improve the availability and reliability of models to cover various aspects of sources, transport, fate and effects.

A thorough analysis of the informal waste management activities in developing countries to mitigate plastic needs to be assessed in order to refine estimates of plastic waste leaving shorelines globally. The industry of waste picking, an informal form of waste management, is capable of removing a significant volume of mismanaged plastic from the ground before it enters the ocean. Waste picking activities also transfer, by burning or informal dumping, valueless plastics into the environment.

The behaviour of micro and nanoplastics in seawater and the pathway to sedimentation needs further analysis, in terms of buoyancy relative to sea state, biofouling and transportation during vertical descent.

2.6.3 *Research priorities*

- Encourage the effective and open exchange of best practice (sampling and analysis, harmonization) and data on the distribution, fate and effects of marine litter, to encourage cost-effective and integrated monitoring, assessment and management strategies.
- Identify leakage points of plastic debris to the ocean, including the influence of the globalized trade in waste plastic.
- Quantify release from industry (spills during production, transport, and incidents)
- Quantify the contribution of the sources of microplastics to oceans (from macro debris to car tyre dust, textile and netting fibres and microbeads).
- Better understand the sources and fate of fibres and nanoplastics.
- Establish accurate estimates of fluxes from point sources.
- Identify local waste streams before and after entry points, e.g. wastewater influents vs. effluents.
- Measure efficiency of interventions at the source.

- Quantify sources of microplastics from atmospheric depositions.
- Improve hindcasting, i.e. where did plastic come from?
- Develop and implement monitoring systems in river catchments and wastewater outfalls.
- Improve repository statistics.
- Understand stakeholder responsibilities for marine litter and incentives for taking action.

3 DISTRIBUTION, FATE AND 'HOT-SPOTS'

Key points

1. Microplastics movement is complex and driven by many factors including winds, buoyancy (plastics properties), biofouling, polymer type, size and shape, local and large-scale currents and wave action.
2. Microplastics are distributed between the ocean surface, the water column, the seafloor, the shoreline and in biota. Understanding fluxes of microplastics and hot-spots of microplastics distribution requires understanding movement between these compartments.
3. Physical, chemical and biological processes acting on the microplastics within each reservoir or compartment differ, and in most cases are poorly quantified.
4. Harmonizing the multiple existing approaches to sampling, measuring and quantifying microplastics will improve local, regional and global understanding and support much-needed, large-scale syntheses.

3.1 Lessons from the first assessment

The first report identified key components about the complex plastics issue that are needed to make an accurate assessment of the transport, distribution and fate of microplastics in the ocean. The report also identified the need to identify 'hot-spots' for microplastics in the ocean and the complex nature of such assessments. These complexities are due to several factors. As an example, it is thought that microplastics are present throughout the ocean and are distributed both horizontally and vertically in the water column. Thus, it is very difficult (and may not be possible) to detect the full size spectra of microplastics in situ on a large scale, and thus there are very few direct measurements. The first GESAMP report highlighted the utility of numerical modelling as a tool to predict (or hindcast) the location of microplastics given an estimated source (or observed final location). There are several issues regarding modelling approaches which need to be considered, including uncertainties around the age of plastics (how long they are in the water column vs. on shore, for example), how particles change density while being transported by ocean currents, how plastic degrades over time (primary vs. secondary sources), unknown rates of biological transportation, coupling coastal and open ocean hydrodynamics, and integrating 3D circulation (the plastic loop) with temporary and permanent deposits (e.g. sedimentation and resuspension). However, numerical modelling to predict and hindcast microplastics is a valuable tool which has strong merit.

In this chapter, we describe further information regarding such topics through discussion of the different compartments in the ocean where there is contamination from microplastics and the transport and flux between compartments.

3.2 Microplastics in ocean compartments

3.2.1 *Compartments in brief*

Microplastics are distributed between five main ocean compartments: i) on or near the ocean surface (including the upper layers mixed by wave action); ii) in the water column; iii) on the seafloor; iv) on the shoreline, including buried in intertidal sediments; and, v) in biota (Figure 3.1). In addition, microplastics may be found in the atmosphere-ocean interface. Microplastics are transferred both between and within these compartments, although the processes involved are poorly understood. The physical, chemical and biological processes acting on the microplastics within each reservoir will differ. Consequently, the risks and opportunities for mitigation might also be different.

With the exception of perhaps the surface ocean, there is a severe paucity in data on the amount of plastic in each compartment, and there is even less known about the fluxes of microplastic between compartments. Closing the global microplastic budget will require large-scale, targeted sampling and modelling of all of the compartments.

3.2.2 *Microplastics on the ocean surface*

Of the compartments, the surface ocean is probably the best sampled (see tables 10.1 and 10.3 of Lusher, 2015). Decades of extensive trawling data (Law et al. 2010, 2014; Cózar et al. 2014; Eriksen et al. 2013, 2014) have recently been combined into a global data set of more than 11,000 trawls (van Sebille et al. 2015). While coverage of this data set is still strongly biased towards some regions such as the North Pacific and North Atlantic, this data set reveals clear patterns of microplastic abundance. These studies, while different in their approaches, all come to a global estimate of the microplastic abundance of anywhere between 5 and 50 trillion particles, at a mass of 32,000 to 236,000 metric tonnes (van Sebille et al. 2015). Microplastics have also been observed in some of the most remote marine environments, including surface waters of the Arctic (Lusher et al. 2015), Arctic sea ice (Obbard et al. 2014) and in the Southern Ocean (Barnes et al. 2010).

Approximately half of the floating microplastic in the open ocean resides in the subtropical gyres of the North and South Atlantic, North and South Pacific and the Indian Ocean, where abundances can be a million times higher than in other regions such as the tropical Pacific and Southern Oceans. High concentrations of microplastics are also found in some areas of highly populated marginal seas such as the Mediterranean Sea, which is characterized by an anti-estuarine circulation (Cózar et al. 2014). First order, physical oceanographic understanding, including Ekman theory, can

explain these patterns (see Section 3.3) with microplastic accumulating in areas where large-scale winds cause convergence of the surface flow (van Sebille et al. 2015). After some time in the gyres, particles may be exported (or lost) to other oceanic or coastal areas (Majer et al. 2012) or sink due to physical degradation of larger items of floating debris and biofouling. It is very likely that the sea-surface micro-layer (upper 50 to 100 µm) has significantly higher concentrations of microplastic than the underlying layer (Song et al 2015). After sinking, microplastics may be re-dispersed by deep-sea currents to more remote waters and a proportion may accumulate in the water column or on the ocean floor, depending on the settling rates of sedimenting particles. High concentrations of floating macroplastics occur on mid-ocean islands, partly as a result of direct wind forcing.

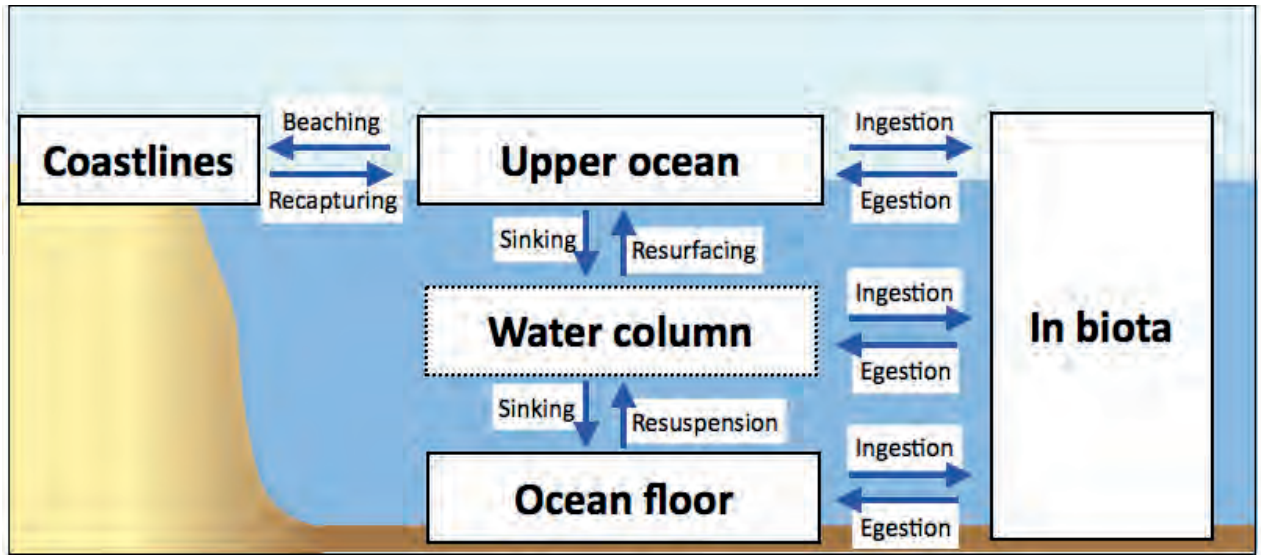


Figure 3.1 *Overview of compartments and fluxes of marine microplastics. Figure prepared by Erik van Sebille*

3.2.3 *Microplastics in the water column*

It is less understood how much microplastics reside just below the ocean surface. Recent modelling (Kukulka et al. 2012) and observations with vertically stacked trawl nets (Reisser et al. 2015) show that, depending on sea state, a significant fraction of microplastics may be mixed down due to wave breaking and mixing in the upper few metres of the ocean surface. Since most 'standard' trawls only skim the top 10 cm of the ocean surface, they may miss a considerable fraction of microplastics, especially in rough seas. There may also be microplastics deeper in the water column, below the mixed layer. A proportion of this microplastic debris will be neutrally buoyant, a proportion may be settling to the seafloor and a proportion ascending towards the sea surface following the breakdown of organic or inorganic (e.g. calcium carbonate dissolution) binding substances.

3.2.4 *Microplastics on the seafloor*

Sediments in the deep ocean are suggested to be a long-term sink for microplastics (Cózar et al. 2014; Eriksen et al. 2014; Woodall et al. 2014). Microplastics have been reported in marine sediments worldwide (Claessens et al. 2013; Van Cauwenberghe et al. 2013; Woodall et al. 2014) and the first report in subtidal sediments dates back to 2004 (Thompson et al. 2004). Deep sea sediments were demonstrated more recently to also accumulate microplastics (Van Cauwenberghe et al. 2013; Woodall et al. 2014) with composition that appears different from surface waters. Fibres were found at up to four orders of magnitude more abundant in deep-sea sediments from the Atlantic Ocean, Mediterranean Sea and Indian Ocean than in contaminated sea surface waters (Woodall et al. 2014).

Estimating the accumulation of microplastics in surface sediments requires a better understanding of biogeochemical and physical processes that affect sinking and accumulation, particularly to identify probable areas of accumulation. In the Lagoon of Venice for example, Vianello et al. (2013) detected the lowest microplastic concentrations where water currents are higher (outer lagoon, >1 m s^{-1}) when the inner lagoon, which is characterized by lower hydrodynamics, had a higher fine particle (<63 mm) fraction in the sediment. On the deep sea floor, circulation is not well explained and pathways are different from surface circulation. Submarine topographic features may also favour sedimentation and increase the retention of microplastics at particular locations such as canyons and seeps or smaller scale structures (e.g. holes, rocks, geological barriers). As for larger debris, human activities may also affect composition and repartition, as shown with the high densities of microplastics found in harbour sediments (up to 391 microplastics/kg of dry sediment; Claessens et al., 2011). Similarly, in Slovenia (Laglbauer et al. 2014), between 3 and 87 particles per 100g were found, with coastal areas more affected.

Table 3.1 Abundance of microplastics in sediments worldwide. Modified from Van Cauwenberghe et al. 2015

Continent	Location	Location specification	Depth (m)	Particle size	Measured abundance	Reference
Americas	US	Maine subtidal		0.25 - 4 mm	105 items/L	Graham and Thompson 2009
	US	Florida subtidal		0.25 - 4 mm	116 - 215 items/L	Graham and Thompson 2009
	Brazil	Tidal plain		1 - 10 cm	6.36 - 15.89 items/m^2	Costa et al. 2011
Asia	India	Ship-breaking yard		1.6 - 5 mm	81.4 mg/kg	Reddy et al. 2006
	Singapore	Mangrove		1.6 - 5 mm	36.8 items/kg dry	Nor and Obbard 2014
Europe	UK	Estuary			2.4 - 5.6 fibres/50 mL	Thompson et al. 2004
	Sweden	Subtidal		2 - 5 mm	2 - 332 items/100 mL	Norén 2007
	Belgium	Harbour		0.38 - 1 mm	166.7 items/kg dry	Claessens et al. 2011
		Continental Shelf	0 - 200		97.2 items/kg dry	
	Italy	Subtidal		0.7 - 1 mm	672 - 2175 items/kg dry	Vianello et al. 2013
	Slovenia	Infralittoral	(<50m)		30 - 800 items/kg dry	Laglbauer et al. 2014
Oceanic sediments	polar ocean, Mediterranean, North Atlantic, Gulf of Guinea	Deep sea	1176 - 4848	5 - 1 mm	0.5 items/cm^2	Van Cauwenberghe et al. 2013
	NW Pacific	Deep sea trench	4869 - 5766	0.300 - 5 mm	60 - 2020 items/m^2	Fisher et al. 2015
	Subpolar /North Atlantic	Deep sea mount Slope	1000 - 2000	0.032 - 5 mm	10 - 15 pieces per 50 ml	Woodall et al. 2014
	North East Atlantic	Canyons/slope	1400 - 2200	0.032 - 5 mm	6 - 40 pieces per 50 ml	Woodall et al. 2014
	Mediterranean	Canyons/slope/Basin	300 - 3500	0.032 - 5 mm	10 - 35 pieces per 50 ml	Woodall et al. 2014
	SW Indian	Sea mount	500 - 1000	0.032 - 5 mm	Up to 4 pieces per 50 ml	Woodall et al. 2014

3.2.5 *Shoreline/Coastal regions*

Although the total amount of microplastics on coastlines is not known, there are examples of studies quantifying microplastics at local and regional scales. Coastal studies have been carried out in many places across the globe, including Japan (Kako et al. 2010), Hawaii (Carson et al. 2011; Ribic et al. 2012; Agustin et al. 2015), South Africa (Madzena et al. 1997), Brazil (Santos et al. 2009), Australia (Hardesty et al. 2014), and Portugal (Antunes et al. 2013). An extensive summary of plastic studies along coastlines of the Pacific and Atlantic is given in tables 10.2 and 10.4 of Lusher et al. (2015).

Plastic on beaches is the most recognized form of visible marine plastic, and therefore attracts great attention from the general public. Still, it is not clear what the ecological impact of plastic on coastlines is. Although microplastics are hard to readily observe in sand, there are relevant studies showing microplastics on beaches and even a study on the significant amount of plastic buried on a beach in Brazil (Turra et al. 2014).

It is also important to realize that there are coastlines other than beaches, and that these can retain microplastics too. Again, there is very little large-scale data about the distribution of microplastics on non-beach coastlines, although mangroves, for example, are thought to retain large amounts of plastic litter (Debrot et al. 2013).

3.2.6 *Biota*

Several studies have reported the ingestion of microplastics by marine organisms from the field (e.g. marine mammals, birds, fish, bivalves, polychaetes, and crustaceans; see Chapter 4, Figure 4.4) and laboratory experiments (fish, polychaetes, bivalves and plankton; see Chapter 4, Figure 4.4). Active ingestion (filter-feeding or confusion with prey) and ventilation are commonly deemed to be the main pathways for ingestion of microplastics by marine fauna. Thus, biota may represent an important sink and potential transport mechanism for microplastics.

3.3 Transport within compartments

3.3.1 *Upper ocean*

Microplastics floating on the surface of the ocean can be considered passive, to a first approximation, and subject to surface currents. However, the exact depth at which the microplastics reside has large impacts on its pathway, as the currents in the upper ocean vary quite significantly over the top 50 metres or so (in a spiral-like fashion called the Ekman spiral, where currents a few tens of metres deep can be in the opposite direction of those at the surface). The buoyancy of the microplastics and the amount of wind mixing and waves breaking make it very difficult to predict where plastic particles reside. In general, there appears to be an exponential decay of microplastics with depth (Kukulka et al. 2012, 2015; Reisser et al. 2015; Brunner et al. 2015).

Beyond vertical mixing, waves and wind also affect the horizontal transport of microplastics. Stokes drift within waves can be a significant factor in the pathway of plastic, especially in coastal regions. Wind forcing has an important role in transporting macroplastic debris that has some part of the debris above the water's surface when floating in the ocean, but is less likely to affect microplastics.

The properties of plastic objects and particles (size, density, shape etc.) may change as a result of physical, chemical and biological processes, which will influence their subsequent behaviour and distribution. Fragmentation is a physical and mechanical process; oxidation, mediated by solar UV radiation, breaks the chemical bonds and facilitates fragmentation. The same process also occurs in thermal oxidation. Polyolefins undergoing auto oxidation are believed to also undergo chain scission as a part of the propagation reaction step. Where it is facilitated by solar UV radiation it is reasonable to expect the fragmentation changes to be localized to a surface layer demarcated by the depth of penetration of the UV radiation, and therefore also dependent on biofouling.

Table 3.2 Specific gravity of common plastics and seawater (adapted from Andrady, 2011)

Plastic type		Specific gravity
Polypropylene	PP	0.83-0.85
Low-density polyethylene	LDPE, LLDPE	0.91-0.93
High-density polyethylene	HDPE	0.94
Polystyrene	PS	1.05
Thermoplastic polyester	PET	1.37
Poly(vinyl chloride)	PVC	1.38
Seawater		1.03

Fragmentation does not change the density of polymers but alters their sizes (and therefore the specific surface area), which largely affects the transport and distribution of the plastics. Higher ambient temperatures on beaches, termed thermal loading, accelerate this fragmentation process relative to that for plastics in seawater. However, the rates of fragmentation or half-lives of plastics on beaches or in seawater surfaces are not known. The research literature has addressed the issue of comparative degradation rates on land and sea surface (as well as ocean sediment), but the investigations have solely focused on loss in mechanical properties that occur as a prelude to any fragmentation. A recent study showed that in the Mediterranean plastic debris are dominated by millimetre-sized fragments with higher proportion of large plastic objects than

the microplastic fragments in oceanic gyres (Cózar et al. 2014). These observations may reflect the closer connection of the Mediterranean with pollution sources or the 'closed system' nature of the region. However, the upper Mediterranean Sea kinetics and rates of fragmentation are still unknown.

Finally, disintegration of plastics by interactions with at-sea vessels may be caused by the mechanical stresses encountered in collisions, grinding in propellers, or from passage through circulation systems. Though expected to be a minor process compared to other disintegration mechanisms, such anthropogenic processes may be non-negligible, especially for polystyrene foam that may comprise as much as 90% of litter floating in coastal zones (Hinojosa and Thiel 2011) and 18% of microplastics in the Mediterranean Sea (Collignon et al. 2012).

The transport of plastic on the surface of the ocean can also drive the dispersal of marine organisms. From microbes to invertebrates, many organisms have always attached to natural floating substrates (macroalgae, feathers, wood and pumice) and one might therefore ask why we should be concerned about plastic transporting organisms? One important difference is the longevity of plastic relative to most of the natural substrates, allowing more mature communities to form and persist, perhaps even breed, and thus transport viable populations further. The distribution of plastic is different from that of natural substrates, and plastic has substantially increased the available substrate in oligotrophic open ocean regions, potentially altering the distributions of marine organisms (Goldstein et al. 2012; Majer et al. 2012), including the Southern Ocean (Barnes et al. 2003). Also, considering the volume of floating debris that leaves coastlines following catastrophic events, there is a concern that populations of organisms, as opposed to individuals, may survive the long journey from one continent to another.

3.3.2 *Water column*

Plastics with a density that exceeds that of seawater (Table 3.2; >1.02 kg/dm^3) will eventually sink and accumulate in the sediment, while lower-density particles tend to float on the sea surface or in the water column. It has been suggested that even low-density plastics can reach the seafloor. Biofouling can lead to an increase in density resulting in the sinking of microplastics (Andrady 2011). Indeed, analysis of polyethylene bags submerged in seawater showed a significant increase in biofilm formation over time, accompanied by corresponding changes in physicochemical properties of the plastic, such as a decrease in buoyancy (Morét-Ferguson et al. 2010; Lobelle and Cunliffe 2011). These studies suggest that biofouling can contribute to the sinking and eventual burial in sediments of previously buoyant plastic. Thus, biomass accumulation on plastic may help to partly explain the reported discrepancy between observed concentrations of floating microplastics in the open ocean and that quantity estimated as having been introduced into the marine environment (Cózar et al. 2014; Eriksen et al. 2014; van Sebille et al. 2015), but the extent of this effect has not been quantified. In addition, aggregation with organic matter (i.e. as faecal pellets or marine snow) has been suggested as a route of transport for microplastics to deep-sea sediments (Van Cauwenberghe et al. 2013b).

Although the kinetics of fragmentation and the particle size spectrum that results remain unknown for even the most commonly used plastics at sea, many processes in the marine environment that cause disintegration have been identified. The mechanical energy required for disintegration may come from physical, biological or anthropogenic processes. The wind, sand and wave action at the sea surface, on sea floor or on beaches abrade or alter weakened plastics. Some animals also reduce object size by biting or chewing materials and marks from large fishes, including sharks, or birds have been reported, especially on polystyrene debris (Cadée 2002; Carson 2013). Grinding ingested plastics may also reduce the size of plastic marine debris, altering hundreds of tons annually for tube-nosed seabirds only (van Franeker et al. 2011) and even minor disintegration in fish stomachs could represent a non-negligible contribution to particle fragmentation.

3.3.3 *Deep ocean*

Mechanisms influencing the distribution of microplastics on the sea floor are not well understood. Microplastics are more likely to be influenced by advection than larger items and, more generally, circulation patterns at all ocean levels (Woodall et al. 2014). Ocean dynamics could then explain the accumulation of plastics in the deep sea or shallower waters depending on size and density. Recently, Ryan (2015) suggested that small items should start sinking sooner than large items because it requires less biofouling to make them negatively buoyant.

Deep ocean currents are extremely enigmatic, and it is not clear at all whether there are circulation patterns near the ocean floor that could create hot-spots. It could be hypothesized that microplastics would accumulate in deep canyons, as material might over time be slowly drawn down by a combination of turbulence and gravity. However, there is little empirical evidence for these accumulation patterns. In any case, we currently know so little of our ocean floor (our maps of the planet Mars are 25 times more accurate than those of the ocean floor) and mapping is so expensive that a global estimate of the amount of plastic in the deep ocean may be decades away.

Due to non-availability of light, lower temperatures, and lower oxygen levels at the ocean bottom, plastics there tend to accumulate close to their original form for long (as yet undetermined) periods of time.

3.3.4 *Coastlines*

Microplastics on coastlines are influenced by a number of physical and chemical processes, including weathering degradation and transport by waves and wind. Transport is likely to be greatest during storms, and particles can be moved farther inshore by ballistic 'jumps'. Furthermore, microplastics may get buried in the sand, either through naturally occurring beach erosion and sedimentation, or through beach engineering work such as replenishment. Turra et al. (2014), for instance, found large amounts of microplastic pellets

deeper in the sand on a Brazilian beach, revealing that sandy beaches may act as permanent or temporary sinks for microplastics. There is some evidence that the presence of microplastics can alter the rate at which beach sands change temperature (Carson et al. 2011).

Coastlines may be a large sink of plastic, as plastic is deposited on shorelines. Regular shoreline clean-up activities can remove significant quantities of litter, though such activities typically remove larger debris items (Ocean Conservancy 2015), and are restricted to beach or coastal regions. The coastlines are arguably the most convenient and cost-effective place to collect marine plastic litter, as no vessels are needed and working on land is typically easier than working on the ocean. However, there is a severe lack of globally standardized data on the amount of plastic removed from beaches in (volunteer-led or government-led) clean-ups, in particular for microplastics. This prevents a holistic large-scale understanding of spatial and temporal trends.

3.3.5 *Biota*

Biological processes (e.g. fouling, ingestion, aggregation), and their interaction with the above physical processes, will influence how microplastics are transported within and between different ocean habitats. Properties of the microplastic particles themselves (e.g. type, density) will affect how they interact with these biological processes. For example, polypropylene is a common type of plastic used in rope and has a density of 0.9 g/cm^3 (Hidalgo-Ruz et al. 2012). It will therefore float in seawater (assuming an average seawater density of 1.02 g/cm^3), which means that surface-feeding pelagic organisms are more likely to ingest it. Heavier microplastics such as those composed of polyvinyl chloride (PVC) and polyethylene (PET) are more likely to sink and therefore be ingested by benthic organisms. It's also important to note that over time, low-density polymers may become fouled and sink (Morét-Ferguson et al. 2010; Long et al. 2015), in which case these lighter plastics may become available to benthic organisms.

Phytoplankton communities can have impacts on microplastic distribution in the water column (Long et al. 2015). Depending on the ballasting properties and aggregation of phytoplankton communities, the removal and export of microplastic to the sea floor can be enhanced. In addition, within the food web, microalgae attached to microplastics are assumed to be more easily captured by filter feeders than free microplastics in the water column.

Some marine animals are indiscriminate feeders that will ingest anything in the appropriate size range. Others use visual, chemical and electrical cues for finding and selecting food, so the probability of a piece of microplastic being ingested depends not only on size and encounter rate, but also on a number of other cues including shape, colour, smell and taste. The smell and taste of microplastic will be influenced by the microbial biofilm on the surface, and microbes colonize plastic in seawater very quickly; within a week most of the surface is covered. This thin layer of living organic matter and by-products like extracellular polymeric substances (EPS) "slime" make the plastic smell and presumably taste like nutritious particles. This increases the likelihood of ingestion by animals that use chemoreception to select food particles and thus impacts flux from the water to biota. Likelihood of ingestion and impact on the organism ingesting it will vary depending on the composition of microbial community including whether it includes potential pathogens.

Another challenge is that the microbial community associated with microplastics also varies regionally and seasonally (Oberbeckmann et al. 2014), as well as on larger scales such as between the Atlantic and Pacific Ocean basins (Amaral-Zettler et al. 2015). This variability suggests that risk management will have to vary regionally to be effective.

3.4 Fluxes between compartments

Equally important is understanding how microplastics move between compartments, for example from the upper ocean to the deep sea sediments. These mechanisms include various physical (e.g. density), mechanical (e.g. waves and currents), chemical (e.g. oxidation) and biological (e.g. bio-transport and biofouling) processes.

It remains an open question how microplastics leave the ocean surface. From this compartment, it can move to any of the other four compartments: to the water column and seafloor by sinking (most likely through density changes resulting from biofouling; Andrady 2011), to the shoreline by beaching or stranding (which may be event-driven as storms wash up large amounts of microplastics), and into biota through ingestion and aggregation in organic matrices. However, details of movement or transport between compartments is poorly understood.

A number of oceanographic processes could aid in the transfer of microplastics to depth. As stated in Woodall et al. (2014), these processes include dense shelf water cascading, severe coastal storms, offshore convection and saline subduction. All these induce vertical and horizontal transfers of large volumes of particle loaded waters, including grains of various sizes and nature, as well as litter and contaminants, from shallow ocean layers and coastal regions to deeper ones. Submarine canyons act as preferential conduits for larger debris (Galgani et al. 1996; Pham et al. 2014). Lighter weight plastics may also find their way onto the seafloor if they pass through the gut of organisms and are released in faecal pellets or bound in other excretory materials (e.g. mucous). The incorporation of plastics into sediments provides an additional marker to the beginning of the Anthropocene (Zalasiewicz et al. 2015).

Microplastics reach coastlines by beaching, which in itself depends on the currents, sea state, wind, tides and coastal properties. It may very well be that beaching is very intermittent, with low fluxes most of the time and then some large fluxes in short time windows associated with storms (Agustin et al. 2015). Even less is known about how much plastic is recaptured into the ocean from coastlines.

3.5 'Hot-spots' and scale-dependency

In this context, the term 'hot-spots' is used to help describe the heterogeneity observed in the distribution of marine plastic litter; i.e. there are locations of relatively high abundance. The use of the term should not be taken to imply an ecological or human health hazard (as may be the case when considering environmental health guidelines), but to help focus attention and possible mitigation efforts. The mechanism by which 'hot-spots' form depends to a large degree on how plastic moves within and between the five different compartments. Understanding the ocean budget for microplastics requires knowledge on both the inventory (stock) and movement (flux). Because it is much harder to measure fluxes than stock, this is an area where even less is known.

As discussed above, microplastics accumulate in open-ocean and coastal areas and can be vectors for pollutants and pathogens, imposing multiple stressors on marine biota. However, the risk of such impacts depends on the type, size and amount of plastic present in the environment, the presence of contaminants in the region and contact with sensitive biota.

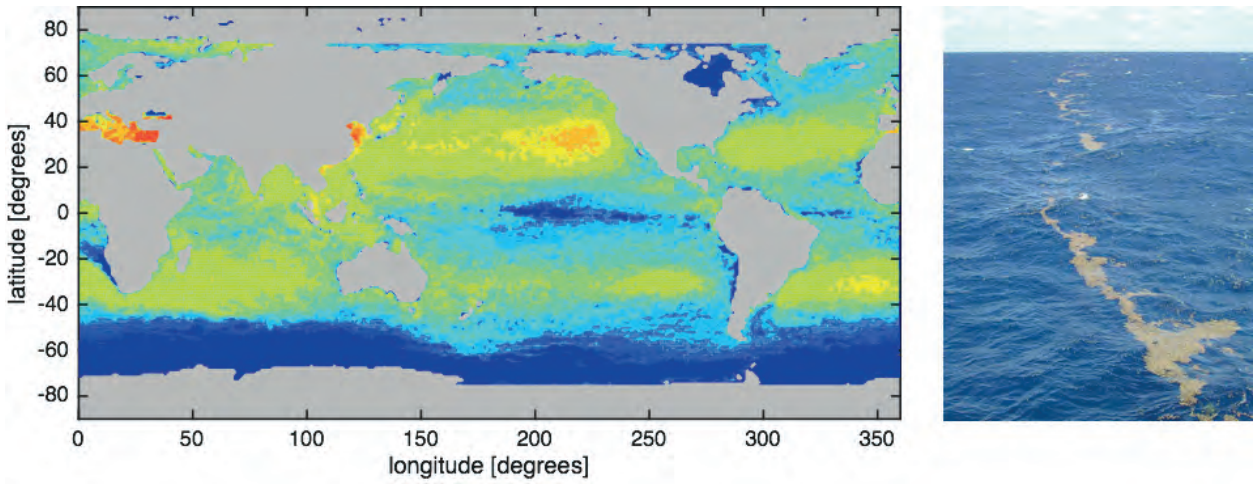


Figure 3.2 *A map and a photo showing the different scales on which microplastics can accumulate. The map (left) is from van Sebille et al. (2015) and provides an estimate of the distribution of small (<20 cm) floating plastics in the global ocean (numbers of particles km^{-2}; colour scale – from red >1x10^{6} to blue <1x10^{0}). The photo (right) shows pumice accumulating in a wind row (through NOAA Ocean Explorer) which could be representative for floating microplastics. Colours range from dark blue to yellow to red in order of increasing plastic accumulations*

One of the difficulties in assessing the amount of plastic in the ocean is that the distribution of microplastics tends to be 'patchy' (see Figure 3.2). On many different scales, from global (in accumulation zones), to regional to very local (e.g. Langmuir cells), the amount of plastic can vary by orders of magnitude (Law et al. 2014). For example, wind rows (produced by the so-called Langmuir circulation, where waves and winds create a complicated surface circulation) can create very large density differences within metres.

Models can aid in identifying hot-spots, especially if their ability to accurately simulate plastic behaviour and pathways improves. As hot-spots are areas where the density of microplastics are highest, models might find it easier to simulate these areas than their lower-density counterparts.

The existence of hot-spots has implications for impacts and risk assessment (see Chapter 8), as well as for monitoring strategies (see also Chapter 7). On the other hand, the patchiness and existence of hot-spots can provide an opportunity for strategic and cost-effective intervention points.

In addition to the variability of plastic sources, sinks, pathways and movement on different temporal scales, there is also tremendous spatial variability. It is important to consider source hot-spots and how these may be similar or different from accumulation hot-spots. On the global scale, surface plastic accumulates in subtropical gyres (Lebreton et al. 2012; Maximenko et al. 2012; van Sebille, 2015), demonstrating the heterogeneity in accumulation of microplastics. Small-scale processes such as wave interactions, Langmuir circulation and (sub) mesoscale eddies create a heterogeneous, patchy debris field on the surface of the ocean. Concentrations of floating plastic might therefore vary considerably on length scales of less than 100 m. There is relatively little known about the patchiness at such fine-scale resolution, even though patchiness is an important concept when interpreting surface trawl microplastics data. It is entirely conceivable that hitting or missing a high-concentration patch with a trawl might impact the results of an observational study (Law et al. 2014). The patchiness in microplastic accumulation on the sea surface requires a less-patchy sampling effort, meaning more surface trawls over a wider area may smooth out the count and weight estimates that are sometimes compromised by random high or low accumulations.

On slightly larger scales (e.g. 100s of km), the concentrations of floating plastic are also heterogeneous. Local patches of down-welling creates accumulation zones of a few tens of kilometres or less in size. Importantly, there are large knowledge gaps of where these mesoscale accumulation regions are located. While the model results from Maximenko et al. 2012, Lebreton et al. 2012, and van Sebille et al. 2014 agree roughly on the location of the large-scale open-ocean accumulation zones in the centres of the gyres, the three models place these meso-scale accumulation zones at very different locations (van Sebille et al. 2015). These meso-scale accumulation zones

might hold a significant amount of floating plastic and, because they are often located much closer to shorelines and biologically productive regions, might have a disproportionately large impact on marine life (Wilcox et al. 2015).

3.6 Conclusions, knowledge gaps and research priorities

3.6.1 *Conclusions*

This chapter focuses on the distribution, fate and hot-spots for microplastics in the ocean. In general, microplastics in oceanic compartments are patchily distributed and movement and distribution is not well understood from empirical data. However, recent work is improving our knowledge in this area, though we are learning how microplastics reach coastlines and may be re-suspended. Microplastics move with currents, wave action, are likely lofted in windy conditions and are distributed throughout the water column. Because, however, there is little empirical information about the distribution of microplastics in most compartments, it is difficult to understand and identify microplastics hot-spots, as well as to quantify microplastic distribution and densities in space and time. Furthermore, because of these knowledge gaps, it can be challenging to make meaningful predictions about the relative transport and exchange of microplastics between compartments. However, lessons can be taken from recent modelling of ocean plastics movement and distribution, and applying lessons learned from recent work will improve our understanding here.

3.6.2 *Knowledge gaps*

There are currently much better data for plastics distribution of larger (meso, macro) plastic than currently exists for micro- and nanoplastics. This is due in part to difficulties in identifying and quantifying smaller particles, and is partly due to the vastness of the ocean and the difficulty in applying consistent, robust sampling techniques at scale. There are more comprehensive data available on larger plastics (and plastic fragments), particularly from land-based sources, than there are for microplastics in the ocean, as macroplastics have been systematically monitored in some regions for up to five decades. Systematic monitoring of microplastics, as distinct from opportunistic sampling on research cruises, is in its infancy. Making model predictions based upon best-available information can help to resolve some of these challenges and will enable us to better identify geographic target regions and compartments to identify threats and risk posed by microplastic particles.

How (and how much) microplastics leave the ocean surface remains an open question. We currently have little information on the quantities of primary, manufactured microplastics entering our waterways as well as their fragmentation and breakdown rates. These knowledge gaps necessarily restrict our understanding of the distribution, fate and hot-spots of microplastics in the marine environment, though there have been recent studies summarizing the state of knowledge of marine plastics in general. Applying the knowledge gained from this recent work will improve our understanding of the vertical distribution of microplastics in the ocean.

3.6.3 *Research priorities*

Overall, research should relate small to large-scale sampling, monitoring and modelling, considering:

- Identification of plastic sources (amount and type) in coastal areas;
- Use of circulation and tracking drifters models to link hot-spots to pathways;
- Improvement of plastic biogeochemical processes in models;
- Standardization of modelling techniques, including time and space resolutions, (e.g. use particular sites with detailed information to inform particular models) and include evaluation and calibration based on empirical information as possible;
- Couple ocean circulation with coastal drift models to improve understanding of movement, transport and fate of microplastics;
- Use of inverse Lagrangean models to detect potential sources of plastics and evaluate the influence of changing climate in plastic dispersion;
- Apply scenario modelling to evaluate potential environmental, economic and socio-cultural risks;
- Establish how hot-spots link to ecological impacts;
- Improve our understanding of how hot-spots arrive, form and persist (spatially, temporally and with respect to vertical distribution), including physical processes;
- Integration of expertise from several scientific areas (e.g. ecology, chemistry, ecotoxicology) into discussion; and
- Estimate contamination of coastal sites (such as sandy beaches, estuarine silts and mudflats) by Persistent Organic Pollutants (POPs) and heavy metals due to plastic dispersion.

Some additional research priorities include:

- Developing better methods to age or date plastics, associated with developing weathering and fragmentation models to better understand secondary microplastic generation;
- Better understanding how microbial interaction affects the fate and behaviour of microplastic;
- Predicting dispersal of species on microplastic;
- Understanding the fate and impacts of nanoplastics;
- Understanding the fate of and impacts from biodegradable plastics; and
- Studying sinking phenomenon to understand vertical transport of microplastic.

4 ECOLOGICAL IMPACTS OF MICROPLASTICS

Key points

1. Microplastics have been documented in a diversity of habitats and in over 100 species.
2. Microplastics can impact an organism at many levels of biological organization, including at the levels of populations and assemblages. Still, the majority of the evidence is at levels that are sub-organismal (e.g. changes in gene expression, inflammation, tumour promotion) or affect individual organisms (i.e. death).
3. Microplastics can be a source and sink of hazardous chemicals to organisms, but its relative importance as a source of chemicals to wildlife relative to others (e.g. water, sediment, diet) remains under investigation.
4. Nano-sized plastics are probably as common as micro-sized plastics, yet the hazards may be more complex.
5. Microplastics can transport invasive species, including harmful algal blooms and pathogens.

4.1 Lessons from the first assessment

The GESAMP 2015 report demonstrated that a wide range of marine organisms across all trophic levels, including invertebrates, fish and seabirds, are contaminated with microplastics. In some cases, the incidence of ingestion is widespread across populations. Marine organisms are exposed to microplastics via feeding (including filtration, active grazing and deposit feeding) and transport across the gills (ventilation). The uptake, accumulation and elimination of microplastics by marine organisms depends on the size of the particle. The risk of associated impacts following exposure to microplastics depends on: i) the number of particles; ii) the type of particles (e.g. polymer type, size, shape and age; iii) the duration of exposure; iv) the concentrations and type of contaminants associated with the plastic; and, v) the physiology and life-history of the organism.

The GESAMP 2015 report laid out the state of the evidence regarding the impacts of microplastics. It reported that microplastics can have toxic effects, including decreasing energy reserves, changes in feeding behaviour, movement, growth and breeding success. Moreover, small microplastics can cross cell membranes into cells and tissues and may cause particle toxicity (e.g. provoke an immune response with associated inflammation and cell damage). Furthermore, chemicals associated with microplastics can concentrate in tissues. This has been shown in animals during laboratory experiments. Still, there is little evidence from the field to demonstrate the extent that this occurs under natural conditions (and relative to other sources of anthropogenic chemicals to wildlife) and thus the relative importance of contaminant-exposure mediated by microplastics as compared to other sources requires further research.

Lastly, the 2015 report pointed out some areas where information regarding impacts from microplastics is lacking. The previous report points out that many of the demonstrated impacts have only been demonstrated in the laboratory, often at high exposure levels, for short time periods and without dose-response measurements. Furthermore, there is concern about the potential of nano- and micro-sized plastic debris to translocate non-indigenous species, including pathogenic organisms. As such, more ecologically relevant studies and additional observational experiments in nature are required because we still do not understand ecological- and ecosystem-level impacts of nano- and micro-sized plastic debris.

This report aims to fill in some of the gaps pointed out in the last report by diving deeper into the existing evidence and highlighting some of the new evidence since publication of the first report and through October of 2015. This chapter, in particular, reviews some of our current understanding regarding how microplastic debris and its associated chemicals and microbiota impact wildlife. The contents of this chapter are organized to facilitate risk assessment by outlining what we know about the exposure and impacts of microplastic pollution. Specifically, this chapter first discusses exposure and impacts related to microplastic itself, followed by the impacts related to microplastic-associated chemicals. In addition, we discuss the burgeoning evidence regarding the exposure and impact of nano-sized plastics and the role of microplastic in transporting microbiota.

4.2 Occurrence of microplastics in biota

4.2.1 *Microplastics in the marine environment*

The spatial extent and quantity of microplastic particles in the marine environment are raising concern among environmental managers and policy-makers regarding impacts to ecosystems (Eriksen et al. 2014; Thompson et al. 2004). As a result of widespread contamination, a diverse array of wildlife is exposed to microplastics. Contamination in the form of ingestion has been recorded in tens of thousands of individual organisms and over 100 species (Gall and Thompson 2015; Lusher et al. 2013, 2015). In some species, ingestion is reported in over 80% of sampled populations (e.g. Murray and Cowie 2011; Kühn et al. 2015), which may be an issue if the exposure causes an impact. The physical particle, the associated chemicals and/or associated pathogens can cause adverse effects and are discussed in more detail in this chapter.

To assess the impacts of microplastic contamination in wildlife, it is important to know the level and nature of the exposure. Exposures will vary based upon many factors, including location, habitat type and life-history strategies. For example, animals that live in the accumulation zones in subtropical gyres and feed from the surface are likely to be exposed to relatively large concentrations of microplastic fragments. The risk of an impact from exposure will likely depend on many

factors, including the concentration, type, size and/or shape of microplastics as previously outlined in the GESAMP 2015 report.

The quantity and frequency of occurrence help us understand the dose of microplastics to animals in nature. Microplastic debris has been reported in multiple oceanic habitats globally. A recent study estimates that there are more than 5 trillion pieces of plastic particles (>0.33 mm) floating in pelagic habitats globally (Eriksen et al. 2014), and other studies reveal the presence of microplastics in remote habitats such as on seamounts and coral reefs in the deep sea (Woodall et al. 2014). It is important to have an understanding of how much microplastics are in different types of habitat in the environment, and the types and shapes that are found. A variety of types and concentrations of microplastics have been reported in the environment. See Chapter 2 for more information regarding quantities and types in different habitats globally. This information can be used in risk assessment and to ensure scientists design ecologically relevant laboratory experiments measuring the impact of microplastics at realistic exposure scenarios and concentrations to organisms.

4.2.2 *Exposure pathways and concentrations of microplastics in marine organisms*

Globally, marine organisms across many trophic levels interact with microplastics via a number of pathways (Figure 4.1). As a consequence, there are many mechanisms by which an organism can take up this material. Microplastics can adhere to the body (i.e. attached to external appendages; Cole et al. 2013) and/or be absorbed (i.e. taken up by the organisms into the body through cell membranes). Absorption of microplastics has been demonstrated in phytoplankton (Bhattacharya et al. 2010; Long et al. 2015). Alternatively, microplastics can be taken up across the gills through ventilation, which has been demonstrated in crabs (Watts et al. 2014). Lastly, organisms can ingest microplastics directly or indirectly. Direct ingestion has been demonstrated in over a hundred marine species (reviewed in Lusher 2015; see Section 4.2.4). Organisms can ingest microplastics as food, unintentionally capturing it while feeding or intentionally choosing it and/or mistaking it for prey (Lusher 2015). Organisms may also indirectly ingest plastic while ingesting prey containing microplastic, i.e. trophic transfer (e.g. Farrell and Nelson 2013).

To assess our understanding of how microplastics may be impacting wildlife, it is important to understand exposure pathways and concentrations used in laboratory experiments in comparison to those found in the environment. The following section reviews existing evidence from laboratory studies and observational studies in nature published up to the first quarter of 2016. Laboratory studies allow us to better understand mechanisms of uptake and consequential effects. They provide thresholds for toxicity and can inform risk assessment. In turn, knowing the concentrations of microplastic in wildlife also informs risk assessment and future experiments that are more environmentally relevant.

4.2.3 *Laboratory studies*

Historically, laboratory studies with microplastics were used to document ingestion rates and retention time of particles to understand feeding behaviour (Hart 1991; Ward et al. 1998; Bolton and Havenhand 1998; Greiller and Hammond 2006). More recently, scientists have used them to demonstrate uptake of microplastic debris (e.g. Thompson et al. 2004; Browne et al. 2008; Cole et al. 2013; Watts et al. 2014) and begin to learn about the impacts of microplastics (e.g. Browne et al. 2008; Teuten et al. 2009; Wright et al. 2013; Rochman et al. 2013a). To date, many laboratory studies have demonstrated that uptake of microplastics can occur in a range of species.

Examples of laboratory studies examining uptake of microplastics in multiple different taxa are summarized below. This compilation of studies demonstrates the exposure scenarios (i.e. range of concentration levels, exposure duration and species used) for exposures. It also provides information regarding the type of uptake that occurred, where relevant. Although some studies also tested and demonstrated impact, these results are summarized in a later section. See Table AIII.1 in the appendix for extensive tables that provide more detailed information about each laboratory study examined through October 2015.

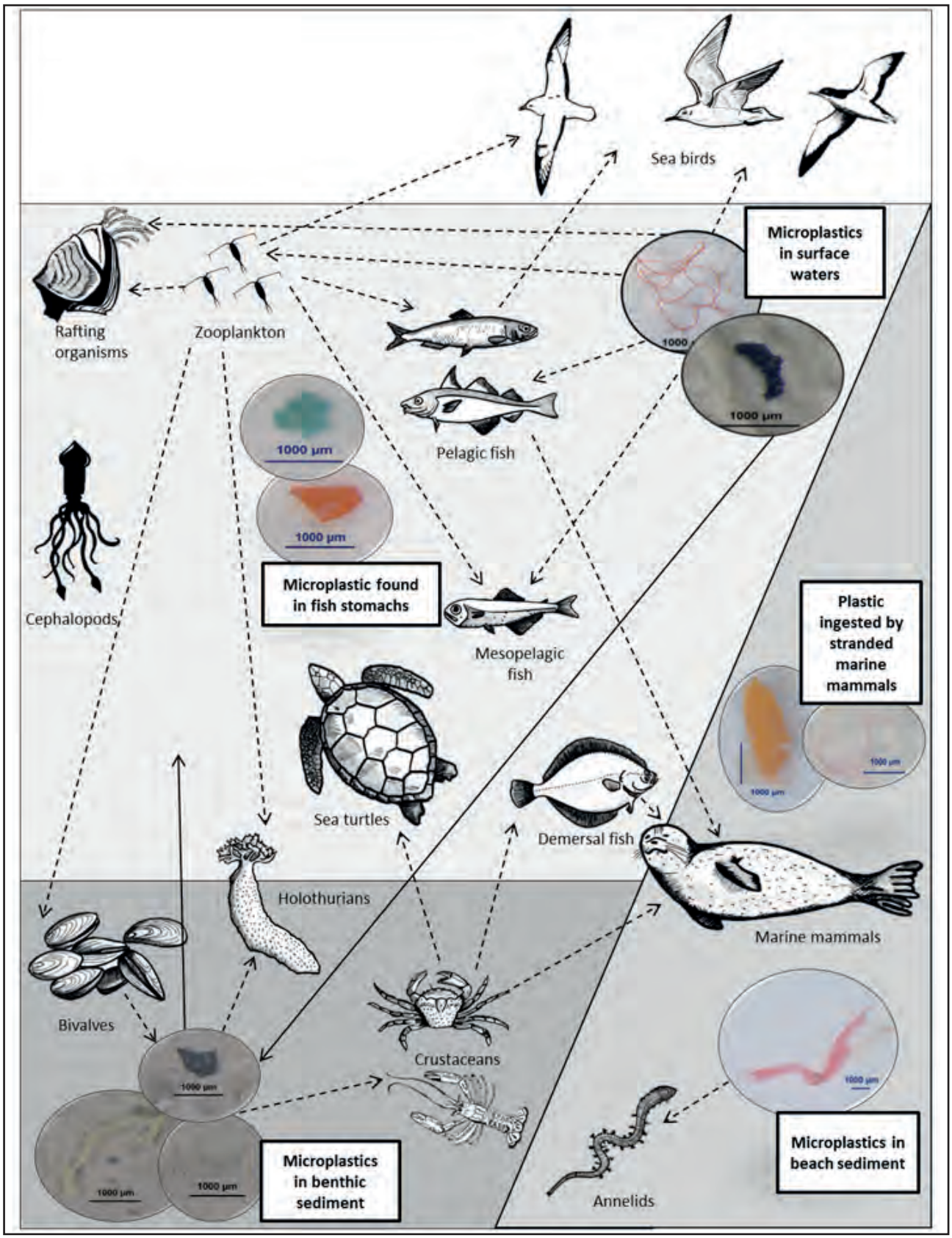


Figure 4.1 *Microplastics interactions with physical and biological matrices in the marine environment. Solid arrows represent environmental links (i.e. how microplastic may transfer between sediment and water) and dashed arrows represent biological links (i.e. how microplastic may transfer among trophic levels). (Reproduced from Lusher 2015, images and photos of microplastic: A. Lusher)*

Phytoplankton:

Exposure concentrations: 0.000046 to 40 mg/mL,
0.01% (w.w),
9 x 10^4 particles per mL

Exposure duration: 1 to 96 hr exposure

Interactions with microplastics: adhesion, absorption

References: Bhattacharya et al. 2010; Cedervall et al. 2012; Long et al. 2015; Davarpanah and Guilhermino 2015; Sjollema et al. 2016.

Zooplankton:

Exposure concentrations: 635 to 10,000 items per mL

Exposure duration: 1 to 24 hr exposure

Interactions with microplastics: adhesion, ingestion

References: Cedervall et al. 2012; Cole et al. 2013, 2014, 2015; Lee et al. 2013; Setala et al. 2014.

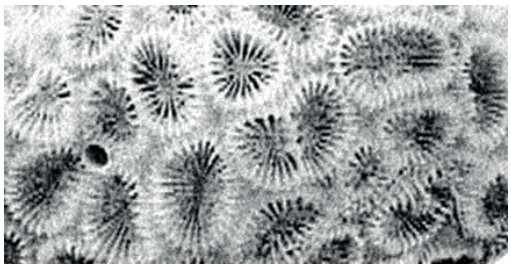
Cnidaria:

Exposure concentrations: 0.395 g microplastic per L

Exposure duration: 48 hr exposure

Interactions with microplastics: ingestion

References: Hall et al. 2015.



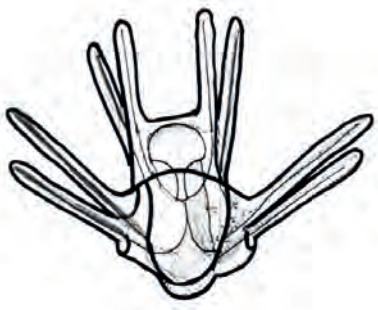
Echinoderms:

Exposure concentrations: 1 to 300 particles per mL
10 g to 60 g per 600 ml sand

Exposure duration: 20 hr to 9 d

Interactions with microplastics: ingestion, retention, egestion

References: Hart 1991; Graham and Thompson 2009; Kaposi et al. 2014; Nobre et al. 2015.



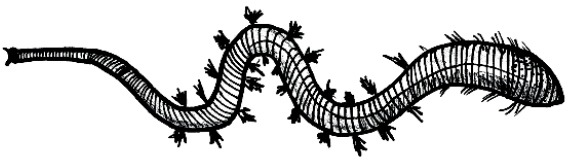
Annelids:

Exposure concentrations: 1.5 g/L
0 to 5% by weight
0 to 100 particles per L
2000 particles per mL

Exposure duration: 20 min to 28 d

Interactions with microplastics: ingestion

References: Bolton and Haverhand 1998; Thompson et al. 2004; Besseling et al. 2013; Browne et al. 2013; Wright et al 2013.



Mollusca:

Exposure concentrations: 1.05 to 3000 particles per mL
0.5 to 2.5 g/L
50 µL in 400 mL
1 to 199 µg/mL

Exposure duration: 45 min to 96 hr

Interactions with microplastics: ingestion

References: Lei et al. 1996; Brilliant and MacDonald 2000, 2002; Browne et al. 2008; Ward et al. 2009; von Moos et al. 2012; Wegner et al. 2012; Cole et al. 2013; Farrell and Nelson 2013; Avio 2015; Canesi et al. 2015. See also review in Ward and Shumway 2004.

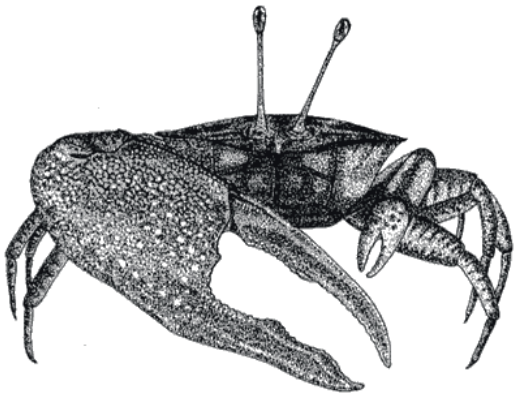
Crustacea:

Exposure concentrations: 5.25 x 10^5 to 9.1 x 10^{11} particles per mL
40 to 10,000 particles per mL
0.3 to 120 mg/g
108 to 1000 mg per kg

Exposure duration: 15 min to 2 months

Interactions with microplastics: ingestion, ventilation

References: Thompson et al. 2004; Murray and Cowie 2011; Ugolini et al. 2013; Chua et al. 2014; Hamer et al. 2014; Watts et al. 2014; Brennecke et al. 2015.



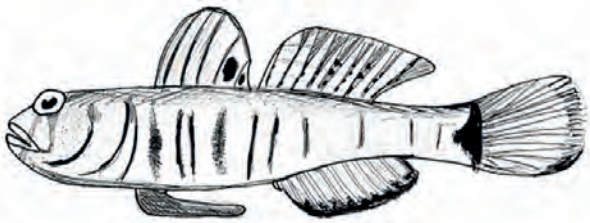
Fish:

Exposure concentrations: 10% of diet
3000 particles per mL;
0.216 mg/L

Exposure duration: 3 min to 2 months

Interactions with microplastics: ingestion

References: dos Santos and Jobling 1991; Cedervall et al. 2012; Oliviera et al. 2013, 2014 ; Rochman et al. 2013a, 2014a; Mazuras et al. 2014, 2015; De Sa 2015; Luis et al. 2015.



Sea Turtles: No laboratory studies to report

Seabirds:

Exposure: to contaminated resin pellets resulting in approximately 100 ng of PCB exposure per chick for 42 d

Exposure duration: 1 day

Interactions with microplastics: Ingestion

References: Reviewed in Teuten et al. 2009.

Marine mammals: No laboratory studies to report.

The above studies help us understand what animals may be impacted by microplastics and the mechanisms of uptake. It is noteworthy that many of these studies suffer from a lack of environmentally relevant concentrations and exposure scenarios making them less useful in understanding risks from current environmental concentrations. Below, we describe concentrations of microplastic that have been found in animals in the wild that we hope can inform future laboratory studies to measure impacts of microplastic as well as risk assessments.

4.2.4 *Field studies*

With numerous studies demonstrating the widespread distribution of microplastics in the marine environment, researchers began looking for evidence of microplastic uptake by wildlife. To date, microplastics have been found in a diversity of organisms with different feeding strategies (e.g. suspension feeding, deposit feeding, filter feeding, grazing, scavenging and predation) and at different trophic levels (Gall and Thompson 2015). Most studies have focused on the identification of microplastics in gut contents.

When microplastics are found in an organism, it is generally assumed that the debris was ingested directly. There is potential for microplastics to transfer from prey to predator and move up the food chain. At present, there is little evidence of this in natural systems. There are a few studies demonstrating trophic transfer of microplastics in laboratory settings (Cedervall et al. 2012; Farrell and Nelson 2013; Watts et al. 2014) and potential trophic transfer of microplastics in wild-caught animals (Eriksson and Burton 2003). Although research demonstrating the transfer of microplastics through the food web is limited, several species that represent key links for trophic transfer are known to ingest microplastics (e.g. small pelagic fish, copepods) and thus trophic transfer, with the possibility for increasing concentrations of particles to be found in higher trophic-level organisms (i.e. biomagnification), is likely to occur.

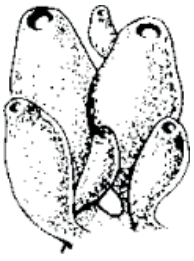
Although most of the existing studies have looked for microplastics inside the gut, microplastic may be exported into other parts of the body after ingestion or absorption via translocation. Browne et al. (2008) were the first to demonstrate that small microplastics have the potential to translocate from the digestive tract to the circulatory system of exposed mussels *Mytilus edulis.* Within three days after exposure to small polystyrene microspheres (3 and 10µm; 40 particles.mL^{-1}), microplastics were detected within the haemolymph of the organisms and persisted there for over 48 days. Smaller particles seem to undergo translocation more readily than larger ones (Browne et al. 2008). As such, more research is necessary to look for microplastic in wildlife in different parts of the body in addition to the gut content.

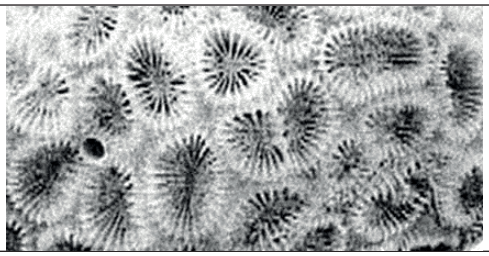
Examples of several of the studies demonstrating contamination of wildlife by microplastic debris are summarized below to demonstrate the presence and amount of microplastics in a range of wild-caught animals. One thing to note is that there is very limited information regarding retention time and excretion in an animal.

More extensive tables providing more detailed information from field studies are included in Tables AIII.2 and AIII.3 of the Appendix.

Phytoplankton: No field studies to report.

Zooplankton: Microplastic was identified in *Neocalanus cristatus* and *Euphausia pacificas* and other species captured from the wild (up to 1 particle per 7 zooplankton). **References:** Desforges et al. 2015	

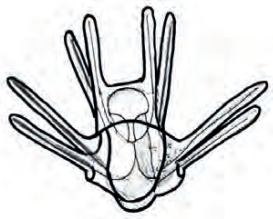
Porifera: The sponge *Hymeniacidon perlevis* was reported to contain 50 x 10^3 microplastic particles per kg dry weight. **References:** Karlsson 2015	

Cnidaria: The sea anemone *Actinia equina* contained about 10 x 10^3 microplastic particles per kg dry weight. **References:** Karlsson 2015	

Echinoderms:

The brittlestar *Ophiura* sp. contained 66 x 10^3 microplastic particles per kg dry weight.

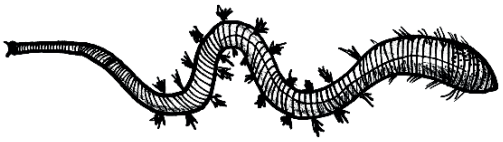
References: Karlsson 2015



Annelids:

Lugworms *(Arenicola marina)* ingested an average of 1.2 (± 2.8) microplastic particles per g wet weight.

References: Van Cauwenberghe et al. 2015



Mollusca:

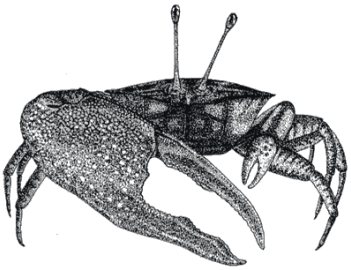
Several studies confirmed contamination of field-collected bivalves. *M. edulis* collected in Europe contained on average 0.2 to 0.5 microplastic particles /g wet weight, mussels sampled in Canada contained 34 to 178 microplastic particles/mussel, Humboldt squid contained plastic pellets. Microplastic has also been found in commercially sold oysters cultured on the eastern Pacific and in several species of commercial bivalves in China. For more detail on commercial shellfish, please refer to Chapter 5.

References: Braid et al. 2012; De Witte et al. 2014; Mathalon and Hill 2014; Van Cauwenberghe and Janssen 2014; Li et al. 2015; Rochman et al. 2015a; Van Cauwenberghe et al. 2015

Crustaceans:

These do not include copepods, which are discussed above. Microplastics have been found in Gooseneck barnacles, *Lepas spp,* Brown shrimp *Crangon crangon* and Norway lobster *Nephrops norvegicus.* These studies found up to 30 particles (majority <1 mm) per individuals.

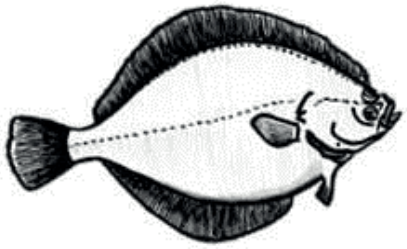
References: Murray and Cowie 2011; Goldstein and Goodwin 2013; Devriese et al. 2015.



Fish:

A large variety of pelagic, demersal and estuarine fish species have been documented to ingest microplastics. The size of microplastics ingested by fish has been reported from 0.1 mm to 5 mm. Particles reported include fibres, fragments, films and pellets. For example, estuarine fish affected include catfish Ariidae, (23% of individuals examined) and estuarine drums, Scianenidae (7.9% of individuals examined). Similarly, 13.4% of Gerreidae contained microplastic in their stomachs. For more information on commercially targeted species, see Chapter 5.

References: Carpenter et al. 1972; Karter 1973, 1976; Boerger et al. 2010; Davison and Ashe 2011; Possatto et al. 2011; Dantas et al. 2012; Ramos et al. 2012; Gassel et al. 2013; Lusher et al. 2013; Choy and Drazen 2013; Foekema et al. 2013; Kripa et al. 2014; Sulochanan et al. 2014; Collard et al. 2015; Avio et al. 2015; Lusher et al. 2015a; Neves et al. 2015; Rochman et al. 2015a; Romeo et al. 2015



Sea turtles:

Juvenile Green turtles *(Chelonia mydas)* stranded in Rio Grande do Sul, Brazil were found to contain up to 11 plastic pellets in their stomachs.

References: Tourinho et al. 2010

Seabirds:

Many species of seabirds are reportedly contaminated by plastic (see Figure 4.2). Nearly 50 species of Procellariiformes were found with microplastic in their stomachs. Ingested microplastic appeared to comprise primarily of plastic pellets and fragments.

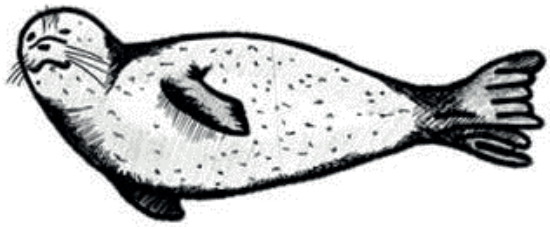
See Table AIII.3 in the appendix for detailed information.

References: Colabuono et al. 2010; Tourinho et al. 2010; Avery-Gomm et al. 2012, 2013; Kühn and van Franeker 2012; Lindborg et al. 2012; Rodrigues et al. 2012; Bond et al. 2013, 2014; Codina-Garcia et al. 2013; Tanaka et al. 2013; Acampora et al. 2014

Marine mammals:

Microplastic was found in stomachs (11%, n = 100) and intestines (1%, n = 107) of harbour seals *(Phoca vitulina).* It was also found in True's beaked whale *(Mesoplodon mirus)* and in the stomach of a Humpback whale *(Megaptera novaeangliae).*

References: Bravo Rebolledo et al. 2013; Besseling et al. 2015; Lusher et al. 2015b ENVIR. POL



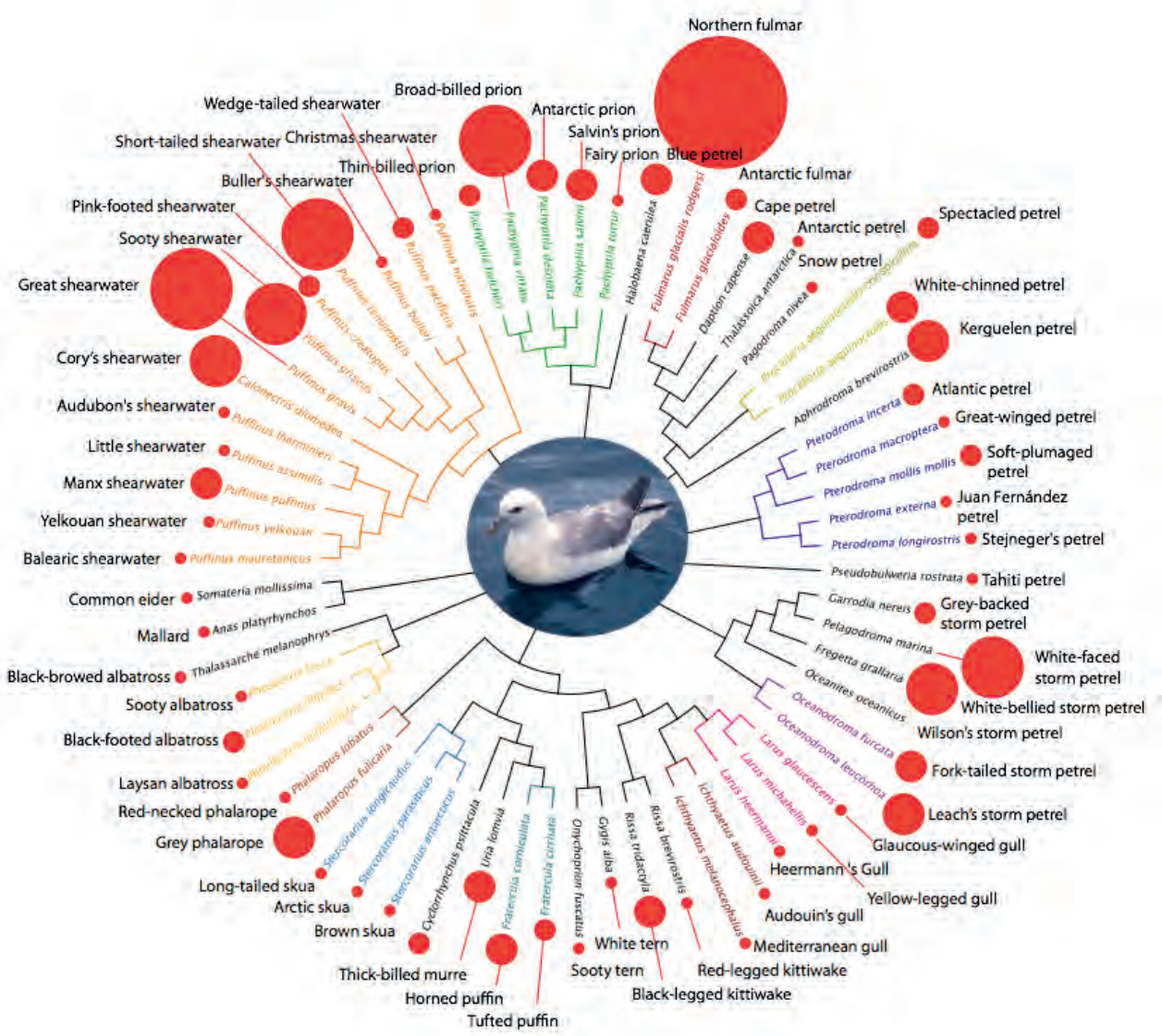


Figure 4.2 *Species of seabirds that have ingested microplastic debris. This figure shows a genealogy of seabirds found with ingested microplastic in their guts shown with both common and Latin names based on cytochrome b genes inferred using the neighbour-joining distance method in Geneious R8 (Biomatters Ltd., Auckland, New Zealand). The tree, based on an alignment of 63 different species and 411 homologous nucleotide positions, illustrates circles proportional to the number of field studies conducted for a given species. The image in the centre is the Northern Fulmar (Fulmarus glacialis), the most intensively studied seabird to date with respect to microplastic ingestion. For more quantitative data refer to Table AIII.3 in the Appendix. Photo credit: © 2013 Simon J. Tonge*

The above examples show that many marine organisms are interacting and consequently contaminated with microplastics. This raises concerns regarding physical and chemical impacts related to ecologically relevant amounts and types of microplastics in marine habitats. Physically, microplastics can perforate the gut, cause organisms to feel full or even translocate outside the gut and cause cellular damage (Browne et al. 2008; Gregory et al. 2009; von Moos et al. 2012). Chemically, microplastics may be a source of toxins to wildlife at levels that are harmful. The next section summarizes what we currently understand about impacts to marine organisms.

4.3 Impacts of microplastics on marine organisms

4.3.1 *Impacts and the level of biological organization*

The science relevant to the impacts of microplastic debris in the marine environment is still in its infancy. While we have been measuring quantities and impacts of larger plastic debris for decades, we only began investigating the science of microplastics in depth over the last decade. As such, we are only beginning to understand impacts of microplastics on marine organisms.

For several environmental stressors, especially during the early stages of research, effects are only demonstrated at lower levels of biological organization

(e.g. molecular, cellular, organism; Underwood and Peterson 1988; Adams et al. 1989). For microplastic debris, this is the case in that the majority of current knowledge remains at these lower levels. Moreover, many examples of demonstrated impacts are from laboratory rather than field studies. Although ecological impacts are generally considered those relevant to higher levels of biological organization (e.g. populations, assemblages, species and ecosystems), understanding responses at lower levels of biological organization can provide insight into causal relationships between stressors and their effects at ecological levels (Adams et al. 1989; Browne et al. 2015b). As such, they are relevant. Below, we include examples of impacts that have been demonstrated across several levels of biological organization from laboratory and field studies. This information, in addition to the exposure pathways and concentrations of microplastics in various marine organisms above, can be useful for risk assessment and to help design ecologically relevant experiments measuring the impact of microplastic to organisms in the environment.

4.3.2 *Impacts demonstrated in laboratory experiments*

As mentioned above the majority of evidence regarding impacts of microplastics on marine organisms comes from laboratory studies. These studies have generally been on bivalves, crustaceans, annelids or fish with unrealistically high concentrations of microplastics compared to the natural environment. Below, we summarize some of the experimental work that has been done for different taxa (Figure 4.3; see Appendix Table AIII.1 for more detailed information).

At the bottom of the food chain is the plankton. For phytoplankton, there have been a few studies that looked for impacts of microplastics. One study found that the exposure of phytoplankton to microplastic did not produce adverse effects (Long et al. 2015). Another study demonstrated that charged PS nano-sized plastics (0.02 µm) can sorb to microalgae, inhibiting microalgal photosynthesis and consequently reducing population growth and chlorophyll concentrations in the green alga *Scenedesmus obliquus* (Bhattachyra et al. 2010). Lastly, one study demonstrated that microalgal growth of Dunaliella *tertiolecta* was negatively affected by uncharged polystyrene particles (0.05 µm), but only at high concentrations (250 mg/L), and the PS beads did not affect microalgal photosynthesis (Sjollema et al. 2016). For zooplankton, microplastic can adhere to external and internal body parts, including the alimentary canal, furca and urosome, and swimming legs of copepods (Cole et al. 2013). The copepod, *Calanus helgolandicus,* ingested and egested microplastics (20 µm in size; polystyrene; 75 particles per ml for 23 h) which caused effects on fecundity, survival and feeding (Cole et al. 2015). Lee et al. (2013) ran an experiment using polystyrene microbeads that were 0.05, 0.5 and 6 µm in diameter. They demonstrated mortality in copepods after exposure to 12.5 µg/mL and 1.25 µg/mL concentration of 0.05 µm size microplastic. The study also demonstrated a decrease in fecundity for 0.5 and 6 µm PS beads at 25, 12.5 and 1.25 µg/mL.

For other invertebrate taxa, some experimental work has also been done. In echinoderms, a toxic effect on the embryonic development of the green sea urchin *(Lytechinus variegatus)* was observed as a result of exposure to PE microplastic particles (Nobre et al. 2015). However, Kaposi et al. (2014) reported only a limited threat to the sea urchin *Tripneustes gratilla* using more environmentally relevant concentrations of microplastic. For Annelids, some experimental work has been done with *Arenicola marina,* an important prey source for many marine species due to its high lipid content. *A. marina* selectively feeds in sediment and will ingest microplastic particles. Long-term chronic exposure to environmentally relevant levels of PS (400 to 1300 µm) resulted in a dose dependent reduction in feeding capacity (Besseling et al. 2013). Increased microplastic concentration in sediments (0.02%, 0.2% and 2%) significantly increased the metabolic rate of individuals. Bioturbation was also affected, smaller and fewer casts were produced by organisms with microplastic present in sediment (Green et al. 2016). Reduced feeding, weight loss and oxidative stress were also observed (Browne et al. 2013; Besseling et al. 2013). For crustacea, no negative effects have been observed, but translocation between tissues was demonstrated. A 2-month exposure resulted in PS microplastic (180 to 240 µm) in the gills stomach, and hepatopancreas of crabs (*Uca rapax;* Brennecke et al. 2015)

A lot of the toxicological work has been done with molluscs. A number of lab experiments have been performed to assess the potential adverse effects of microplastic in *Mytilus edulis* (see Appendix Table AIII.1). Wegner et al. (2012) demonstrated increased production of pseudofaeces and reduced filter-feeding activity after exposure to 30 nm polystyrene nano-sized plastic particles (0.1, 0.2 and 0.3 g/L). In other studies using different sizes and concentrations of microplastic particles, no significant reduction in feeding activity or decrease in energy budget were demonstrated (Browne et al. 2008; Van Cauwenberghe et al. 2015). Von Moos et al. (2012) observed significant effects from exposure to microplastic of a larger size range (>0 to 80 µm; 2.5 g/L). The microplastic accumulated in epithelial cells of the digestive system (more specifically the digestive tubules), where they induced a strong inflammatory response accompanied by notable histological changes after only 3 hours of exposure. With increasing exposure times, the measured biological effects became more severe.

For vertebrates, laboratory studies assessing effects have been conducted with different species of fish. de Sá et al. (2015) observed a significant decrease in the predatory performance of *P. microps* (common goby) after exposure to microplastics. Oliviera et al. (2013) fed 1 to 5 µm polyethylene microplastics to fish at concentrations of 18.4 and 184 µg/L and observed an increase in AChE activity. Cedervall et al. (2012) fed nano-sized polystyrene (1 to 100 nm; 0.01% w/v) to fish and observed weight loss, changes in metabolic performance and changes in feeding behaviour. Rochman et al. (2013a, 2014a) fed polyethylene microplastic (<0.5 mm) to Japanese medaka at 0.001% w/v and observed changes in gene expression related to endocrine disruption and liver toxicity.

Although limited, impacts from microplastics have been observed in the laboratory (Figure 4.3). The majority of published effects include sub-lethal responses of organisms to microplastics. Microplastics can reduce the health, feeding, growth and survival of organisms from lower trophic levels.

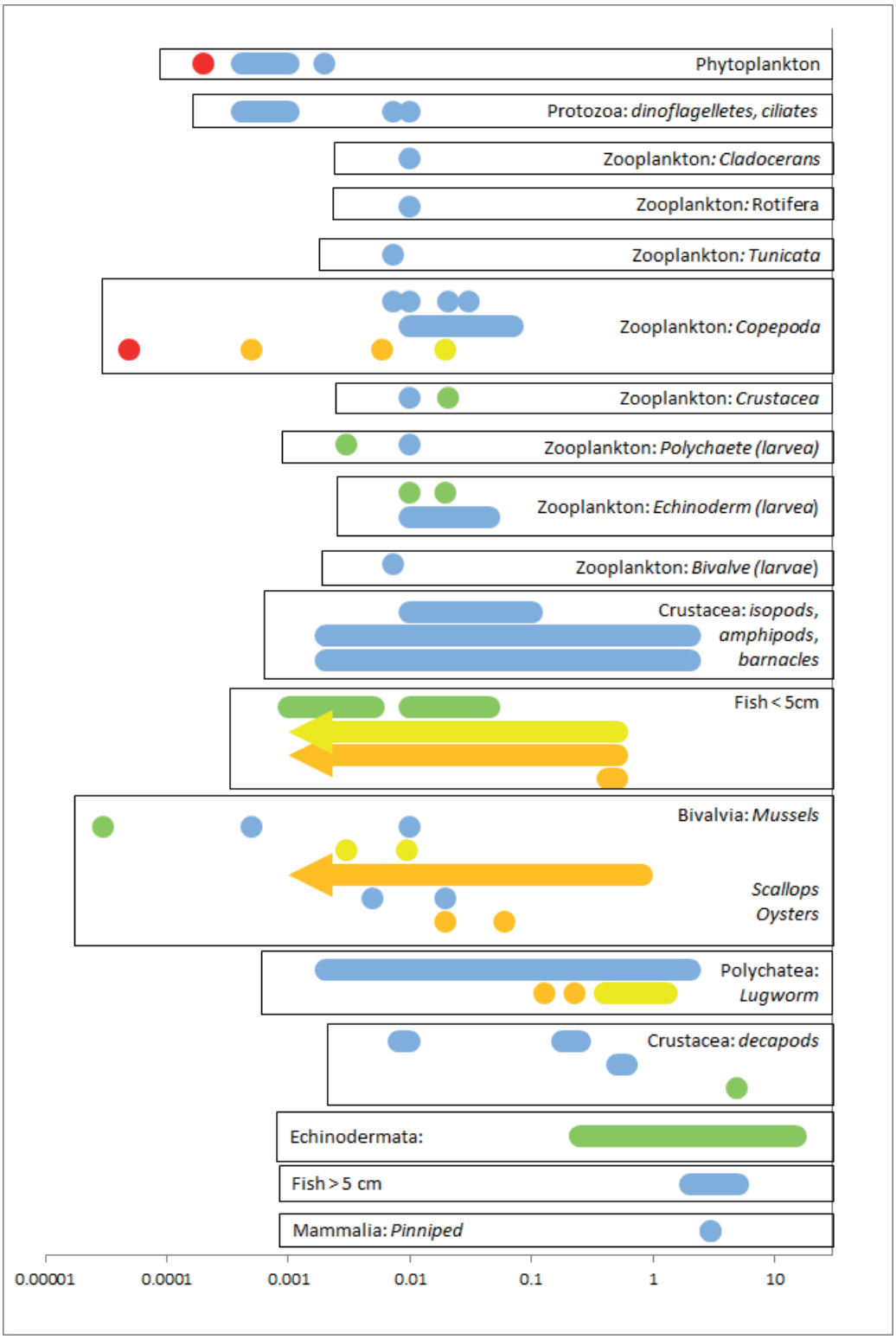


Figure 4.3 *A summary of laboratory experiments published up to November 2015, in which marine organisms were exposed to high concentrations of microplastics. Details on the studies included can be found in Table AIII.1 of the Appendix. The x-axis shows the size of the plastic debris in mm on a log-scale. The severity of the effect is rated from blue to red, where: blue = no observed effect, interaction occurs but organism is unaffected (or interaction ends following egestion); green = minor effect, interaction occurs for short or long period of time, some energy loss associated with interaction; yellow = marginal effect, interaction causes reduction in function, or transfer between tissues; orange = critical effect, interaction causes reduction in function and subsequent biological effect; red = major effect, biological processes affected leading to mortality. Where more than one interaction was observed per study, the most severe effect is reported. Where no minimum size range was reported, arrows pointing to the lowest size range is displayed*

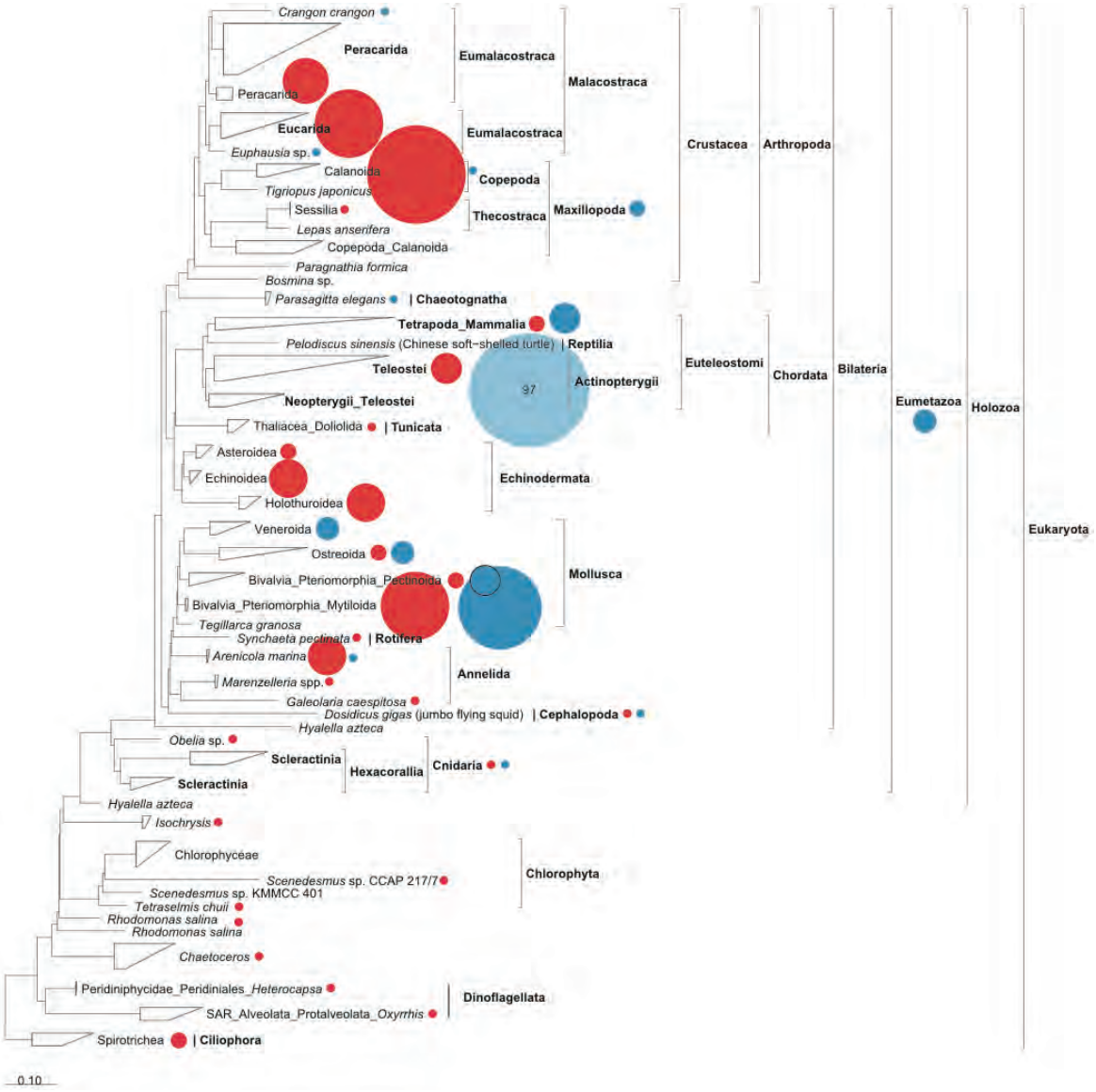


Figure 4.4 *Eukaryotic Tree of Life with major phyla represented that have been the targets of laboratory (RED) and/or field (BLUE) microplastic exposure/ingestion studies published as of November 2015. Relative proportions of studies are indicated by the size of the spheres directly to right of the relevant taxonomic level. The tree is based on a pruned version of the Silva-ARB version 123 small subunit ribosomal RNA (rRNA) gene reference tree (http://www.arb-silva.de/). If a given species used for a study did not possess a sequenced rRNA gene, then the next higher taxonomic level was chosen to illustrate the study. The tree highlights the need for additional fieldwork on microbial species (occupying the branches on the lower portion of the tree) and also illustrates that the largest field efforts to date have focused on bony fishes (Teleostei) and mussels (Mytiloida). Fish (Teleostei) field studies were too numerous and thus not shown to scale in the figure. Please refer to Figure 5.4 for an expanded view of field studies employing fishes including sharks and rays (not included on the tree). Likewise, bird field studies were not included in this figure but they were highlighted earlier in Figure 4.2*

4.3.3 *Evidence from the field*

Compared to evidence from laboratories, there is very little direct evidence for physical impacts of microplastic in nature. More is understood about the impact of macroplastic debris on organisms than microplastic debris in the marine environment. The only study that we are aware of testing impacts from microplastic specifically in nature showed that in the North Pacific Subtropical Gyre, the increasing population of *Halobates sericeus,* a marine insect, was linked to the increasing concentrations of microplastics in the region (Goldstein et al. 2012). Future field research is thus imperative to truly understand impacts to wildlife.

4.3.4 *Summary of taxa included in recent research*

This section described several studies that have demonstrated impacts from microplastics across a range of taxa and levels of biological organization. Below we highlight the taxa that have been included in new research, as well as how the evidence informs ecological impacts from microplastic debris.

Diversity of taxa studied

The majority of studies conducted in the laboratory are with molluscs, crustaceans (including copepods) and fish. This is likely due to ease of experimentation with these animals in a laboratory. For field studies, the taxa are more diverse and include many species of vertebrates and invertebrates. Moreover, the number of studies that measure interactions and impacts from microplastics with eukaryotic single-celled species (microbial eukaryotes) are far fewer than those that have targeted multicellular species, regardless of size. Most of the laboratory studies targeting microbial eukaryotes targeted photosynthetic eukaryotes (chlorophytes, haptophytes, photosynthetic dinoflagellates (formerly referred to as dinophytes), and diatoms) with the exception of a few studies that looked at particle ingestion by the dinoflagellate *Oxyrrhis marina* and ciliates *Strombidium sulcatum* and *Tintinnopsis lobiancoi.* As nano-sized particles continue to raise concerns with respect to environmental impacts, it will be important for research to begin focusing on a broader spectrum of the microbial members that constitute the majority of the biomass in the ocean and impact the microbial food web and base of the food web. No studies to date have looked at microbial ingestion of microplastic/nanoplastics in the field.

Ecological impacts

It is clear from above that there remains little demonstrated evidence regarding ecological impacts of microplastic debris. In this report, we did not have the capacity to systematically review the existing peer-reviewed literature so we discuss the results of a recent review that did, but for marine debris in general. To evaluate the weight of evidence regarding the ecological impacts of marine debris (including both plastic and non-plastic debris), a recent study (Rochman et al. 2015b) systematically and critically reviewed relevant literature regarding effects of microdebris (plastic and other) at several levels of biological organization, spanning the fields of medicine, biological oceanography, conservation biology, toxicology and ecology, asking the question: What are the demonstrated impacts of microdebris including microplastics? For each study, they recorded the size classes of debris, the level of biological organization, whether an impact was demonstrated and the nature of the impact. For many papers, impacts were discussed at multiple levels of biological organization and sizes of debris. Overall, the study found evidence of 175 demonstrated impacts from microdebris, 78% of which were caused by microplastic debris. In total, the study found numerous impacts at suborganismal levels, several at the organismal level demonstrating clear evidence that marine debris can be the cause of death in individual organisms and little at the ecological levels demonstrating that marine debris can alter assemblages. Thus, their findings do demonstrate impacts from microplastic debris, but mostly highlight the need for an improved understanding of ecological impacts of microplastic before any clear general ecological conclusions could be reached. A large reason for this is because researchers are not designing experiments that truly measure ecological impacts from microplastic debris.

In addition to physical impacts of the microplastic particles themselves, microplastic is associated with a complex mixture of chemicals that may transfer to an animal upon exposure. Many of these chemicals are considered as priority contaminants by governments because they are persistent, bioaccumulative and/or toxic (Rochman et al. 2013b). As such, it is important to also discuss impacts related to the mixture of chemicals associated with microplastic debris.

4.4 Impact of plastic-related chemicals

4.4.1 *Concentration of chemicals associated with microplastic in the environment*

A complex mixture of chemicals is associated with microplastic debris. Chemicals in this mixture include those that are ingredients of plastic materials (e.g. monomers and additives), byproducts of manufacturing (e.g. chemicals released during the combustion of the raw material petroleum) and/or chemical contaminants in the ocean that accumulate on plastic from surrounding environmental media (e.g. persistent organic pollutants (POPs) and metals in ambient water or air). Two recent non-targeted screening analyses looking at the chemicals associated with plastic debris, detected a total of 231 to 251 organic compounds on plastics, including hydrocarbons, UV-stabilizers, anti-oxidants, plasticizers, flame retardants, lubricants, intermediates and compounds for dyes and inks (Gauquie et al. 2015; Rani et al. 2015).

Since Carpenter et al. (1972) first reported PCB contamination on polystyrene microplastics in the early 1970s, there has been a series of studies monitoring the mixture of chemicals in floating, beached or ingested plastic particles. The reported concentration ranges of target chemicals are summarized with other information in Table 4.1 (see Table for reference). These studies report concentrations of targeted chemicals including persistent organic pollutants (POPs), polychlorinated biphenyls (PCBs), dichlorodiphenyltrichloroethane (DDT), polycyclic aromatic hydrocarbons (PAHs), organochlorine pesticides (hexachlorocyclohexanes (HCHs), hexachlorobenzene (HCB), chlordanes and mirex), brominated or fluorinated flame-retardants (polybrominated diphenylethers (PBDEs), hexabromocyclodecanes (HBCDs) and perfluoroalkyl acids (PFAAs)) and additive ingredients (bisphenol A (BPA), nonylphenol (NP) and octylphenol (OP)). A few studies also targeted metals. The plastic particles analysed were generally pre-production resin pellets in the size range of 1 to 5 mm or plastic fragments with a size range of up to tens of millimetres. Most studies analysed polyethylene (PE) and (or) polypropylene (PP) plastics and a few studies analysed other types of plastics (Table 4.1).

The concentration of this mixture of chemicals in and on microplastic is governed by many factors, including whether the chemical was added during manufacturing or sorbed from the environment, physicochemical properties of plastics and chemicals, the size of plastics, concentration in the surrounding water, and other environmental factors (e.g. pH, temperature). For example, a recent study compared concentrations

of PCBs and PBDEs in small (0.3 to 1 mm) and large (1 to 5 mm) microplastic from Tokyo Bay, Japan and the pelagic waters of the Pacific Ocean (H. Takada, unpublished results). PCB concentrations in smaller polyethylene microplastic from Tokyo Bay were on the order of hundreds of ng/g, while those in the open ocean were a few ng/g. The concentration ranges and spatial patterns (i.e. urban coast >> open ocean) were similar to those observed in previous International Pellet Watch (http://www.pelletwatch.org/). Moreover, no PBDE congener 209 (BDE-209) was detected in smaller sized microplastic in the open ocean, whereas BDE-209 was detected in this same size range from the estuary of Tokyo Bay. This suggests that the increase in surface area on smaller microplastic could facilitate the leaching and photodegradation of BDE-209 faster than in larger sized microplastic. Still, additive chemicals (e.g. PBDEs and NP) have been found at large concentrations on some particles of plastic in pristine and open oceans, suggesting there may be a greater risk of plastic being a source of chemical additives (e.g. PBDEs, NP) in pristine and remote areas than the absorbed chemicals (e.g. PCBs, DDT).

Ranges of concentrations found from various studies are listed below in Table 4.1. The concentration ranges provided can be used in risk assessment and to help design ecologically relevant laboratory experiments for measuring the chemical impact of microplastic to organisms. Note, congener-specific data can be extracted from the cited literature.

Table 4.1 Summary of chemical concentrations in plastic particles collected from marine environments

Chemicals[a]	Polymer type	Size (mm)	Concentration Min - Max (ng/g plastics)	Concentration Median of maximum[b] (ng/g plastics)	Reference
PCBs	PE, PP, PS	0.1-35	ND[c]-5,000	240	1[d], 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14, 15, 16
DDTs	PE, PP, PS	-	ND-7,100	88	1, 3, 5, 12, 13, 14, 15
HCHs	PE	1-5	0.14-112	20	10, 12, 13
Chlordanes	PE, PP	-	4.29-14.2	-	3
HCB	PE, PP	-	12.4-17.5	-	3
Mirex	PE, PP	-	6.48-14.6	-	3
PBDEs	PE, PP	~35	ND-16,444	412	8, 16, 17, 18
HBCDs	PS	1-5	0.06-512	-	19
PFAAs	-	2-6	0.01-0.18	-	20
PAHs	PE, PP, PS	1-35	ND-12,000	1,335	1, 5, 6, 8, 10, 12, 14, 16, 21
BPA	PE, PP	~35	ND-729.7	284	8, 16
NP	PE, PP	1-35	ND-16,000	2,660	8, 11, 16
OP	PE, PP	~10	ND-154	40	8

[a] See text for abbreviations
[b] Median of the maximum values reported in the each study from the literature
[c] Not detected
[d] Numbers refer to: 1=Antunes et al. (2013); 2=Carpenter et al. (1972); 3=Colabuono et al. (2010); 4=Endo et al. (2005); 5=Frias et al. (2010); 6=Gauquie et al. (2015); 7=Gregory (1978); 8=Hirai et al. (2011); 9=Hosada et al. (2014); 10=Karapanagioti et al. (2011); 11=Mato et al. (2001); 12=Mizukawa (2013); 13=Ogata et al. (2009); 14=Rios et al. (2007); 15=Ryan et al. (2012); 16=Teuten et al (2009); 17=Tanaka et al. (2013); 18=Tanaka et al. (2015); 19=Al-Odaini (2015); 20=Llorca et al (2014); 21=Karapanagioti et al. (2010)

4.4.2 *Transfer of chemicals from microplastic to marine organisms*

One question often asked by policy makers is whether or not these chemicals can transfer from plastic to marine organisms. This section describes the processes by which transfer may occur and the current state of the evidence through 2015 from laboratory, field and theoretical studies addressing bioaccumulation. Lastly, this section discusses the parameters we need to include to further understand plastics as a source of chemicals to the environment and some examples of how current information can be used to guide estimates of chemical transfer for risk assessment.

Processes of transfer

There are several processes by which microplastics can act as a source of chemicals to marine organisms. It is important to note that here we are discussing more than via the ingestion of plastic. These chemicals may be transported directly via ingestion of plastic. They may also be transferred indirectly if chemicals leach from microplastics into water and are taken up by an organism via indirect bioaccumulation, also called bioconcentration, or if a predator eats a prey item that is contaminated with plastic (Figure 4.5).

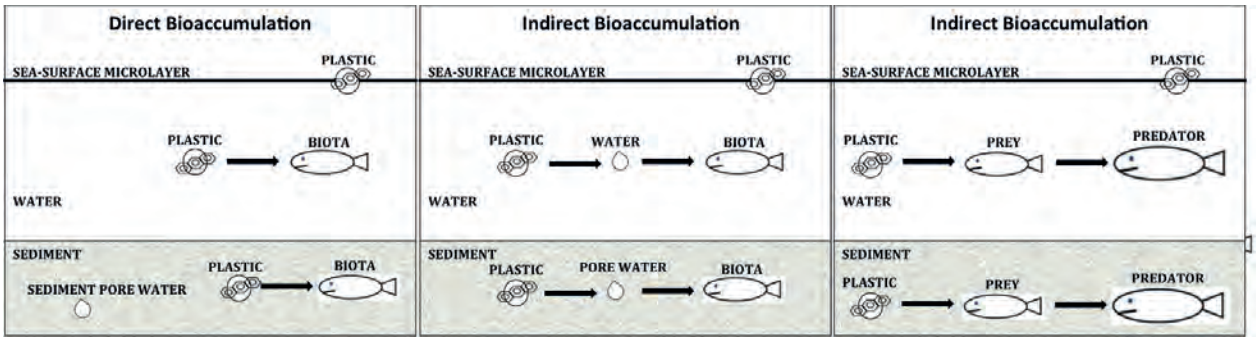


Figure 4.5 *Mechanisms for the bioaccumulation of chemicals from plastic debris. The diagram depicts different pathways for how chemicals may transfer from plastic to biota in aquatic habitats. Bioaccumulation may occur directly via ingestion of plastic (left) or indirectly via desorption of chemicals from plastic into other environmental media followed by bio-concentration from the water (middle) or trophic transfer via a prey item that is contaminated with plastic which may lead to biomagnification, i.e. increasing levels in predators (right). Note, this figure does not include the many other sources of industrial chemicals to wildlife and only shows how microplastic may contribute to the transfer of chemicals in aquatic systems*

There is no doubt that these processes *can* occur, but there is uncertainty about the extent that they *do* occur in nature. There is a separate discussion regarding the "importance" of plastic as a source of chemicals to organisms, i.e. the relative contribution of microplastic as a source of chemicals compared to other sources such as surrounding water and/or prey. This discussion is based on fugacity gradients, which according to first principles in environmental chemistry will drive the direction that the chemical moves, i.e. from plastic to animal or vice versa. The various matrices, including water, biota, sediment and plastic, will strive to reach equilibrium and thus chemicals in the environment will move in the direction toward equilibrium. Thus, you can imagine a piece of plastic, a fish and the water column that have been in the ocean for 1 year will be at or near equilibrium. Thus, if the fish ingests the plastic, the PCBs may not transfer at all. In other words, the ocean is already contaminated with chemicals that come from a number of different sources. As such, animals can accumulate hazardous chemicals via several processes, including uptake from surrounding water, air or sediment and ingestion of particles in the water and/or their diet (Van der Oost et al. 2003). The introduction of microplastic to the ocean introduces another potential source of additive chemicals and sorbed contaminants from the environment to wildlife (Farrington and Takada 2014). Thus, an animal exposed to microplastic is likely already contaminated with chemicals from other sources and the plastic may not act as a significant additional source of chemical contamination.

Modelling studies are useful to conceptualize these ideas regarding how plastic may be an important source and sink for chemicals in the environment relevant to other media (e.g. diet, sediment, water). Published studies using such models conclude that whether plastic acts as a source of chemicals to animals via ingestion depends on the fugacity gradient between the chemical contaminant concentration in the plastic vs. in the lipid stores of the animal (Koelmans 2015; Koelmans et al. 2013). Thus, when an organism is relatively clean of contaminants, model studies (based upon fugacity gradients) predict that chemicals will transfer from the plastic into the lipid (Figure 4.6c). This may occur when microplastics in the ocean are not in equilibrium and have sporadically large concentrations of additives or sorbed contaminants. Alternatively, if an organism has a greater body burden of chemicals than the introduced plastic debris, the model studies predict (based on fugacity) that the plastic debris will "clean" the lipid (Figure 4.6a). This may occur if an organism in the ocean for example ingests a raw or relatively clean plastic pellet. Lastly, when an animal and plastic have a similar level of contamination, any change in contaminant levels between the organism and the plastic may be negligible in comparison to other sources (Figure 4.6b; Gouin et al. 2011; Koelmans et al. 2013, 2014; Koelmans 2015). Thus, modelling exercises conclude that chemicals from plastic can transfer to animals upon ingestion or the other way around, dependent on fugacity gradients, but generally the transfer is only measurable when (a) plastic is a larger source of chemicals than other media, and (b) there is sufficient fugacity gradient for transfer, and (c) the effect is larger than measurement error and biological variation (Gouin et al. 2011; Koelmans 2015).

Figure 4.6 *The figure above is a simplified depiction describing the scenarios discussed above. The image on the left (a) depicts a scenario where a contaminated fish eats a relatively clean piece of plastic and the chemical moves from the fish to the plastic. The image in the middle (b) depicts a scenario where a contaminated fish eats a contaminated piece of plastic and no transfer occurs. The image on the right (c) depicts a scenario where a relatively clean fish eats a contaminated piece of plastic and the chemicals transfer to the fish*

Evidence of transfer from the laboratory

Laboratory observations suggest that chemicals from plastic can transfer to aquatic animals. Such trends have been described in a number of species including lugworms (Browne et al. 2013), amphipods (Chua et al. 2014) and fish (Rochman et al. 2013a). Some researchers have suggested that chemicals can transfer from plastic to biota using simulated gastric conditions (Bakir et al. 2014; Tanaka et al. 2015) and other studies have demonstrated transfer by exposing lab animals to plastic with different sorbed chemicals, including PBDEs, nonylphenol, phenanthrene and triclosan.

Some studies measured plastic as a source of chemicals relative to other media (i.e. water, sediment and/or food). One study asked if microplastic particles mediated greater transfer of PBDEs to amphipods than seawater. Similar to what theory predicts, animals that were exposed to PBDEs in the presence of clean microplastics had a smaller body burden of PBDEs than those exposed to PBDEs dissolved in seawater alone (Chua et al. 2014). Similarly, a study measured the relative difference in bioaccumulation between sand and microplastic by exposing clean lugworms to microplastic or sand spiked with phenanthrene and nonylphenol. They found that lugworms exposed to chemicals via sand bioaccumulated $>250\%$ more phenanthrene and nonylphenol than animals exposed to plastic (Browne et al. 2013).

Other studies aimed to measure the importance of plastic as a source of chemicals in the presence of a contaminated system (i.e. to try to better understand scenarios in nature). One study exposed contaminated amphipods to microplastic spiked with PBDEs (Chua et al. 2014). They found no significant difference between concentrations of PBDEs in animals exposed to clean plastics (i.e. the microplastics did not "clean" the organisms of PBDEs) versus those exposed to microplastics with environmentally relevant levels of PBDEs, and an increase in PBDEs in amphipods exposed to microplastic with concentrations of PBDEs greater than their starting concentrations (Chua et al. 2013). In another study, fish that were already contaminated with PAHs, PCBs and PBDEs were exposed to plastic with environmentally relevant concentrations of the same chemicals and at concentrations of microplastic relevant to what is found in the subtropical gyres. For PAHs and PCBs, significant transfer of chemicals to fish was not observed. In contrast, the transfer of PBDEs was significantly greater (Rochman et al. 2013a). Another study consisted of tanks with plastic, sediment and worms with concentrations of PCBs all at equilibrium (Besseling et al. 2013). Lugworms exposed to smaller concentrations of plastic had greater concentration of PCBs in their tissues, but lugworms exposed to larger amounts of plastic accumulated similar concentrations of PCBs as lugworms that were not exposed to plastic (Besseling et al. 2013). Differences in conclusions highlight that further research is necessary to determine the importance of plastic debris as a source of chemicals in nature.

Evidence of transfer in the natural environment

In nature, animals are exposed to chemical contamination via multiple sources, and thus it is difficult to demonstrate that plastics are the source of bioaccumulation in wildlife. Still, some researchers who have conducted observational experiments in nature have suggested that burdens of chemical contaminants in wildlife were introduced by plastic debris. Recent studies have looked for associations between plastic debris and bioaccumulation in whales (Fossi et al. 2012, 2014), basking sharks (Fossi et al. 2014), seabirds (Teuten et al. 2009; Tanaka et al. 2013; Lavers et al. 2014; Hardesty et al. 2015; Yamashita et al. 2011) and fish (Gassel et al. 2013; Rochman et al. 2014b).

Some studies suggest that chemicals can transfer qualitatively from microplastics to animals in nature. These studies have simply noted the large presence of plastic debris in the feeding grounds of animals, the presence of plastic in their gut contents and/or plastic-associated chemicals in surrounding media, and the detection of plastic-associated chemicals in the animal of concern (Fossi et al. 2012, 2014; Hardesty et al. 2015 Gassel et al. 2013). Such studies include discussion of phthalates in fin whales (Fossi et al. 2012) and seabirds (Hardesty et al. 2015 and PBDEs in fish (Gassel et al. 2013) and seabirds (Tanaka et al. 2013).

Other studies have aimed to quantitatively demonstrate positive correlations between plastic debris and bioaccumulation of hazardous chemicals. Such results should be taken with caution, as correlation does not always mean causation, and there are many other sources in the environment that may also be correlated. Researchers have found that the concentrations of some PCBs (Teuten et al. 2009; Yamashita et al. 2011; Ryan et al. 1988) and trace metals (Lavers et al. 2014) in seabirds and higher-brominated PBDEs in fish (Rochman et al. 2014b) are positively correlated with plastic debris. It is worth nothing that the study examining fish could not find any significant correlation between plastic debris and the bioaccumulation of bisphenol A, nonylphenols and PCBs (Rochman et al. 2014b).

Overall, it is clear that plastic can be a source and sink of chemicals to animals. What is less clear is the extent to which plastic is a source to wildlife in nature and how it is relevant to risk. As noted above, quantifying plastic as a source of chemicals for bioconcentration and bioaccumulation is difficult to isolate from other sources in nature. There are several parameters that will influence whether or not transfer of chemicals will occur to an extent that causes harm. Chemical transfer will be influenced by external factors such as plastic type, size and amount, concentration and properties (e.g. hydrophobicity, susceptibility to metabolism) of chemicals on the plastic and in the organism, ecology (especially trophic level) and physiology of the animal and the retention time in the animal. As such, it is important that we continue to investigate the issue. In parallel, we can use the existing information to estimate the transfer that we might expect in nature under different scenarios to begin to think about risk.

Estimating chemical transfer

The estimation of chemical exposure to organisms from microplastics is useful in a risk assessment framework, but it is complex and thus requires several assumptions. Published models may be adopted as a framework to estimate chemical transfer (for example, see Koelmans et al. 2013). Values for each variable needed to solve the equations within these models can be gathered from the existing literature, including the studies that are discussed above (as in Koelmans et al. 2016).

For example, most studies provide the range, mean and (or) median value of concentrations of chemicals found in plastic. Most studies also specify concentrations of plastic in the environment and/or in animals which can be used to calculate the mass of ingested plastic. The rate that chemicals transfer from plastic to an animal after exposure to plastic in a laboratory can be calculated using these equations with this information from previous laboratory studies. The receptor organisms can be chosen based on the objectives of the risk assessment. For example, assessments may include lugworms, bivalves (e.g. blue mussels), small fish (e.g. brown goby), large fish (e.g. Atlantic herring) and seabirds (e.g. northern fulmar). Each of these groups is known to ingest microplastics in nature and laboratory studies measuring transfer of chemicals have been conducted on similar organisms.

Such an exercise can be used to roughly estimate the exposure and resulting concentrations in wildlife under different scenarios. But note, as with all modelling exercises, that each will be based on several assumptions and some uncertainty. In nature, there are many factors that will influence the assimilation rate such as leaching, desorption and the partitioning of chemicals between the microplastic and the gut and tissue of organisms. In addition, the physicochemical properties of plastics and target chemicals, metabolic capacity of organisms, retention time of plastics, chemicals and organisms.

4.4.3 *Impacts of chemicals from microplastics on organisms in the laboratory*

Discussions regarding microplastics as a source of hazardous chemicals to wildlife has raised concerns regarding adverse biological effects. While several studies have examined adverse health effects from the ingestion of clean microplastics, as discussed in the previous section, only a few laboratory studies have tested hypotheses regarding the impacts associated with the complex mixture of plastic and sorbed contaminants to organisms. One study found that the combination of PVC with sorbed triclosan altered feeding behaviour and caused mortality in lugworms (Browne et al. 2013). Another study demonstrated that polyethylene deployed in San Diego Bay, CA (i.e. allowing the plastic to accumulate environmentally relevant concentrations of priority pollutants) caused hepatic stress, including glycogen depletion, lipidosis, cellular death and tumour development, in fish exposed to microplastic for a 2-month period (Rochman et al. 2013a). Moreover, fish exposed to the combination of polyethylene and priority pollutants showed signs of endocrine disruption via changes in gene expression and abnormal growth of germ cells in the gonads (Rochman et al. 2014a). In both studies, adverse effects were demonstrated from the plastic alone, but organisms suffered greater effects when exposed to the mixture of plastic with sorbed chemical contaminants (Browne et al. 2013; Rochman et al. 2013a), suggesting that the combination of plastic debris and priority pollutants may be a multiple stressor in the environment.

4.4.4 *Conclusion*

Plastic debris is associated with a cocktail of hazardous chemicals, some unique to plastic debris as additives and monomers and others that are ubiquitous in nature from other sources. As such, plastic debris is often discussed as a source of chemical pollutants to the environment and potentially to wildlife, raising concerns regarding how plastic debris may impact the health of ecosystems. Several priority chemical pollutants are associated with plastic debris. Such chemicals are designated a priority based upon their persistence, toxicity and their ability to biologically accumulate in organisms and magnify in foodwebs (Teuten et al. 2009; Rainbow 2007; Vallack et al. 1998). Ecotoxicological work has shown that priority pollutants can alter the structure and functions of ecosystems. Physiological processes of organisms (e.g. cell-division, immunity, hormonal regulation) can be disrupted, causing disease (e.g. cancer) (Zhuang et al. 2009; Vasseur and Cossu-Leguille 2006; Oehlmann et al. 2009), reducing the ability to escape predation (Cartwright et al. 2006) and altering reproductive success (Brown et al. 2004). Furthermore, priority pollutants can alter interactions among species (e.g. competition; Roberts et al. 2008), which may lead to structural (Roberts et al. 2008) and genetic (Pease et al. 2010) changes in biodiversity (Johnston and Roberts 2009). Thus, further research is needed to understand the extent that plastic debris is a source of chemicals to the marine environment and any ecological hazards that may be associated.

4.5 Nano-sized plastic debris

4.5.1 *Definitions*

Little research has been done looking at the effects of nano-sized microplastics on marine organisms. Our knowledge and understanding is limited to short-term laboratory studies of fish and invertebrate species exposed to high concentrations (GESAMP 2015). In fact, several of the demonstrated impacts in laboratory exposures displayed above were due to nano-sized plastic (see Annex Table AIII.1). In addition, there is a greater body of research measuring effects of nano-sized plastic debris on animals and even humans from the fields of nanotechnology and medical sciences (summarized in GESAMP report 2015). Parallels may apply between the fields of engineered (non-polymeric) nanoparticles (ENP) and nano-sized plastic where particle characteristics (size, surface charge, density, composition, shape, etc) largely affect toxicity. This section first defines nano-sized plastics and its different properties and then summarizes some of the recent findings on the fate and effects of nano-sized plastic on humans and marine organisms.

The definition of nanoplastics is ambiguous and a clear definition of what should be named a "nanoplastic" has not yet been established (Koelmans et al. 2015; Mattsson et al. 2015). In the scientific literature at least two different definitions of nanoplastics have been adopted: i) Nano-sized plastic particles <1000 nm (e.g. Browne et al. 2007; Andrady 2011; Cole et al. 2011); and ii) Nanoplastics <100 nm (in at least one of its dimensions) as defined for non-polymer nanomaterials in the field of engineered nanoparticles (ENP) (e.g. Koelmans et al. 2015; Bergami et al. 2015). Nano-sized plastic (also termed nanoplastic) falls within the definition of microplastic adopted by GESAMP (2015) and according to our definition comprise all polymeric particles <1000 nm (in at least one of its dimensions). They include the synthetic nano-sized plastic with a polymer core and variable functional groups and the polymeric ENPs based on nanotechnology that might exhibit additional properties including non-polymeric nano-scale additives. Therefore, special reference should be made to polymeric ENPs (by definition <100 nm) as they represent a unique group with different properties. In general, nano-sized plastics are potentially more hazardous than micro-sized plastics (Koelmans et al. 2015; Bergami et al. 2015; Mattsson et al. 2015; Della Torre et al. 2014) and their uptake and toxicity will depend on their intrinsic properties such as size and surface charges, that affect their interrelationships and their interaction with exposure media (Bergami et al. 2015). In the case of polymeric ENPs, release of non-polymeric nano-scale additives from the product fragments, as a consequence of possible nano-fragmentation, may further add to the overall hazard (Koelmans et al. 2015).

4.5.2 *Evidence of nano-sized plastic debris in the environment*

Due to limitations in methodology (see Chapter 7), concentrations of nano-sized microplastic in the environment are unknown. Moreover, they tend to flocculate based on their properties, and thus it is difficult to quantify individual particles. This issue is the same for ENPs. However, due to the fate of larger microplastics, we are relatively certain of their presence in marine habitats. Still, without an understanding of the amount in the environment, it is difficult to identify an environmentally relevant dose for experimentation or risk assessment.

4.5.3 *Potential fate and impacts of nano-sized plastic to humans and other biota*

A number of studies have demonstrated that nanoparticle toxicity is extremely complex and that the biological activity of nanoparticles will depend on a variety of physicochemical properties such as particle size, shape, agglomeration state, crystal structure, chemical composition, surface area and surface properties (e.g. Hofmann-Amtenbrink et al. 2015). Particle characterizations can affect the likelihood of sorption properties, uptake and effects. For aquatic behaviour of nanomaterials such as polymeric ENPs, homo- and hetero-aggregation are important processes to consider (see Koelmans et al. 2015). Particle aggregation leads to a reduced surface area to volume ratio and new surface structures (Mattsson et al. 2015). Photooxidation and photoreduction affect coatings, oxidation state, generation of oxygen species and persistence. Particles may interact with natural organic materials such as proteins, forming 'bi-molecular corona' that may affect the behaviour of the material, including surface charge, aggregation state and reactivity, thereby affecting transport, bioavailability and toxicity (for review see Mattsson et al. 2015).

Nano research has documented that ENPs primarily are transported over the cell membrane via endocytosis, and thus may serve as a cellular-vector (Trojan horse) for other chemicals or nano-additives (carbon materials and metal ions) (see GESAMP 2015; Mattsson et al. 2015; Galloway 2015). There is evidence that similar effects can occur with the nano-sized plastic in marine organisms. The capability of marine organisms to translocate assimilated small plastic particles within their tissues has been demonstrated (von Moos et al. 2012). These authors exposed blue mussel to HDPE powder in a size range of >0 to 80 µm and demonstrated intracellular uptake of microplastic particles into the cells of digestive tubules and transition into cell organelles of the lysosomal system. However, it remains unclear whether the "Trojan horse" mechanism can work for chemicals associated with nano-sized microplastics, and the resulting chemical effects from translocated particles into cells and tissues will require further research.

It is plausible that under environmental conditions this defence mechanism would deliver plastic particle-associated POPs and additive chemicals to different tissue types and locations than those resulting from uptake from food and water. Given the long residence time of such sequestered particles relative to the lifetime of the organism, even slow chemical release may cause low but chronic delivery within the animal (see GESAMP report, 2015). This unstudied vector effect may provide a unique process to deliver chemicals to specific organs, especially for very small plastic particles that can cross membranes, and should be an important focus for future studies (see also Syberg et al. 2015; GESAMP 2015; Koelmans et al. 2015). Nano-sized plastics exhibit strong sorption affinities for toxic compounds (Velzeboer et al. 2014; Mattsson et al. 2015).

Effects of nano-sized plastic particles on a variety of marine organisms have been demonstrated in laboratory experiments (see elsewhere in this report and GESAMP 2015). Several of these studies have shown that uptake and toxicity depend on the intrinsic properties of the particles, such as size and surface charges that affect their interaction with exposure media (Della Torre et al. 2014). In addition, a number of recent studies have demonstrated effects of PS nanoparticles on feeding, behaviour and physiology of early life stages, such as brine shrimp (Bergami et al. 2015) and sea urchins (Della Torre et al. 2014; Canesi et al. 2015). The study of Bergami et al. (2015) is highlighted below. These authors studied the effects of 40 nm anionic carboxylated (PS-COOH; negatively charged) and 50 nm cationic amino (PS-NH2; positively charged) polystyrene nanoparticles (PS NPs) on brine shrimp *(Artemia franciscana)* larvae. PS-COOH NPs were massively sequestered inside the gut lumen and this likely limited

food intake. Likewise, PS-NH2 (5-100µg/ml) accumulated in larvae (48h) but also adsorbed at the surface of sensorial antennules and appendages probably hampering larval motility. This study demonstrates the bioavailability of nano-sized PS for planktonic species and also that surface charge of the particles might play a significant role in determining the ultimate effect.

ENP based research showed that nanoparticle interactions with biological systems can stimulate inflammatory or allergic reactions and activate the complement system. Nanoparticles can also stimulate immune response by acting as adjuvants or as haptens, and cause immunosuppressive effects (Kononenko et al. 2015). Recently, similar immunological effects have been reported for micro- and nanoplastics interactions with marine invertebrates (Avio et al. 2015; Canesi et al. 2015). The latter study investigated the *in vitro* effects of PS-NH2 in hemocytes of the blue mussel *(Mytilus edulis)* and demonstrated that in mussels the immune function can represent a significant target for PS-NPs. In *Mytilus hemocytes,* PS-NH2 affected several immune parameters and induced pre-apoptotic processes (Canesi et al. 2015).

The above studies illustrate the potential of nanoplastics to affect plankton and early life stages, to decrease biological fitness (through immunosuppression) and reproductive and predator avoidance behaviours, with potential consequences at the population level or food webs over time. However, nano-sized plastic exposure levels and associated effects in the field are currently unknown and the laboratory results based on short-term and high exposure concentrations, hampering extrapolation of these findings to the field situation. The potential impacts of micro- and nanoplastics on human health are described in Chapter 5.4.

As described above, environmental nanoplastics are in fact complex cocktails of contaminants that can act via different modes of action and thus require a multi-stressor risk assessment approach. For example cumulative particle and chemical toxicity effects may occur once NPs have been internalized into tissues and cells. Since all plastics in the marine environment contain multiple potential chemical toxicants, individual and combination effects of the chemicals should be accounted for. Our knowledge on the ecotoxicity and fate of nano and microplastic can benefit from the more advanced areas of (eco) toxicology of ENP and mixture toxicity (Syberg et al. 2015), and a comparison between the two fields should be further encouraged, for example comparing the effects in target species of microplastics in the nano-sized range with those of ENPs. For more information the reader is referred to the reviews by Koelmans et al. 2015; Mattsson et al. 2015 and Syberg et al. 2015.

4.6 Transport of non-indigenous species

4.6.1 *Processes*

Finally, microplastic debris hosts diverse assemblages of species, some distinct from surrounding seawater (Table AIII.4; Zettler et al. 2013), through the creation of novel habitat which may drift long distances and pose an ecological impact via transport of non-native species (Barnes et al. 2005).

The availability of microplastics for settlement has become an important issue, offering opportunities for settlement in areas where natural sources of flotsam are uncommon. From the perspective of a settling organism, microplastic particles are another hard substratum. But microplastic debris is unique from some other substrata, as it has limited movement speed and a potential for widespread dispersion that is much greater than an organism may travel during straight trips on ships.

Many species of marine organisms are known to attach to marine plastics (Barnes 2002; Barnes and Milner 2005; Astudillo et al. 2009; Gregory 2009; Majer et al. 2012; Zettler et al. 2013; Goldstein et al. 2014) and there is some evidence that microplastics translocate non-indigenous species. Although many of these reports refer to plastic pieces larger than 5 mm, they include species that could easily be transported by microplastics. For example, Calder et al. (2014) identified 14 species of hydroids on debris from the March 2011 Japanese tsunami that washed ashore on the west coast of the United States. At least five of these had not previously been reported from that coast. An extensive review of organisms found on floating plastics has been published by Kiessling et al. (2015).

In the smaller size range, microplastics in seawater rapidly develop a biofilm that includes a diverse community of microbes (Figure 4.7). Biofilm formation was visibly apparent on submerged larger plastic items after 1 week (Lobelle and Cunliffe 2011). This biofilm is a miniature ecosystem that includes primary producers, consumers, predators and decomposers and has been described as a “complex, highly differentiated, multicultural community” analogous to “a city of microbes” (Watnick and Kolter 2000). The microbial biofilm encourages the attachment of larger organisms that use chemical and/or physical characteristics as a cue to settle (Zardus et al. 2008; Hadfield et al. 2014). Most of our current knowledge on the development of biofilms on plastic surfaces comes from large settlement plates (>10 cm diameter), but successional dynamics of biofilms on small microplastic surfaces might be different, and should be examined in the future.

The large quantities of plastic debris released into the ocean environment over the past half-century increase the opportunities for the dispersal of pathogens that may pose threats to humans and marine organisms. However, the relative importance of plastic debris compared to natural floating debris is not known. Fish pathogens may attach to plastics (Zettler et al. 2013; De Tender et al. 2015), potentially toxic dinoflagellates have been shown to be transported on plastics (Masó et al. 2003) and studies have demonstrated that bacteria from the genus *Vibrio* are commonly attached to microplastic (De Tender et al. 2015; Zettler et al. 2013) and they have the potential to “bloom” on plastics under the right conditions (Zettler et al. 2013). Plastic debris from the Belgian coast has been found to contain *Vibrio* and potential human pathogens, some distinct from surrounding water and sediment, indicating that plastic debris can act as a distinct habitat and source of these (potential) pathogens (De Tender et al. 2015).

To date, concentrations of these agents at sea remain very low and may not be relevant in terms of risk. However, the behaviour of certain microbes such as vibrios, known to have very fast growth rates, can change when exposed to the gut of a potential host. This may be different in the case of marine species concentrated in aquaculture facilities and ingested microplastics that has been contaminated with harmful microorganisms. In addition, Conn (2014) pointed out that many infectious diseases affect both animals and humans, and aquatic invasive species may be sources of diseases to previously unaffected areas. This study focused on freshwater systems, but a number of infectious diseases can survive in seawater as well.

4.6.2 Impacts

Marine organisms from microbes to invertebrates have always attached to natural floating substrata (macroalgae, feathers, wood, pumice), so one might ask why we should be concerned about plastic transporting organisms? The distribution of plastic is different from that of natural substrata, and plastic has substantially increased the available substratum in oligotrophic open ocean regions, potentially altering the distributions of marine organisms (Goldstein et al. 2012). Another important difference is the longevity of plastic relative to most of the natural substrata, allowing more mature communities to form and persist, perhaps even breed, and thus transport viable populations farther (Kiessling et al. 2015). This may alter connectivity and gene flow and cause effects at the population level.

Figure 4.7 *Scanning Electron Micrograph of the surface of a piece of microplastic particle from the Atlantic Ocean. Cracked surface showing biofilm of attached microbes including heterotrophic bacteria (smallest rods), photosynthetic diatoms (ellipses) and a predatory suctorian ciliate (centre with "tentacles")*

To provide some examples, plastic pellets act as an oviposition site for marine insects such as *Halobates micans* and *Halobates sericeus* (Goldstein et al. 2012; Majer et al. 2012), having a positive effect on the population size and dispersal of this species. Moreover, large abundances of a monospecific foraminiferal assemblage of the benthic foraminiferan, *Rosalina concinna,* were among the rich fauna found on floating microplastics sampled in the northwestern Mediterranean Sea (Barras 2014). This very rare foraminiferal taxon with a planktonic (Tretomphalus) stage is favoured by sexual generation, producing large floating chambers before the release of gametes when surface waters are at temperatures above 18°C. *R. concinna* was found at a density of about 20 individuals per 100 cm^2, comparable to its density on natural substrata. Its ability to colonize floating microplastics leads to a significant extension of the available niches, which could substantially modify the dispersal efficiency of this highly opportunistic taxon and enable a benthic species to colonize the pelagic environment. The dynamics of hard-substratum-associated organisms may be important to understanding the ecological impacts and dynamics of floating plastic on these species but also the connectivity between the various compartments of the marine environment. Lastly, Duarte et al. (2012) pointed out that the increase in human structures in the ocean may be contributing to the increase in jellyfish blooms. The proliferation of microplastic particles provides substratum for attachment and development of jellyfish hydroid life stages. Because pelagic surface waters are typically substratum-limited, microplastics represents another factor that could be contributing to jellyfish blooms. Pyrotag sequences of DNA extracted from microplastics in the Atlantic matched those for a number of jellyfish that have both medusa and attached polyp stages (Amaral-Zettler unpublished).

There is some evidence that microplastics can translocate pathogens, interrupt ecological connectivity and impact population size and dispersion of species. In addition, hygienic contamination of ingested microplastics may pose health risks to marine organisms and humans. Future work is necessary to understand the extent and scale of any impact.

4.7 Conclusions, knowledge gaps and priorities

4.7.1 *Conclusions*

It is clear from the current weight of evidence that microplastic debris has infiltrated nearly all marine habitats and over 100 marine species of wildlife. There is evidence that this debris can impact organisms at many levels of biological organization, with the majority of the evidence at levels that are sub-organismal. Much of this evidence has been demonstrated in laboratory studies typically at high concentrations and there is only limited evidence from nature. Thus, there is a clear need for further research regarding the impacts related to microplastic debris.

4.7.2 *Knowledge gaps*

In this chapter, we were able to discuss evidence regarding impacts from microplastics of all sizes, including nano-sized microplastics. These impacts can be both physical and chemical in nature and can impact individual organisms via exposure and/or populations and communities by acting as a vessel for species dispersal. But, the weight of the evidence remains small and our review of the literature highlights many gaps in our understanding and thus a critical need for continued research. Below, we outline what we think are research priorities.

Research priorities

Understanding the ecological impacts of microplastic debris answers many of the "so what?" questions regarding this environmental issue. While the science around this topic has advanced over the last few years, it remains a burgeoning scientific discipline. As such, there remain many research questions to fill the gaps in our understanding. We recommend:

1. Designing studies that answer hypotheses regarding impacts at higher levels of biological organization (e.g. population, species, assemblage, ecosystem).

2. Designing experiments that are generally more environmentally relevant and measure impacts in situ.

3. Determining what concentration of microplastic debris will have an impact on populations, assemblages and species by

    a. Designing and carrying out experiments in situ or in the laboratory that are ecologically relevant to determine what concentration causes an impact at higher levels of organization.

    b. Designing observational experiments in nature to look for evidence of ecological impacts occurring in wildlife.

    c. Using existing theory and data and applying it to mathematical models.

4. Designing experiments and studies that help us understand the impacts of nano-sized plastic debris on marine organisms.

    a. Designing methods for quantifying concentrations in the environment to inform exposure concentrations and scenarios.

    b. Understanding how nanoplastics behave in the water to inform toxicity.

    c. Measuring how size and charge affect toxicity.

    d. Understanding fate of nano-sized plastic debris in water and organisms and how that affects toxicity.

5. Designing studies that help us understand how and if microplastic moves through foodwebs.

6. Designing studies that further clarify the fate of contaminants to and from microplastic debris (both sorbed chemicals and additive ingredients).

    a. Measuring the role of microplastics as a source of chemicals to the marine environment, including how this differs by polymer type, size and under different environmental conditions (e.g. temperature, pH, salinity).

    b. Measuring the role of microplastics as a sink for chemicals from the marine environment, including how this differs by polymer type, size and under different environmental conditions (e.g. temperature, pH, salinity).

    c. Measuring the role and relative importance of microplastics as a source of chemicals to marine organisms, including how this differs by polymer type, size, amount, chemical type and concentration, taxa, and under different environmental conditions (e.g. contamination, temperature, pH, salinity).

7. Designing studies that measure the impact of chemicals associated with microplastic under environmentally relevant exposure scenarios.

8. Designing studies that measure the impact of the mixture of microplastics and chemicals under environmentally relevant exposure scenarios.

9. Designing studies that help us better understand the role microbes have in facilitating the fouling of microplastic by organisms, the ingestion of microplastic by organisms, and potentially the transformation of toxins.

10. Better understanding the relationship between pathogens and microplastic by

    a. Designing experiments that determine what and how many pathogens associate with microplastic.

b. Designing experiments that measure if there is transfer of pathogens from microplastic into wildlife, and if so any consequential impacts.

11. Establishing threshold levels for physical, chemical and ecological impacts in various habitats and species.

12. Performing risk assessments that help clarify the various ecological impacts that may be a consequence of the widespread contamination of microplastics in the marine environment.

13. Using existing information regarding the amounts of microplastic pollution globally to develop a map that identifies hot-spots for risk and identify priority species.

5 COMMERCIAL FISH AND SHELLFISH

Key points

1. Capture fisheries and aquaculture sectors provide an important protein source that may be negatively affected by microplastic pollution.
2. Microplastics have been documented in finfish, shellfish and crustaceans which are consumed by humans.
3. The impacts of the consumption of microplastics by food fish are unknown; however studies on non-commercial species suggest microplastics have the potential to negatively affect organism health.
4. Although there are no data yet regarding impacts to human health, the occurrence of microplastic in fish and shellfish that are consumed by humans has raised concern about food safety and security.

5.1 Lessons from the first assessment

The GESAMP 2015 report did not specifically address fisheries, aquaculture and aquatic species of commercial value. However, the report's summary of microplastics and potential impacts helps to identify the level of risk that microplastics may pose to these sectors. The report also mentions the potential for fisheries and aquaculture to act as a source of microplastic pollution and the potential for industry to help mitigate the problem. In the first report, there were knowledge gaps presented that are also relevant to how microplastics may affect commercial fisheries and aquaculture, such as information about nanoparticles, contaminant transfer and impacts on organisms at different life history stages.

Microplastics have been detected in a wide range of marine organisms (see Chapter 4), including several commercially important finfish and shellfish species. The impacts of the ingestion of microplastics and their translocation to the most commonly consumed tissues are largely unknown. This chapter summarizes the current status of our knowledge and how microplastics may affect commercial species, aquaculture and fisheries. These issues are relevant to food security, which includes food safety, which has implications for human health.

5.2 Global fisheries and aquaculture sectors

Fish provide an important source of protein globally. In some places, seafood comprises >50% of the total protein consumed. New evidence demonstrates that wild and cultured seafood products are contaminated with microplastics, but we do not yet know to what extent. There is also concern regarding fisheries and aquaculture as a source of microplastics to the marine environment because both sectors use plastics that may degrade/fragment into microplastics. Furthermore, microplastic exposure may be higher in aquaculture systems that use plastics (e.g. nets, pens) compared to wild caught seafood.

Microplastics are a relatively new and emerging contaminant; therefore, the threat of microplastics to fisheries and aquaculture sectors is currently difficult to assess. To improve our understanding of the risk from microplastics to these sectors we have summarized available information on the importance of fisheries and aquaculture, their use of plastics, the exposure of commercial species to microplastics and potential impacts.

5.2.1 *An introduction to seafood*

The world population has become dependent on fisheries and aquaculture resources to meet protein needs, promote health and reduce hunger and poverty. Today, 10% to 12% of the global population relies on fisheries and aquaculture for their livelihood. Such populations tend to live in developing countries, where both employment opportunities and food resources are limited (FAO 2014). Global fish production has grown over the last 50 years and aquaculture has become one of the fastest growing food sectors, providing almost half of all human food fish (FAO 2014).

In 2012, 91.3 and 90.4 million tonnes were produced from capture fisheries and aquaculture respectively with aquaculture reaching another all-time high in terms of value ($144.4 billion) (FAO 2014). Fish is an important food source due to their micro-nutrients. They currently represent 17% of the global animal intake of protein; however, this portion can exceed 50% for some countries (e.g. 51% in Ghana, 65% in Cambodia, 70% in Sierra Leone, 71% Maldives) (FAO 2012).

A declining, but significant portion (21.7 million tonnes) of fisheries production is used for non-food purposes (e.g. not directly consumed by humans). The majority of this production, 75% (16.3 million tonnes), is used for fishmeal and fish oil (FAO 2014). Fishmeal is largely used as a high protein feed, while fish oil is used in the aquaculture industry and for human consumption (FAO 2014). Typically fishmeal is produced from the whole fish, fish remains or other fish products (e.g. heads, tails, bones, offal). Small pelagic oily fish, especially Anchoveta, are the main groups of species used (FAO 2014). As a result, there has been improved regulation and control of feed fisheries that, along with increased demand for fishmeal and fish oil, has contributed to the increase in their value (FAO 2014). Alternatives to replace fishmeal and fish oil (e.g. zooplankton species) are now being sought and more recently the use of these products in compound feeds for aquaculture has been decreasing (FAO 2014).

Box 5.1 What are food fish?

For the purposes of this document we use the FAO definition of "food fish" as follows – finfishes, crustaceans, molluscs, amphibians, freshwater turtles and other aquatic animals (such as sea cucumbers, sea urchins, sea squirts and edible jellyfish) produced for the intended use as food for human consumption (FAO 2014).

In addition to larger scale, industrial producers, small-scale fisheries are a vital part of the health and diversity of global fisheries. The governance of small-scale, traditional fisheries is discussed by Johnson (2006). It is important to note that because of the great diversity of small-scale fisheries there is no single, agreed definition for this subsector (FAO 2015). Small-scale fisheries are particularly important in developing countries for their contributions to nutrition, food security, sustainable livelihoods and poverty alleviation (FAO 2014).

Global fisheries and aquaculture, at both large and small scales, are a vital part of global communities. Microplastic contamination of seafood could pose a threat to these industries. Thus, it is crucial to assess and understand potential impacts to both wild capture and cultured fisheries resources.

5.2.2 *Capture fisheries*

Production and species – The global capture fishery production in 2011 was the second highest ever at 93.7 million tonnes (compared to 93.8 in 1996). China was the largest marine capture fisheries producer with almost three times the volume of Indonesia (second in rank), followed by the US, Peru and Russia (FAO 2014).

The top five ranking marine capture species in 2012 were Anchoveta *(Engraulis ringens),* Alaska Pollock *(Theragra chalcogramma),* Skipjack tuna *(Katsuwonus pelamis), Sardinella* species and Atlantic herring *(Clupea harengus)* (FAO 2014). Twenty-three species comprise approximately 40% of the total marine catch; almost 2/3 of which are small pelagic fish that have large fluctuations due to environmental regimes and, in some cases, are used as raw material in reduction to meal or oil (FAO 2014). Shrimp are also an important capture fishery and hit an all-time high of 3.4 million tonnes in 2012, more than half of which comes from the Northwest and Western Atlantic. The ingestion of microplastics by several of these species and/or similar species (i.e. with a similar life history strategy) has been documented and is described in section 5.3 below, along with potential impacts to fish health.

Fishing gear – A variety of fishing gear and methods are used in industrial and small-scale fisheries, some of which are outlined in Chapter 2. Fishing gear for capture fisheries includes surrounding nets (e.g. purse seines), seine nets (e.g. beach seines), trawl nets (e.g. bottom, otter and midwater trawls), dredges, lift nets, traps and hook and lines (Figure 5.1; Thiele and Prado 2005). Nets and floats are made from a range of plastics include polypropylene, polyethylene, nylon, polyvinyl chloride, polyamide and polystyrene.

Figure 5.1 *Herring caught in seine, British Columbia, Canada taken by Brian Kingzett*

5.2.3 *Aquaculture*

Production and species

The aquaculture industry is increasingly regarded as an alternative to wild capture fisheries to meet the protein demands of a growing human population. As such, it is one of the fastest growing food production sectors and contributed to a record 42.2% of the total 158 million tonnes of food fish from capture fisheries and aquaculture combined (FAO 2014). Asia is the largest contributor, producing 88% of global aquaculture by volume (FAO 2014). Global aquaculture can be divided into inland and mariculture. Mariculture production includes all food fish cultured in the sea, intertidal zones and those with land-based production facilities (FAO 2014). Inland (freshwater) culture has been growing faster than mariculture and now contributes 63% to total farmed food fish production as of 2012. It is estimated that over 60% of the fishery/aquaculture livelihoods are generated by inland systems (FAO 2014).

In 2012, finfish mariculture species contributed 12.6% of the total farmed finfish production. However, their total value comprised 26.9% because of the large proportion of carnivorous species (e.g. Atlantic salmon, trouts) that have higher value per unit compared to farmed freshwater fish (FAO 2014). By species group, world mariculture production was comprised of finfish (22.4%), crustacea (15.9%), molluscs (60.3%) and other species (e.g. sea cucumbers, sea urchins, sea squirts and edible jellyfish, 1.4%). In 2012, a total of 567 aquaculture species (freshwater and marine) were registered in FAO statistics (FAO 2014).

Species farmed and farming practices vary widely between countries and regions. In Norway and Chile the dominant species for 2012 was Atlantic salmon in marine cage culture. Mussels also represented a significant production source in Chile (FAO 2014). Although the Republic of Korea also uses marine cage culture, over half of their food fish production is marine molluscs. In Thailand, half of the production is crustaceans that are mostly marine shrimp species. Indonesia has a large proportion of finfish aquaculture production in brackish-water ponds and has the fourth-largest marine shrimp sector. Finfish aquaculture (mostly milkfish) in marine and brackish water cages dominates production in the Philippines. China produces a wide diversity of species using various farming systems but has a relatively small cage-rearing finfish aquaculture sector (38% of the total volume of national aquaculture production). The type of culture system can be important in regards to microplastics because it can affect the level of microplastic exposure and generation (see Box 5.2).

Box 5.2 Fisheries and aquaculture practices and exposure

Finfish and shellfish are cultured using different techniques throughout the world. Plastics used include, but aren't limited to, polyvinyl chloride, polypropylene, polyethylene and polystyrene. Finfish may be reared on land or open water in ponds, tanks, pens, cages or nets made of plastic materials. Over time, with UV exposure and other weather processes the plastic may become degraded and fragment or shed microplastic particles and fibres that can be ingested. Maintenance practices may also accelerate degradation (e.g. net washing, removal of fouling species).

Examples of culturing systems that use plastics include deep water longline culture where shellfish may be set directly on rope, in bags or in plastic trays (Baluyut 1989) or intertidal culture where shellfish are outplanted in mesh bags or other plastic enclosures (Baluyut 1989) and/or may be covered by anti-predator netting. In fish farming fish are held in nets or pens made of plastics and carnivorous species (e.g. salmon, groupers, snappers) that require fishmeal may be exposed to microplastics directly through their food. No studies have examined if fishmeal contains microplastics; however there is a high likelihood of microplastic contamination in fishmeal given: i) the prevalence of microplastics in fish observed to date (Table AIII.2), including species commonly used in fishmeal (e.g. sardines, anchovies); and, ii) the use of the whole fish in fishmeal production (most plastics have been observed in the gut). As such, direct measurements are warranted. Another consideration is the length of time seafood species spend in close contact with plastic infrastructure (e.g. fishing nets, cages, longlines) and if/how the larger plastic infrastructure can degrade into or generate microplastics.

Habitat can also affect exposure. Environmental microplastic concentrations vary with depth within the water column and within the sediment. This can influence microplastic ingestion as demonstrated in Chapter 4. Further, the interaction between habitats and fishing or aquaculture practices may influence exposure to microplastics. For example a bottom-trawling net may re-suspend sediments with microplastics and potentially have more wear and tear compared to a mid-water trawl fishing net. From a shellfish perspective, mussels reared on longlines made of polypropylene rope may be subjected to higher concentrations of microplastics than scallops reared on hard plastic trays. Fishing and aquaculture practices that use plastic and accelerate degradation processes may put finfish and shellfish at greater risk of microplastic exposure.

Aquaculture practices and systems

Aquaculture practices and systems vary widely around the world according to the species and environment it is cultured in. Below are some examples of common culture practices grouped by environment followed by a detailed description of a select few. For a more detailed review of culture practices see Baluyut (1989).

Finfish and shrimp culture practices

The breeding or rearing of fish in artificial or natural ponds or basins is the earliest form of aquaculture dating back to 1137 B.C. (Baluyut 1989). It is still used for many organisms such as shrimp and finfish in freshwater, brackish water and marine environments. Plastics used in this type of culture include nursery cages for smaller life history stages (Kungvankij et al. 1986), PVC pipes for water drainage and flow, pond liners (often made of high density polyethylene) and mesh screens to prevent undesirable organisms from getting into the ponds.

Fish cages and pens are generally comprised of a net stretched over a framework structure (Figure 5.2). Nets are often made of polyethylene and nylon monofilament twine although wire mesh is used in several countries (Baluyut 1989). Bamboo or other locally available wood is used for the framework structure. Several different cage flotation materials can be used including bamboo, PVC pipes/containers, steel or plastic drums, expanded polystyrene and aluminium floats (Baluyut 1989).

Figure 5.2 *Salmon farming, British Columbia, Canada taken by Bill Pennell*

Mollusc culture practices

Longline culture

Longline culture is used for mussels, oysters, scallops and other species worldwide. This can entail a variety of configurations consisting of longlines hanging from some type of float, raft or a line strung between floats. Some species (e.g. scallops and oysters) may be placed in polyethylene trays or lantern nets (Figure 5.3) hanging from floats. This is similar to the hanging method of oyster culture that uses oyster shells or similar materials as collectors. These are then strung on synthetic twine or heavy monofilament nylon attached to a rack/tray of bamboo or wood (Baluyut 1989). Natural and synthetic ropes are often used for spat collection (e.g. in the Philippines). Natural ropes attract more larvae than polyethylene or polypropylene ropes but don't last as long; therefore, a hybrid of the two is often used (Baluyut 1989). A wide variety of materials are used for scallop spat collection including polyethylene mesh bags, nylon and teased polypropylene rope (Lovatelli 1987).

a)

b)

Figure 5.3 *a) Oyster longline culture, British Columbia, Canada (taken by Bill Pennell); b) Longline oyster raft culture, British Columbia, taken by Centre for Shellfish Research, Vancouver Island University*

Stake method

In shallow waters (<1 m at low tide) with soft sediment the stake method is often used. This involves stakes, usually comprised of bamboo trunks or mangrove branches (Baluyut 1989) spaced apart to serve as attachment for spat (juvenile shellfish).

Hanging method – Oyster or mussels are collected on oyster shells or other collectors and attached to synthetic twine or heavy monofilament nylon. These are then spread out over stakes or a wooden or bamboo platform (Baluyut 1989).

Bottom culture – Shellfish can be cultured on the seafloor in deep or shallow water. Species cultured in this manner include clams, cockles, mussels and oysters. This may involve the use of fencing, often made from low-density polyethylene and/or anti-predator nets made from a variety of polymers including polyethylene and polypropylene.

5.3 Microplastic contamination and impact on fisheries and aquaculture products

Microplastics have been documented in both finfish and shellfish consumed by humans. However, the microplastic concentration in the edible tissues (i.e. flesh) is unknown for finfish. Plastic infrastructure is used widely in both fisheries and aquaculture hence there is concern about these sectors serving as a source microplastics that may contaminate seafood products. The limited studies to date indicate that farmed seafood could have higher microplastic concentrations than wild collected seafood. The ubiquitous nature of plastics and their potential to find their way into seafood products consumed by humans has led to concern about the potential threat of microplastics to seafood safety. Microplastic contamination and impact on commercial finfish, shellfish and other species is discussed below.

5.3.1 *Microplastics in commercial finfish*

Microplastic ingestion has been observed in a wide range of marine organisms (see Chapter 4), several of which are commercially important for both large- and small-scale fisheries (e.g. Anchovy, Indian Mackerel) (Annex Table AIII.2). How long the plastic stays in the stomach (e.g. residence time), and therefore the length of exposure to the microplastics and associated chemicals, is unknown. To date, studies are largely limited to examining microplastics in the gut and/or digestive tract, particularly for finfish, and transfer to other tissues is known only for a handful of invertebrate bspecies. This information is particularly important to fisheries and aquaculture because microplastics, and associated chemicals in or on them, may be transferred into the parts of the food fish that are consumed by humans.

In the following section we have compiled the available information on the ingestion of microplastics by commercially important marine species.

Ingestion Field studies have demonstrated the ingestion of microplastics in several commercial fish species, pelagic and benthic (bottom dwelling) fish, from the English channel (Lusher et al. 2013), the North Sea (Foekema et al. 2013), the Indian Ocean (Kripa at al. 2014), the eastern Pacific Ocean (Rochman et al. 2015a; Choy and Drazen 2013), the Indo-Pacific Ocean (Rochman et al. 2015a) and the north-eastern Atlantic (Neves et al. 2015). Information is available for non-commercial species globally (e.g. Boerger et al. 2010; Jantz et al. 2013), many of which are prey for larger fish. Research from the Mediterranean Sea (Avio et al. 2015), the Arabian Sea (Sulochanan et al. 2014) and the tropical Atlantic (Dantas et al. 2012) confirm the perception that fish are exposed to and ingest plastic particles globally (for an extensive list see Annex Table AIII.2).

Several different species of commercial fish, both pelagic and demersal, have been documented with microplastics in their guts. These include the pelagic bluefin tuna *(Thunnus thynnus),* swordfish *(Xiphias gladius),* albacore *(Thunnus alalunga),* Atlantic herring *Clupea harengus,* sardine *Sardina pilchardus,* European and Pacific anchovies *(Engraulis spp.),* Indian mackerel *(Rastrelliger kanagurta),* benthic/demersal hake *(Merlucius merlucius),* blue whiting *(Micromesistius poutassou),* red mullet *(Mullus barbatus),* small scale gurnard *(Chelidonichthys lucernus)* and common dolphin fish *(Coryphaena hippurus)* (Foekema et al. 2013; Kripa et al. 2014; Rochman et al. 2015a; Romeo et al. 2015; Lusher et al. 2013; Avio et al. 2015; Deudero and Alomar 2015). At present, 89 species of fish have been reported to ingest microplastics. Of those, 49 species are targeted commercially. Insufficient data from different spatial regions prevents geographical comparison.

Although not often commercially targeted, mesopelagic fish are an important component of the oceanic ecosystem (Gjøsaeter and Kawaguchi 1980). They have also recently been identified as potential future target species for fishmeal. Mesopelagic fish from the family Myctophidae have been reported with microplastic debris from both the Atlantic (Boerger et al. 2010) and the Pacific Oceans (Davison and Asch 2011). In the North Atlantic, 11% of individuals from 10 species of mesopelagic fish contained microplastics (Lusher et al. 2015 ICES JMS). Their high lipid content would benefit the growing demand from aquaculture for fish proteins and oil (FAO, 2010). With a global biomass estimated >1,000 million tonnes (Irigoien et al. 2014), this fisheries resource is still underutilized.

The number and size range of microplastics found in fish gut contents varies from 0 to 83 items per fish and between 0.1 mm to >5 mm (Annex Table AIII.2), the biggest ones being found in large predators. Information on fish (species, common name, numbers studied, % containing microplastics, mean number/range of particles ingested, type, size, location fish were caught) are presented in Annex Table AIII.2. Common plastic polymers found in fish are polyethylene, polypropylene, polystyrene, polyethylene terephthalate, polyvinylchloride and nylon. Their presence is related to the worldwide use of these plastics in many applications. The sources of microplastics found in commercial fish are unknown, although some of the plastic polymers reported in the English Channel and in the coast of Portugal are representative of those used in the fishing industry, which may allude to a possible source (Lusher et al. 2013; Neves et al. 2015).

Impact of ingestion Species of commercial fish do ingest microplastics, but at present we know very little about the impact to fish health. Microplastics may be egested along with faecal material, retained within the digestive tract, or translocate between tissues (this is more likely for nano-sized plastics). The retention and possible translocation of microplastics raises concern regarding whether the chemicals associated with microplastics may transfer into the tissues (i.e. the meat) of an organism. Microplastics accumulate contaminants from the environment and leach additives introduced during manufacturing (see Chapter 4). Because there is potential for chemicals associated with plastics to transfer to fish, research is needed to assess the impact of this interaction. In particular, studies should focus on contamination of the edible fractions that may pose a risk to human health. At present, we can only extrapolate results from laboratory feeding studies and observations in nature that focus on non-commercial fish species. These studies have looked at contaminant transfer and endpoints, such as accumulation in the tissues and altered predatory behaviour, and are described below.

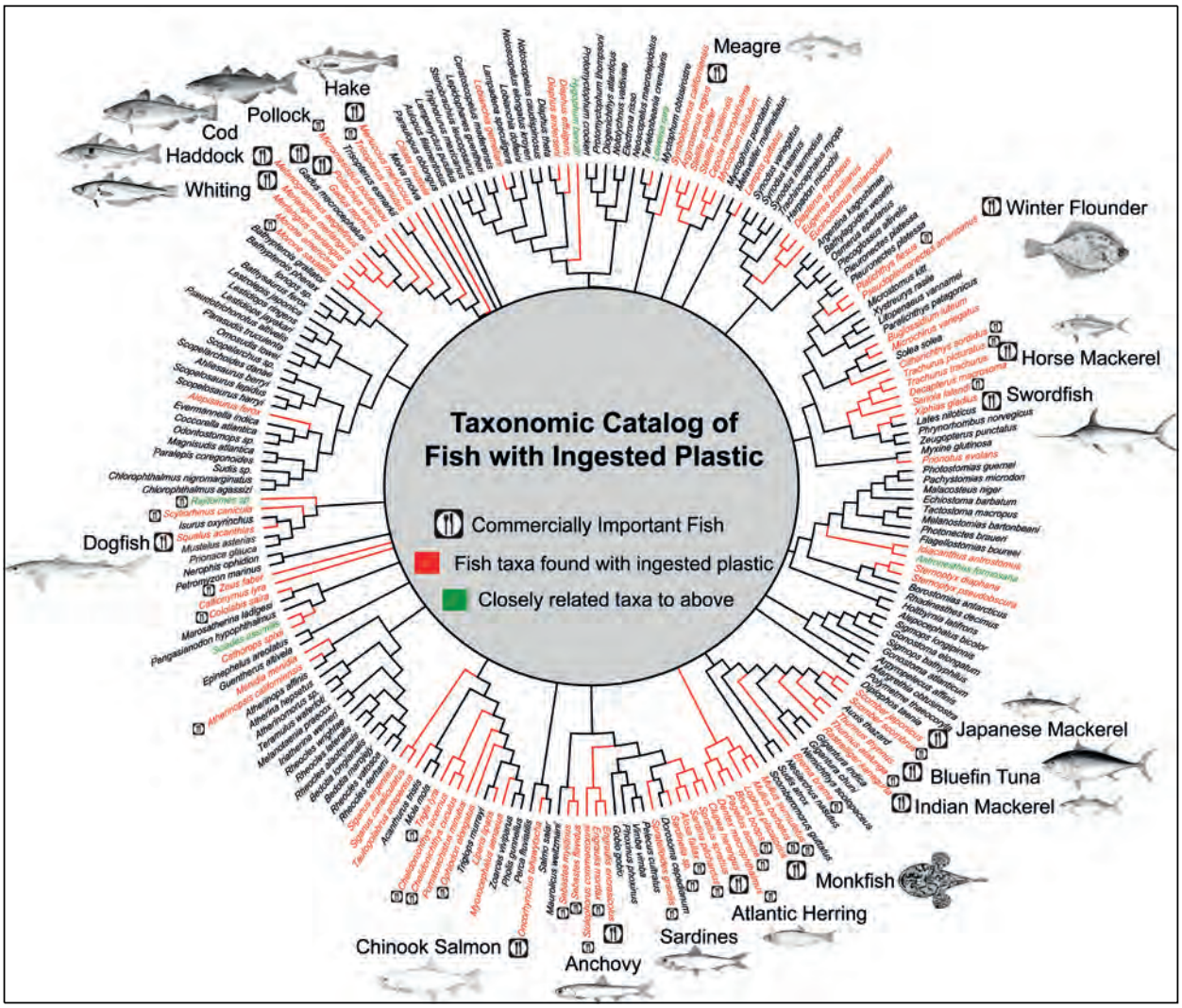


Figure 5.4 *A maximum likelihood inferred genealogy of fish based on barcoding gene cytochrome oxidase I (COI) depicting the fish species found to contain microplastics (in red) reported in the literature as of November 2015. Commercially important species are denoted with a food symbol while species in green are closely related to those genera found to contain microplastics but for which no COI gene was available in public databases. The following species are represented by closely related species in the tree: Astronesthes indopacificus, Diaphus phillipsi, Hygophum reinhardtii, Myctophum aurolaternatum and Sciades herzbergii. Species in black may also consume microplastics but have not yet been reported in the literature to do so. The tree was constructed using RAxML 7.2.8 as implemented in Geneious R8 (Biomatters Ltd, Auckland New Zealand) and included 216 species and 585 homologously aligned nucleotide positions. The COI gene is not recommended for establishing more distant relationships between species so branching order should be interpreted with discretion. See Table AIII.2 for additional relevant data*

Very few laboratory experiments have examined exposure to microplastics in commercial fish species. These include the recent work of Mazurais et al. (2014) who looked at sea bass larvae *(Dicentrarchus labrax),* Avio et al. (2015) with the grey mullet *(Mugil cephalus)* and an earlier study of gastric evacuation in the cod *(Gadus morhua)* by Santos and Jobling (1992). Concentrations of microplastics in these experiments are many times above what is commonly found in the environment. Particles have been used alone or in combination with metals or natural prey to assess different endpoints after variable exposure times (see Table AIII.1). Microplastic particles decreased growth rate of sea bass larvae, but no effects were detected in grey mullets *(Mugil cephalus),* though microplastics could be found in the stomach and liver.

Other studies have examined the impacts from microplastics in fish that are non-commercial but can be used as models for other species of fish, including commercial fish. In juveniles of the common goby *(Pomatoschistus microps),* environmentally realistic concentrations of microplastics caused a decrease in predatory performance due to confusion of microplastics with food and AChE inhibition (a neurotransmitter; Luís et al. 2015; de Sá et al. 2015). These two studies also showed evidence that developmental stage and environmental conditions experienced during development may influence the susceptibility of fish to ingest microplastics. Another study also observed toxic effects in the Japanese medaka *(Oryzias latipes)* from an exposure to microplastic with and without environmentally relevant concentrations of organic chemi-

cal contaminants. Effects including hepatic stress and changes in gene expression were observed (Rochman et al. 2013a, 2014a). Still, it should be noted that findings from controlled laboratory experiments should be interpreted with caution as results are difficult to extrapolate to the field, where multiple factors interact and co-vary.

A large number of wild-caught fish have been reported to ingest microplastics. As studies continue investigating additional species, it is likely that this number will rise. However, given the current level of knowledge and limited availability of data, we cannot interpret the effects of microplastics on commercial fish species. Future research should be directed towards commercially targeted species and those fish that constitute their prey. It will also be important to focus laboratory research on retention times of microplastics to evaluate exposure time and bioaccumulation of associated contaminants (e.g. PCBs, PBDEs) in the tissues; and to be able to relate the observed effects to microplastic concentrations. Furthermore, studies to date have documented microplastics in fish guts rather than the tissues that are typically eaten by humans. There is an urgent need to evaluate the presence of microplastics and associated contaminants in the edible fractions of the fish and other products for human consumption (e.g. fish oil) so that the potential hazard of microplastics to the consumer can be assessed. This information is crucial to predict the risk to fish populations, that may compromise commercial fisheries, and also to human health.

5.3.2 *Microplastics in shellfish and other species*

Microplastics have been observed in many commercial species other than fish, including mussels, clams, oysters and scallops. Research has examined laboratory exposure of microplastics in many of these animals as well as contamination in wild and cultured animals, including some that were store-bought in Europe, North America and Asia (De Witte et al. 2014; Li et al. 2015; Van Cauwenberge and Jansen 2014; Rochman et al. 2015a; Vandermerrsch et al. 2015). The possibility of transfer to human consumers is elevated because shellfish are eaten whole. Below we highlight examples of microplastic ingestion and the potential physical and chemical consequences to commercially important species.

Microplastics identified in shellfish range in size from 5 µm to 5 mm and are composed of fragments, pellets and fibres. Fibres seem to be one of the most common types of microplastics found in invertebrate food fish. In 8 of 9 species of shellfish sampled from an Asian fish market, fibres constituted more than 52% of plastic items per species, with the exception of *Alectryonella plicata* where pellets were most abundant at 60% (Li et al. 2015). In a European study synthetic fibres were also the dominant microplastics and ranged from 200µm up to 1500µm size (De Witte et al. 2014).

Occurrence in mussels and other bivalves

In nature, both wild and cultured mussels *(Mytilus edulis)* have been found to ingest microplastics. Some studies collected animals from the field and others from aquaculture farms or purchased directly from the market. The concentrations of microplastics found in *M. edulis* studies ranged from 0 to 34 particles per g (wet weight) (Li et al. 2015; Vandermeersch et al. 2015; De Witte et al. 2014; Van Cauwenberghe and Jansen 2014; Van Cauwenberghe et al. 2015). Only one study has directly compared microplastic concentrations in farmed and wild *M. edulis* in Nova Scotia, Canada (Mathalon and Hill 2014). Microplastic concentrations were higher in farmed mussels (average 178 fibres per farmed mussel compared to 126 microfibres per wild mussel). The authors suggest this difference may be due to contamination from the plastic rope longlines the mussels are cultured on (Box 5.4).

Although many observations have been made with *Mytilus edulis,* other species of shellfish have also been found to be contaminated by microplastics. The brown mussel, *Perna perna,* is another mussel with commercial value on tropical coasts that is susceptible to microplastic contamination. Microplastics were observed in 75% of brown mussels from the Santos estuary, a highly urbanized area on the Southeast coast of Brazil (São Paulo state; Santana et al. submitted).

Microplastics have also been observed in wild and cultured Manila clams *(Ruditapes philippinarum)* (S. Dudas personal communication), oysters *(Crassostrea gigas, Alectryonella plicatula)* (Van Cauwenberghe and Jansen 2014, Li et al. 2015; Rochman et al., 2015a) and several species sold in a Chinese fishery market such as Scapharca clams, ark clams *(Tegillarca granosa),* razor clams *(Sinonovacula constricta),* scallops *(Placopecten yessoensis),* Mediterranean mussels *(Mytilus galloprovincialis)* and venus clam species *(Meretrix lusoria, Cyclina sinensis* and *Ruditapes philippinarum)* (Li et al. 2015).

Similar to finfish, there is little information regarding the effects of microplastics on shellfish. The effect of microplastic ingestion on feeding modes and gut passage time have only been observed in *Mytilus edulis* and *Placopecten magellanicus* (Brillant and MacDonald 2000, 2002; Ward et al. 2003; Ward and Kach 2009).

As described in Chapter 4, microplastic particles can have physical and/or chemical consequences to an animal upon exposure. There are many studies that have examined the impacts of microplastics in mussels. One study showed that microplastics (2 to 16 µm) can be retained by *Mytilus edulis* following ingestion (Browne et al. 2008) and that the particles in the size range 3 to 9.6 µm can be translocated outside the gut and into the hemolymph. Other studies also observed the transfer of microplastic to the circulatory system and some with consequential toxicity, including reduction in function of the reproductive system and inflammation (formation of granulocytomas) (von Moos et al. 2012; Avio et al. 2015). In contrast, another study found that *Mytilus edulis* reject nano-sized particles of plastics as pseudofaeces before ingesting them (Wegner et al. 2012). Still, there can be an energetic cost associated with pseudofaeces production, thus long-term exposure to microplastics may negatively impact individuals. In oysters, exposure of the Pacific oyster *(Crassostrea gigas)* to microplastics indicated effects on reproduction (Sussarellu et al. 2016).

In a laboratory study, direct bioaccumulation of associated chemicals from microplastics was also demonstrated in clean mussels *Mytilus galloprovincialis.* Mussels that ingested and assimilated polyethylene and polystyrene particles contaminated with polycyclic aromatic hydrocarbons bioaccumulated the chemical in their tissues (Avio et al. 2015).

Box 5.3 Can shellfish depuration reduce microplastic contamination?

When shellfish are grown in waters contaminated by domestic and industrial wastes they must be depurated to ensure satisfactory microbiological and chemical quality of the product for consumption (Baluyut 1989). This is because bivalves filter their food from the water. Along with phytoplankton and microbes, they filter, and can concentrate, contaminants (including chemicals and microplastics) present in the water column. Depuration is a kind of purification system where shellfish are held in clean seawater in conditions that facilitate maximum filtration activity (i.e. to expel the intestinal contents) and that enhance separation of the expelled contents to avoid recontamination (Lovatelli et al. 2008). To date, only one study has examined the potential for reducing microplastic contamination in shellfish through depuration. This study showed that without any depuration, farmed mussels from Germany contained on average 0.36 $\pm$ 0.07 particles/g wet weight. After three days of depuration, this average was reduced to 0.24 $\pm$ 0.07 particles/g wet weight (Van Cauwenberghe and Jansen 2014). In another species, *C. gigas,* microplastic concentration decreased after depuration from 0.47 $\pm$ 0.16 particles to 0.35 $\pm$ 0.05 particles/g wet weight (Van Cauwenberghe and Jansen 2014). More research is urgently needed to investigate the utility of longer depuration times and depuration using running water to reduce microplastic load in shellfish.

Crustaceans

Commercially important crustaceans also ingest microplastics. Green crabs *(Carcinus maenas)* were found to ingest microplastics under controlled conditions (Farrell and Nelson, 2013; Watts et al. 2014). This ingestion was observed through contaminated food consumption (mussels artificially contaminated with microplastics), thereby suggesting the possibility of trophic transfer. Farrell and Nelson (2013) identified the assimilation and persistence of microplastics within the crabs over 21 days. Microplastics were found in the stomach, hepatopancreas, ovary and gills (Farrell and Nelson, 2013). Watts et al. (2014) noted that ventilation through the gills was another route of uptake in crabs.

Lobsters, *Nephrops norvegicus,* sampled from the Clyde Sea (Scottish coast), also had microplastics in their stomachs. About 83% of the individuals examined had ingested plastics that ranged in volume and size, that were mainly composed of monofilaments (Murray and Cowie, 2011).

Natural populations of brown shrimp *(Crangon crangon),* sampled across the English Channel area and Southern part of the North Sea (between France, Belgium, the Netherlands and the UK) were found to be contaminated with microplastics as well (Devriese et al. 2015). Shrimp from different locations did not have significantly different plastic content (Devriese et al. 2015). In total, 63% of the animals examined were contaminated with microplastics, which were mostly composed of synthetic fibres (96.5%, ranging from 200µm up to 1000µm size) (Devriese et al. 2015). *C. crangon* had, on average, 1.03 fibres/g wet weight (Devriese et al. 2015). The amount of microplastic ingested by *C. crangon* varied temporally, possibly due to seasonal fluctuations on the occurrence of plastic (Devriese et al. 2015). The authors also investigated the relationship between the condition of the shrimp and the level of contamination of microplastics within an individual. No relationship was found, indicating that microplastic contamination does not affect the health of the shrimp (Devriese et al. 2015).

Gastropods

Two studies reported on the presence or absence of microplastics in edible snails collected from the Dutch coast: 30 microplastics per gram d.w. in periwinkles *(Littorina littorea)* (Leslie et al. 2013) while microplastic could not be detected in common limpet *(Patella vulgaris)* (Karlsson 2015).

Echinoderms

Sea urchin larvae, *Tripneustes gratilla,* exposed under laboratory conditions to microplastics in various concentrations (1 to 300 particles/ml, with an exposure duration of 1 to 9 days) ingested and egested microplastic particles (Kaposi et al. 2014). The impact of ingestion was not investigated. Earlier research on sea cucumbers found that *Holothuria* sp. selectively ingested plastic particles in preference to food items (Graham and Thompson 2009). The commercial market targets the body of the organism and removes their gut. If microplastics are translocating from the gut to the tissue of the organisms there could be concerns relating to bioaccumulation in the food chain. However, the data available for echinoderms suggest that microplastics are removed along with faecal material.

Microplastics have been observed in several types of seafood cultured and caught for human consumption (Rochman et al. 2015a; Van Cauwenberghe and Janssen 2014). Consequently there is increasing concern for human health and food safety (EFSA 2016). Given the potential for microplastic pollution in edible tissues of commercial fish or in the by-products (e.g. fishmeal and fish oil) there is an urgent need for more research in this area.

Box 5.4 Microplastic contamination in wild versus cultured seafood

There are few studies to date that make direct comparisons between microplastic contamination levels in wild and cultured organisms and they are limited to shellfish. Preliminary studies found that farmed mussels from Nova Scotia had significantly higher microplastic concentrations than wild mussels (Mathalon and Hill 2014). A preliminary study on Manila clams conducted on the west coast of Canada showed higher microplastic concentrations in farmed clams (~12 microplastic particles/farmed clam versus ~9 particles/wild clam (Davidson and Dudas submitted) but these differences were not significant. Differences in methods and the biology of clams versus mussels may explain the different findings. In Mathalon and Hill's study (2014), cultured mussels were purchased from a grocery store rather than being obtained directly from the farms, which introduces the potential for contamination because bivalves often gape when frozen. Additionally, mussel farming methods differ from those used to grow clams. Most mussels on the eastern coast of Canada are grown on long-line (DFO 2015). The fraying of plastic-based ropes in close contact with growing mussels may influence the amount of microplastics ingested compared to other methods with fewer plastic structures (e.g. bottom or rack culture). Farmed mussels *(Mytilus edulis)* and oysters *(Crassostrea gigas)* from Germany were also found to harbour microplastics (Van Cauwenberghe and Janssen, 2014). Farmed mussels from the North Sea (Germany) had an average of 0.36 $\pm$ 0.07 particles/g, which is much lower than the concentration observed in farmed Manila clams above (1.7 $\pm$ 1.2 particles/g). This may be due, in part, to the lack of plastics used for mussel culture in this study. Culture methods, ocean currents, extent of shellfish farming, and coastal development may all affect microplastic contamination. Finally, clams and mussels have very different filtration rates (Cusson et al. 2005; Hadley and Whetstone, 2007) and have variable longevity depending on the age they are harvested for market sales, both of which will influence microplastic particle concentration. All of these factors are important considerations for assessing the risk of microplastics for organismal and human exposure.

5.4 Impacts on food security

Microplastics are found in a variety of species consumed by humans and thus there is concern about their potential to negatively affect food safety and potentially, food security. Impacts will be dependent upon consumption rates and patterns (e.g. species and anatomy consumed). Data on microplastic contamination of seafood products, particularly edible tissues, is very limited thus the risk of microplastic consumption on human health is unknown. The section below describes the potential risk of microplastics on food security and the implications for human health.

5.4.1 *Food safety and security*

Anthropogenic debris has become widespread in the marine environment globally. As such, there is concern about whether the ingestion of anthropogenic debris, such as microplastics by marine animals, can cascade up the food web to influence fish stocks and/or human health. It is clear from scientific studies that microplastics have infiltrated marine food webs to the level of humans via seafood (Rochman et al. 2015a; Li et al. 2015; Van Cauwenberghe and Jansen 2014). The physical harm that anthropogenic debris causes to marine animals at several levels of biological organization (Rochman et al. 2015b) can potentially threaten local food availability in locations where debris is abundant and seafood is a major source of protein to the local population (e.g. Indonesian island communities). Moreover, anthropogenic debris is associated with a cocktail of hazardous chemicals (see Chapter 4). Consequently, there is concern that chemicals from plastic may be transferring to humans via diets containing fish and shellfish, raising important questions regarding consequences for human health. The implications of microplastics for food safety, security and human health are discussed below.

Food safety is a term used to describe several facets (e.g. chemical, microphysical and microbiological) of food handling, preparation and storage to prevent illness and injury (Hanning et al. 2012). Microplastics may affect food safety as a contaminant or via the chemical contaminants on them (see Chapter 4) that could be transferred into food. Food safety and security are interrelated as shown in Figure 5.8 (Hanning et al. 2012). While food safety ensures that the food is safe from chemical, physical or biological standpoints, food security ensures there is enough access to, and enough food for people to lead productive lives (Hanning et al. 2012).

The World Food Summit (1996) states that: 'food security exists when all people, at all times, have physical and economic access to sufficient safe and nutritious food that meets their dietary needs and food preferences for an active and healthy life'. This encompasses the following four dimensions (FAO 2008):

i. Availability – food must be available based on food production, stock levels and trade;

ii. Access – food must be physically (e.g. food supply) and economically accessible (e.g. affordability);

iii. Utilization – the way the body uses nutrients combined with feeding practices, food preparation, diet diversity and household distribution of food will determine nutritional status of individuals and,

iv. Stability – the above three dimensions must be stable over time to ensure food security.

Microplastics have the potential to affect the availability, use and stability dimensions of food security and within them elements of food safety as well.

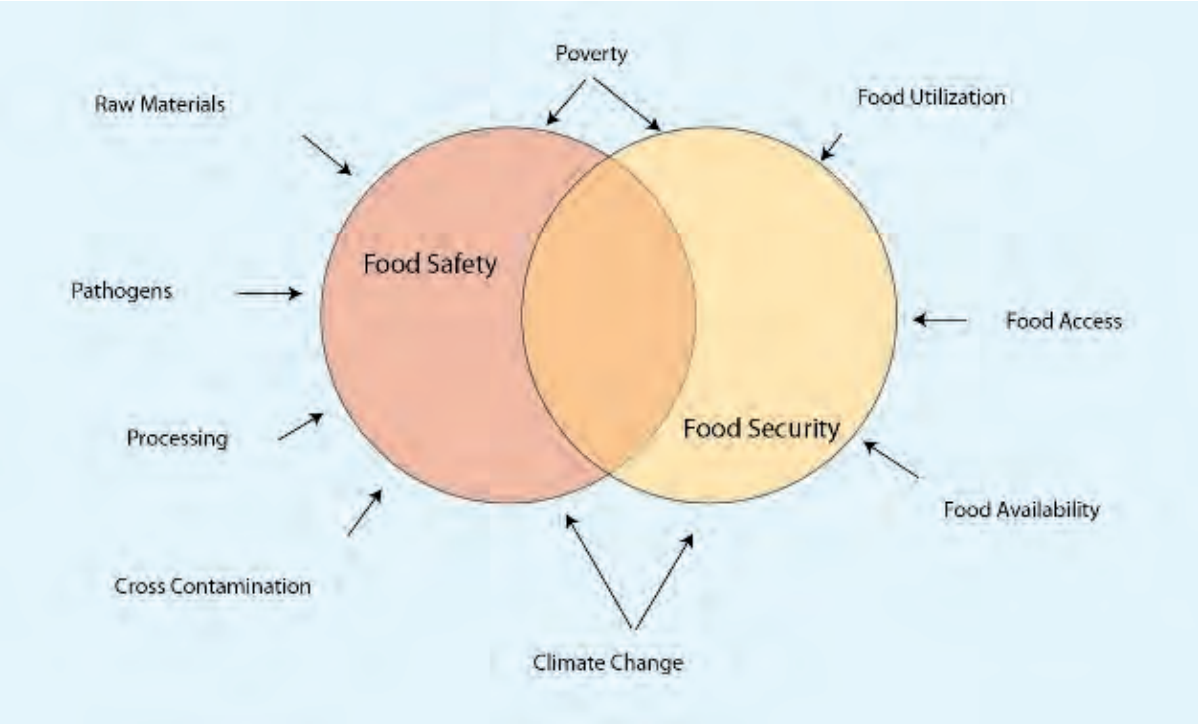


Figure 5.5 *Relationship between elements of food safety and food security (Hanning et al. 2012)*

Availability

To date, there have been no demonstrated impacts of how plastic debris impacts fish stocks. This is in part because researchers have not investigated this and because it is challenging to address. Typical concerns with marine debris and fisheries and aquaculture are focused on the negative impacts of derelict fishing gear such as nets and traps (Kühn et al. 2015). Several studies have shown that many individual organisms are killed by derelict gear (Uhrin and Shellinger 2011; Good et al. 2010), but these studies have not tested whether this leads to population-level declines (Rochman et al. 2015b). Larger marine debris has also been shown to impact subsistence fishers' behaviour (i.e. avoiding preferred fishing grounds) when it is abundant enough to pose a significant entanglement hazard (Nash 1992). These changes in behaviour may affect them economically (see Section 6.3).

Box 5.5 Toxicity of plastics used in fisheries and aquaculture

Plastics used in aquaculture and fishing operations include polyvinyl chloride (PVC), polypropylene, polyurethane foam, polystyrene and nylon. Although microplastics aren't used directly in aquaculture the large plastics used (e.g. cages, rope) may provide a secondary source of microplastics through degradation. Based upon the chemical composition of the plastic material or the sorption properties, some plastic types may be innately less hazardous than others. For example, PVC, polystyrene and polyurethane foam all contain monomers and/or additive ingredients which are known to be hazardous, whereas the monomers and ingredients of polyethylene and polypropylene are benign (Lithner et al. 2011). Moreover, Rochman et al. (2013 ES&T) and Lee et al. (2014) have demonstrated that polyethylene, polypropylene, polyurethane and polystyrene consistently accumulate greater concentrations of persistent organic pollutants than PVC and PET (Rochman et al. 2013; Lee et al. 2014). Sorptive behaviour and chemical ingredients must be considered together. Although PVC sorbs relatively small concentrations of hazardous organic chemicals, its vinyl chloride monomer is classified as carcinogenic and toxic (Lithner et al. 2011). In addition, PVC contains more hazardous additives than polyethylene and polypropylene (Lithner et al. 2011). Polyethylene terephthalate has been suggested as one of the least hazardous plastics (Lithner et al. 2011; Rochman et al. 2013 ES&T) because it sorbs smaller concentrations of chemicals, requires fewer additives and degrades faster than other polymers.

Based upon information regarding impacts from larger plastic debris, people perceive there will be impacts from microplastic, but at present, this has not been tested and/or demonstrated. Microplastics can harbour pathogens that could negatively impact fisheries, as diseases are a major source of loss in aquaculture of molluscs, crustaceans and fish (Zettler et al. 2013; Lafferty et al. 2015). Snoussi et al. (2009) showed that potentially pathogenic strains of *Vibrio* adhere to and persist on plastic surfaces associated with marine fish aquaculture, and more recently bacteria in the genus *Vibrio* have been found on microplastics drifting in the North Atlantic Subtropical Gyre (Zettler et al. 2013). Certain strains of *Vibrio* (e.g. *Vibrio parahaemolyticus)* can cause illness in humans, shellfish, finfish and crustaceans. Shellfish closures due to this illness can be devastating to the industry and have large economic impacts if the closures are prolonged. Another impact may arise from the priority pollutants that are associated with plastic debris which are known to cause

toxic effects at certain concentrations (Rochman et al. 2013b; see Chapter 4).

It is also important to consider the impact of consumer perception on food availability. As our knowledge of microplastics and their presence in fish and other foods increases so does consumer concern. If consumers feel that fish are unsafe to eat then it becomes 'unavailable' to them whether this perception is accurate or not. It may also reduce the value of seafood products that are thought to be contaminated. Because of this, it is important that we do not overstate the impacts of microplastics in marine organisms before we fully understand them. See Section 6.3.2 for further discussions about the potential impacts on consumer perception.

Use

During food preparation and cooking, microplastics in seafood will be subject to heat that can influence the leaching of chemical ingredients and sorbed contaminants from the debris. As such, it is important to consider how food use and preparation may impact the toxicity of the microplastics in seafood. Heating plastic can influence the kinetics of the chemicals present and/or cause them to transform into different forms (e.g. dioxin). For example, plastics can leach bisphenol A, styrenes and phthalates, all of which can have implications for human health (Halden 2010).

Stability

Marine debris and microplastics are a persistent pollutant, but there can be sudden increases from events such as seasonal rains, storms or tsunamis that can cause large amounts of debris to enter the ocean quickly (see Section 2.4.2). Plastic degradation can also be accelerated by environmental conditions that promote wear and decomposition (e.g. storms, higher temperatures). It is possible that microplastics could increase locally or regionally over a shorter period of time due to these stresses and affect the stability of seafood resources.

5.4.2 *Global consumption patterns*

Consumption volumes and species

Japan and the USA have the highest import value of seafood (for individual countries) followed by China with half of their value (FAO 2014). The European Union is the largest trader of fishery and aquaculture products in the world with a value of $47 billion in 2012, representing 36% of total world imports (FAO 2014). In terms of the contribution of fish to animal protein supply, Greenland, Japan, the Philippines and Portugal are among some of the top consumers which consume >10 g per capita/day representing more than 20% of the contribution of fish to animal protein supply (FAO 2014). At a global level, the consumption of fish is around 20 kg/capita/year. This is equivalent to an average intake of 10g of fish protein/capita/day. This number is much higher for high fish consumers. In Asia a relatively high per capita consumption rate is combined with large populations making it the most important fish-consuming region, followed by Europe (FAO 1998). Although average per capita fish consumption is usually lower in developing countries, fish may be the staple food in coastal areas and is an important protein source for the poor (FAO 2014).

Detailed information on consumption of different types of seafood (e.g. finfish, molluscs etc.) is very limited. However, in the recent SOFIA report (2014), fish consumption in the Asia-Pacific Region was assessed using household surveys and some highlights are presented below (Box 5.6). Although species-specific consumption patterns are difficult to ascertain it is crucial information for assessing the risk of microplastics and how they relate to seafood and human consumption.

What parts of the fish are consumed?

Depending on the region, culture, size of fish, and food preparation, different parts of fish and shellfish may be consumed. Shellfish are generally consumed whole with the exception of certain species such as scallops from which the muscle and gonads are consumed. Most countries consume finfish flesh while consumption of fish heads, viscera and other body parts are less common. Solid wastes or by-products generated by fisheries vary by species but can represent a significant portion of the original material (e.g. 65% for the tuna canning industry). Direct human consumption of fish by-products has been increasing in recent years and alternative uses for these by-products are being found (FAO 2014). For example, fish viscera and frames are used as a potential source of protein hydrolysate for its potential as a source of bioactive peptides (FAO 2014). In the salmon industry in Norway, of the 45,800 tonnes of heads, frames, belly flaps and trimmings, 24% (11,000 tonnes) were used for human consumption and the rest for feed (Olafsen 2011 cited from FAO 2014). Often after gutting or filleting salmon, the heads, frames and trimming are purchased for use in soups or other dishes (FAO 2014). These considerations are very important as they will affect the level of microplastic exposure in humans.

Unfortunately current studies only document microplastics in the gut and intestinal tract of fish, highlighting the need for information on contamination of other tissues. This is particularly true for regions with high consumption rates of seafood, such as the Pacific Islands, Cambodia and the Philippines. These countries consume some species of fish that have been found to ingest microplastics. While studies showing translocation to edible tissues are limited, it is possible that microplastic can transfer to the meat, particularly at the nanoscale.

Box 5.6 Fish consumption in the Asia-Pacific region

Bangladesh	Fish and fish products account for 11.1% of total protein consumption, 76% of which is inland. The most commonly consumed marine species is Hilsa shad.
Cambodia	The consumption rate of 63.15 kg/capita/year of fish and fish products appears to be among the largest in the Asia-Pacific region. This represents 37% of the protein consumed of which 71% is from inland fisheries (FAO 2014).
Indonesia	Fish and fish products are consumed at a rate of 12.8 kg/capita/year or 16.4% of the total protein consumed. Of this more than 70% of fish consumed is marine fish and skipjack tuna is the most commonly consumed followed by anchovy and Indian mackerel (FAO 2014).
Myanmar	Consumption of fish and fish products is 21.02 kg/capita/year, representing 22.6% of the total protein consumed. Marine species comprise 23.5% of fish consumed with fish paste being the most common product and hilsa shad the most common marine species eaten (FAO 2014).
Pacific Islands	These islands have the highest annual consumption rates at 110.7 and 87.4 kg/capita/year for Tuvalu and Samoa respectively.
Philippines	Annual fish consumption is 40.15 kg/capita with canned fish and sardines, mackerel scad and milkfish being the most commonly consumed produces and species (FAO 2014).
Sri Lanka	Average annual consumption is 15.3 kg/capita with marine species, most commonly sprat followed by skipjack tuna and goldstripe sardinella, comprising 81% of the fish consumed (FAO 2014).
Thailand	Fish and fish products are consumed at a rate of 31.4 kg/capita/year representing 11.7% of total protein consumption. Marine fish represent 47% of the fish consumed.

5.4.3 *Human health implications*

The impacts of micro- and specifically very small microplastics (i.e. nanoplastics – particles <1000 nm in at least one of its dimensions) on human health are not well documented (Eerkes-Medrano et al. 2015) and our knowledge about the fate and toxicity of plastic particles for humans is unknown (Van Cauwenberghe and Janssen 2014; Bouwmeester et al. 2015; GESAMP 2015). In relation to food safety, the possible impacts of microplastic on human health will rely on dietary exposure via contaminated marine foodstuffs. In general three possible effects of plastic particles can be recognized: 1) particle toxicity caused by the very small (nano-size and lower micro-size range) plastic particles themselves due to interaction with external tissues and cells or after translocation into tissues and cells; 2) chemical toxicity due to the leaching of additives added to the microplastics during manufacturing or the release of pollutants that have accumulated onto the plastics in nature and 3) disease risks due to microbial contamination of microplastics. In theory, cumulative effects can occur through particle and chemical toxicity after the particles have been internalized in tissues or chemical mixture toxicity effects (see Chapter 4).

Box 5.7 Contaminants in seafood

Seafood can become contaminated through environmental exposure or during production. Several contaminants are monitored in seafood to ensure that levels are within acceptable limits for consumption. Many of these contaminants have been documented on plastic either as additives to the plastics or contaminants that have adsorbed from the environment (see Chapter 4). Microplastics have the potential to introduce chemical contaminants into organisms destined for human consumption or to remove them (see Chapter 4). The potential for microplastic to increase the concentration of harmful chemicals in seafood is a concern for food safety. The following are some of the most common contaminants in seafood and their documentation on microplastics (in bold) (Seafish 2015): **lead, cadmium,** mercury, **dioxins and PCBs, PAHs, brominated flame retardants,** marine biotoxins, histamine, radionuclides, melamine and structural analogues.

It is evident that humans are exposed to micro and nanoplastics through the consumption of marine food stuffs, including shellfish, fish and sea salt. In addition to seafood, humans may be exposed to microplastics via other routes, including drinking water, bathing waters, inhalation from air and/or via active contact with cosmetics (Napper et al. 2015). Microplastics have been detected in a variety of terrestrial foodstuffs such as honey, drinking water, beer, sugar and table salt (Liebezeit and Liebezeit 2013, 2015; Yang et al. 2015). An analysis and assessment of the potential health risk of microplastics for humans should comprise dietary exposure from a range of foods across the total diet in order to assess the contributing risk of contaminated marine food items (GESAMP 2015).

An overview of microplastic concentrations in marine, freshwater and terrestrial foodstuffs is given in Table 5.1. Among the various types of seafood, consumption of filter feeding invertebrates, such as mussels or oysters, appears the most likely route of human exposure to microplastics. One study has attempted to estimate potential dietary exposure based on observed microplastic concentrations in seafood and approximated consumption rates. This study estimated dietary exposure for mussel consumers to range between

about 11,000 (Van Cauwenberghe et al. 2014) and 100,000 microplastic particles per person/year (see GESAMP 2015). Dietary exposure for shrimp consumers (90% removed by peeling) may amount to much lower exposure levels of 175 microplastics per year (Devriese et al. 2015).

The commonly used analytical techniques introduce a great bias in the knowledge, since they are only able to detect plastic particles well above the nano-range (see Chapter 7; Bouwmeester et al. 2015; GESAMP 2015). It is plausible that the smaller particles pose a greater risk than the larger particles (>1 micrometre) due to their smaller size, higher surface to volume ratio and associated increased chemical reactivity of the nano-sized group. For example, sorption of polychlorinated biphenyls (PCBs) to nano-polystyrene was shown to be 1 to 2 orders of magnitude stronger than to micro-polyethylene (Velzeboer et al. 2014; see also Chapter 4.4.2). Particles at the smaller end of the size spectrum (nano scales) have also been shown to cross cell in controlled laboratory experiments. Experimental evidence with rodents shows that microplastics >1 micrometre may reach the blood circulation via lymph, but cannot penetrate deeply into organs (Bouwmeester et al. 2015; GESAMP et al. 2015). They might cause local effects on the gut epithelium, the immune system, inflammation, encapsulation (fibrosis) and cell damage (Bouwmeester et al. 2015; GESAMP et al. 2015; see also Chapter 4.4.3). Unlike microplastics, nanoplastics may reach and penetrate all organs, including the placenta and brain (Bouwmeester et al. 2015; see also GESAMP, 2015). It is important to investigate potential impacts through food consumption (Hollman et al. 2013) as accumulation of ingested microplastics through the food chain and the consumption of seafood has not yet been demonstrated as harmful.

Chemicals (i.e. additives and monomers) inherent in microplastics or chemicals sorbed and transported by microplastics (Chapter 4) may contribute to human health impacts. The toxicity of some of their components to humans, especially plasticizers and additives (Flint et al. 2012; Oehlmann et al. 2009) and the possible leaching of poisonous chemicals, may be considered as a potential human health hazard. However, on the basis of the available evidence, which is predominantly based on larger sized microplastics, it appears that adhering persistent organic pollutants (POPs) and leachable additives of ingested microplastics will have a minor impact on contaminant exposure to fish (Bouwmeester et al. 2015). Due to the absence of knowledge on nanoplastic exposure to humans, their potential chemical risk, especially after translocation into tissues and cells remains unknown.

Table 5.1 Examples of microplastic concentration in foodstuffs

Species	#/kg wet weight or litre of product	Reference
Blue mussel (North Sea)	260–13200	Van Cauwenberghe et al. 2014 De Witte et al. 2014 Leslie et al. 2013
Brown shrimp (North Sea)	680	Devriese et al. 2015
Honey (various branches)	0.09–0.29	Liebezeit 2013, 2015 Leslie et al. 2015
Beer (Germany)	2–79 fibres 12–109 fragments 2–66 granules	Liebezeit and Liebezeit 2013, 2015
Table salts (China):		Yang et al. 2015
Sea salts	550–681	
Lake salts	43–364	
Rock/well salts	7–204	

***Note** different methods have been used in each of these studies, which will affect the ability to detect microplastics

Fishing effort (Watson et al. 2013), aquaculture production (FAO 2014) and microplastic distribution (this study) all exhibit significant regional variations in intensity, suggesting that there may be specific geographical regions where the likelihood of microplastics posing a risk to seafood is greater. For example, Asia has both high fish catches and aquaculture production (FAO 2014) and higher estimated microplastic abundance (GESAMP 2015). Temperature will also play a role as there is a correlation between certain pathogens and temperature (e.g. *Vibrio)*. A rigorous risk assessment that accounts for all of the pathways and factors that influence exposure, impact and their variation geographically would be instrumental in identifying hot-spots where any negative impacts are most likely to surface first.

5.5 Conclusions, knowledge gaps and research priorities

5.5.1 *Conclusions*

The study of microplastics is in its infancy; however, information on microplastic distribution, concentration and impacts is increasing rapidly. Plastics used in fisheries and aquaculture sectors can degrade into microplastics that can then contaminate seafood products. Our current level of knowledge indicates there is potential for both ecological and economic impacts that could extend to fisheries and aquaculture sectors. The impacts of the consumption of microplastics by food fish are unknown; however, studies on non-commercial species suggest microplastics have the potential to

negatively affect organism health. Consequently, there is concern that microplastics may affect food security and food safety. In order to properly assess the risk of microplastics to organismal and human health further research is needed as outlined below. Several recommendations are described in Chapter 10 that focus on proactive risk reduction initiatives relevant to fisheries and aquaculture.

5.5.2 *Knowledge gaps*

The ingestion of microplastics and associated impacts have been documented for a wide range of marine species; however, very little is known about the fate of microplastics ingested by commercial species and seafood products (e.g. fishmeal). Further, while the ingestion of microplastic by fish has been documented extensively we don't know how long and if the plastic is retained in the gut and/or if it is translocated to other tissues that may be consumed by humans. These are the most pressing knowledge gaps that must be addressed in order to determine how and if microplastics pose a risk for food safety and potentially food security. Little evidence is available on the cumulative impacts of microplastics with other stressors, such as increased water temperature, and the degree to which chemicals associated with plastics add to the overall body burden of marine species.

5.5.3 *Research priorities*

- Assess level of microplastic contamination in commercial species, seafood products (e.g. fishmeal and fish oil) and in fish prey (e.g. *zooplankton);*
- Determine if there is transfer of microplastics and associated contaminants from one trophic level to the next;
- Assess chemical contaminant transfer from microplastics to seafood;
- Assess microbial pathogen load on MP in different areas of ocean (open ocean, areas impacted by human sewage, aquaculture and fisheries areas);
- Determine if seafood microplastic concentration is higher in cultured versus wild organisms;
- Determine if microplastic in seafood is an objective and perceived risk for human health in regards to food security and safety;
- Determine how microplastics affect different life stages (e.g. are earlier life history stages more sensitive);
- Determine if microplastics impact the quality and palatability of food;
- Conduct a risk assessment to assess hazards of microplastic in fish and shellfish to human health and the ecosystem; and
- Increase awareness and investigate public perceptions about microplastic in seafood.

6 SOCIO-ECONOMIC ASPECTS

Key points

The socio-economic impacts of marine plastic are growing with the ongoing increase in plastic in the marine environment. There is mounting concern, globally and by sector, about the increasing cost both of inaction and action needed across the value chain.

Economics

1. Whilst the benefits of action against macroplastics often outweigh their costs, downstream clean-up actions focusing on microplastics are unlikely to be cost-effective, underlining the need for upstream preventative measures on sources.
2. Many sectors of the economy are sources of microplastics either directly (releases of primary microplastics) or indirectly (macroplastics breaking down to microplastics).
3. It is in the interests of those employed in many sectors of the economy to find strategies to reduce marine litter, as this can help reduce social and economic burdens. Examples include: tourism and recreation, aquaculture and fisheries, and shipping.

Social aspects

1. Citizen consumption of goods and services, personal habits (e.g. use of reusable bags and packaging) and waste practices (littering, waste separation) are key drivers of marine litter.
2. Mitigating marine litter can benefit communities, support long term livelihoods (e.g. links to fisheries or tourism), well-being (e.g. linked to recreation) and social cohesion (e.g. sense of belonging to a clean environment).
3. Human health impacts can be mitigated by removing waste that can harbour pathogens or accumulate pools of water that host insects which are vectors for diseases, like denge fever.
4. A range of factors influence perceptions and behaviour, such as: cultural norms, gender, social standing, education level and economic status. Accounting for these in the design and implementation of measures to encourage behaviour change may result in longer lasting, more effective and lower-cost solutions.

6.1 Lessons from the first assessment

In regards to socio-economics, the first assessment reflected heavily on new research on macro debris and the general risk perception literature and applied these insights to microplastics. Specifically it reviewed the research on public perceptions and the importance and challenges of understanding risk perception. It also looked at the social impacts of microplastics and the role of individuals and groups and regional factors, including barriers and actions towards potential solutions.

The first assessment concluded that people see the health of the ocean as important and that pollution (more generally) is a key problem for the environment and for users of the marine environment. Microplastics specifically were seldom noted in surveys looking at individuals' perceptions, suggesting that the public had little awareness about microplastics. Using printed and digital media to infer public interest and concern, the increasing presence of news articles, microplastic-related searches online, and social media campaigns implied a growing trend that requires further investigation. Social and socio-ecological impacts of marine debris more generally were also identified, re-emphasizing the urgency of addressing this wider issue; as well as acknowledging the role of individual, group and geographical/cross-cultural differences.

The first report concluded positively on how people can be part of the solution through public engagement and formal and informal education, but emphasized the numerous research gaps. The four main areas identified as needing attention were:

1) Social research on a) current knowledge & understanding, b) perceived risks of microplastics, and c) the associated consequences of microplastics to society

2) Greater geographical coverage (research outside of Americas & Europe)

3) Investigation of the economic impacts of microplastics, in terms of cost-benefit to forecast future effects in response to any changes in microplastic use/input

4) Promote the collection and evaluation of examples of public engagement programmes (e.g. citizen science; beach cleans) in terms of their effects on perceptions and actions, including longitudinal follow-ups.

This new GESAMP chapter develops these points further, as well as presenting more insights on practical solutions – what can we do and what are the impacts of action and inaction.

6.2 Introducing the plastics economy

An overview of the flow of plastic through the economy is presented in Figure 6.1. It indicates the flow of plastics to consumers via goods and services, and to the wider environment, either under controlled waste

disposal or via release as waste to land and directly or indirectly to the sea. Much of the work published on the economics of marine plastic debris concerns macro-debris and is summarized in detail in an unpublished report prepared for UNEA by IEEP. The structure of Chapter 6 broadly follows the schematic, starting with a focus on producer responsibility and measures to address the problem (Section 6.3). Three sectors are then looked at in depth; Section 6.4 focuses on fisheries and aquaculture, 6.5 on shipping, and 6.6 on tourism and recreation. Next, the chapter discusses the cost of inaction and action for end-users (i.e. consumer behaviour, 6.7) and the waste management sector (6.8).

Drawing on the behavioural sciences, Section 6.9 provides an overview of some factors influential in encouraging long-term behaviour change, identifying the social costs and benefits of highlighted actions at different stages within the lifecycle of plastics (Sections 6.3 to 6.8). Collaborations across stakeholders on solutions are explored in 6.9, and Section 6.10 presents the conclusions, knowledge gaps and priorities.

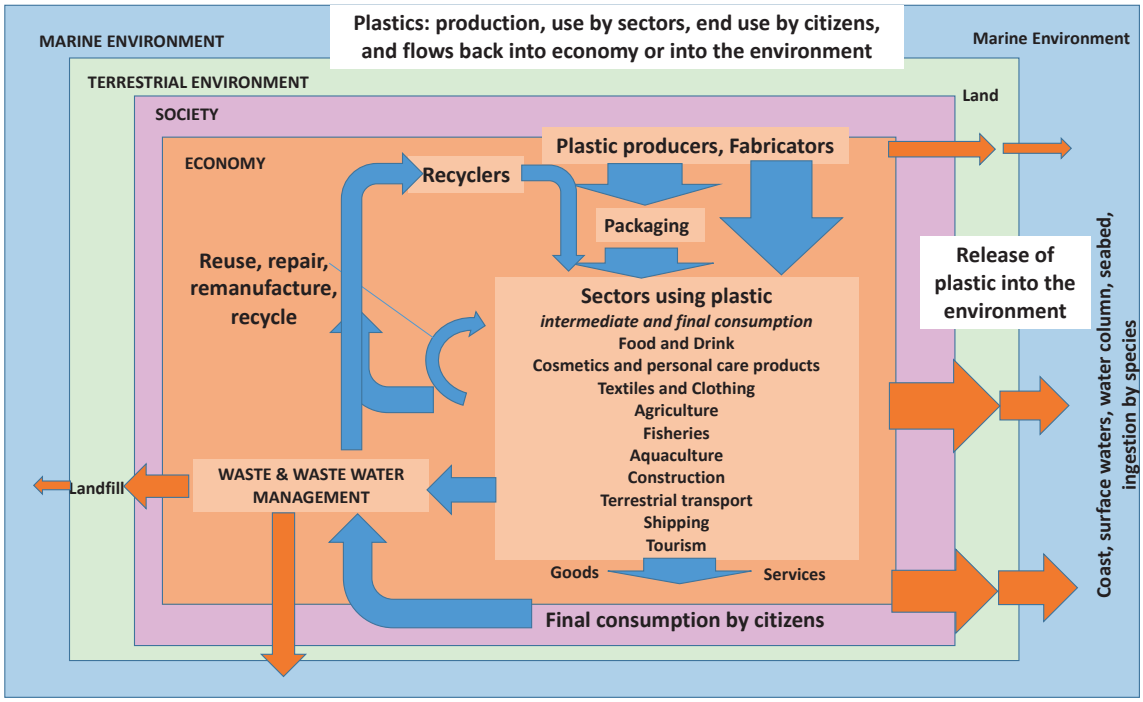


Figure 6.1 *Schematic of plastic production, consumption and waste management and losses (graphic devised by Patrick ten Brink, taken from UNEP 2016)*

6.3 Producer responsibility

6.3.1 *Initiatives and the cost of action to address microplastics*

There is a range of actions that producers can take, and have taken, to contribute to reducing or preventing microplastics entering the marine environment. They can occur across the product chain – from raw material use to production processes, quality of plastic produced, and product design. They can include initiatives characteristic of a more circular economy that facilitates reuse, repair, re-manufacture and recycle plastic (Figure 6.1). The need to establish a more circular plastic production cycle is discussed in more detail in UNEP (2016). Producers can also help in information provision to help intermediate consumers (i.e. other sectors) and final consumers (i.e. citizens) make informed choices. Finally, take-back schemes can encourage wider engagement (by both producers and consumers) in waste collection. Examples are given below.

In terms of their actual products, one of the main ways for producers to prevent marine litter is through sustainable product design. Products can be developed that are more recyclable so that they can easily be captured at the end of their lifecycle, or be designed with an end-of-life use already in mind. In many cases multi-use products are preferable to single-use ones (e.g. re-usable bags rather than single-use plastic bags) since they are less likely to be disposed of immediately. Voluntary initiatives involving groups of actors (such as the Beat the Microbead campaign) may provide motivation to act (e.g. producer initiatives to remove microbeads from personal care products), since they ensure that there are several organizations working towards the same goal.

Several examples of these various types of actions are included in Box 6.1.

The producers of waste that may become marine litter represent one sector that can bear the economic costs. In terms of environmental economics, the creation of marine litter is facilitated because the marginal price of goods on the market (and of disposable plastics in particular) fails to reflect the full marginal cost to society of producing that good: in short, marine litter has an external cost to society that is not adequately borne by the waste producer (or consumer). It is also easy for some waste producers to 'free-ride' (i.e. not contribute to litter prevention/clean-up costs whilst others do).

Box 6.1 Examples of producer initiatives that can help to reduce marine litter

In June 2015, a number of large UK brands and retailers announced their commitments to phase out non-biodegradable plastic microbeads from their own-brand cosmetics and personal care products. The commitments were made public on a voluntary basis through the Beat the Microbead campaign (launched in 2012 by the Plastic Soup Foundation) and the Good Scrub Guide (Fauna & Flora International (FFI) and the Marine Conservation Society). In parallel, legislation has been adopted to ban plastic microbeads from a large sub-set of personal care products, including in the United States and Canada.

The Operation Clean Sweep programme, an international initiative of the plastics industry, aims to prevent the loss of plastic pellets, flakes and powder to the marine environment through good housekeeping and containment practices by all parts of the plastics industry (producers, transporters and processors). A manual has been developed presenting best practice procedures to prevent, contain and clean up spills and losses of pellets, to make employees aware of both their responsibilities and how they can ensure they meet them. Implementing the measures in the manual will of course have cost implications; these costs are funded by those companies pledging their participation in Operation Clean Sweep, but no details are available publically on costs.

Extended producer responsibility (EPR)

One instrument that can be used to put responsibility on the producer is EPR, whereby a producer is made financially and/or logistically responsible for the post-consumer (i.e. waste) stage of a product's lifecycle, as encouraged by Surfers Against Sewage in the UK that currently have a Return to Offender campaign.[8] This concept has been widely implemented in EU and OECD countries, and in recent years emerging economies in Asia, Africa and South America have also begun developing EPR programmes (OECD, 2014). With regards to marine litter, perhaps the most important waste stream that should be addressed by EPR is packaging, since it forms a significant proportion of marine litter. Plastics used for packaging were estimated to require 78 million tons of material in 2013 (WEF, 2016), with a marginal 2% returning to remanufacture. Although 14% was estimated to be recovered, 4% is lost in processing and 8% downcycled to low quality goods. The remaining 86% goes to landfills or incinerators, or ends up in the environment.

Food wrappers and beverage containers (and caps) are regularly featured in the top ten most frequently found items during marine litter surveys; together these items comprised 31% of all items found during the Ocean Conservancy's 2013 International Coastal Cleanup. As a result of EPR for packaging waste, 64% of waste packaging (including composting for biodegradable packaging) was recycled in the 27 EU Member States in 2011, and 77% was recovered (including incineration with energy recovery). In Japan, the level of recycling of containers and packaging waste increased by 27% between 1997 and 2000 (OECD, 2014). The fees paid by producers to EPR schemes are mostly used to cover, or contribute to, the cost of collection and treatment of waste packaging, and only minimally to cover clean-up of litter.

EPR can also be used to promote environmentally friendly design, which helps to increase recycling and reduce waste. For example, implementation of the 2006 revision of the Packaging Recycling Act in Japan contributed to a significant switch by producers from green PET bottles to clear ones with green labels. This helped reduce the cost of collection by removing the need for green bottles to be collected separately and was also attractive to industry given the higher value of clear bottles. A 'bonus-malus' scheme was also introduced by the French organization Eco-Emballages that strongly penalizes (by up to 100% of their fee) producers that place non-recyclable packaging on the market, whilst reducing the fee by up to 8% for producers who reduce the weight or volume of their packaging (OECD 2014).

In Chile a new form of EPR has been proposed that differs from others in that it puts the burden of recovery on specific brands. For example, a producer of beverages bottled in plastic must recover a percentage of their bottles from consumers and the environment. This has two key motivators that can drive innovative design. First, producers of products and packaging will be incentivized to use biodegradable materials rather than plastic for single-use, throw away items, realizing that recovery from remote regions may be too expensive to warrant using plastic in these applications. Second, once producers recover their percentage quota of their plastic product, they must deal with it in some way. Designing their product for end-of-life material recovery and remanufacture is incentivized because the brand is responsible for dealing with their material directly (OECD 2014).

Additional actions may of course be taken by producers to address the problems of marine litter; these typically address downstream waste rather than upstream product design. The role of the waste management sector is presented in more detail in Section 6.7.

6.3.2 *Benefits of action by producers*

Examples of producer initiatives and their benefits are presented in the box below. These are downstream activities, dealing with final marine plastic waste. Whilst these initiatives illustrate the utility of such action, they only address a small fraction of (marine) plastic waste. Even if the quantitative effects on marine litter are very small it is possible that there are benefits in terms of raising awareness. Up-stream mitigation and recycling measures are likely to be more effective in reducing marine litter, and perhaps less powerful for public awareness.

There are also a number of initiatives where socio-economic benefits and/or value are generated from collected marine litter, whether in terms of employment

[8] http://www.sas.org.uk/campaign/return-to-offender/

created, collected recyclable materials (e.g. plastics) sold or upcycled products sold. Furthermore, value can also be generated by collecting and recycling plastic before it becomes marine litter, and often this results in higher values than those obtained from marine litter, given the higher quality (and hence value) of plastic that has not been degraded or contaminated in the marine environment (see Section 6.7).

Box 6.2 Examples of producer initiatives to address marine litter

Bureo Skateboards has created a skateboard deck (the 'Minnow' skateboard) made entirely from recycled fishing nets. This has been done through Net Positiva, a collection and recycling programme for commercial fishing nets in Chile, which was launched in 2013. Through Net Positiva, Bureo Skateboards 'harvest' the waste nets which are then melted down at a recycling plant in Santiago to be made into skateboards. To date, Bureo have recycled 10.8 m^2 of fishing net; each skateboard deck uses over 2.8 m^2 of fishing net.

Interface Carpets, together with the Zoological Society of London (ZSL) and yarn supplier Aquafil, created the Net-Works programme that incentivizes net recovery from beaches in the Philippines. These nets are remanufactured into carpet squares. The Net-Works programme in the Philippines (Danajon Bank and Bantayan Islands) has to date collected 77,792 kg of discarded fishing nets, 51,934 kg of which have already been absorbed into Interface's supply chain. The money from net purchases go into small community banks that provide access to finance (e.g. micro-insurance, savings and loans) for 358 local residents. It is estimated that the funds provided through Net-Works community banks to date would be enough to pay for 268,382 meals. In 2015, Net-Works expanded into a third Philippine collection hub (Northern Iloilo), and also established a programme in Lake Ossa in Cameroon.

6.3.3 *Conclusions*

There are costs associated with inaction; one estimate suggests that environmental damage to marine ecosystems caused by plastics is $13 billion per year (UNEP 2014). While producers implicitly share some responsibility for the cost of inaction, they will not however incur any direct costs of inaction related to marine litter unless EPR applies and they are therefore financially and/or logistically responsible for their products at the end of their useful life. Still, in many locations, once a product is produced and sold, it is no longer the concern of producers. This means that producers frequently do not have a cost-related incentive to take action on marine litter.

There is little data available on the costs of action by producers to prevent and tackle marine litter. Participation in initiatives such as the plastics industry's Operation Clean Sweep or voluntary commitments to phase out plastic microbeads from cosmetics and personal care products will of course have some related costs, but information on costs has not been found. It is therefore challenging to find information on the cost to producers of actions that help to address marine litter.

6.4 Fisheries and aquaculture

6.4.1 *Introduction*

The fishery sector is responsible for and negatively affected by plastic debris, predominantly macroplastics that degrade into microplastics (Figure 6.1). Consequently, when reviewing the cost of inaction and action of tackling microplastics, it is fundamental not to overlook this aspect.

Lost and discarded fishing gear poses a significant impact on ecosystems and wildlife populations and individuals, which may translate into a loss of catch. Abandoned, lost or otherwise discarded fishing gear (ALDFG) can end up indiscriminately catching target and non-target fish for a long time after it is lost at sea (a phenomenon known as ghost fishing) (Macfadyen et al. 2009; Brown 2005; UNEP 2009). With time, and the action of sunlight, waves and sea currents, they degrade into micro- and nanoplastics, which can also have negative impacts on wildlife, contaminate commercial fish and shellfish and potentially affect human health (see Sections 6.3 and 6.6).

Table 6.1 Overview of marine plastic and the fishing sector: sizes, types and impacts

Marine plastic size:	Micro <5 mm	Meso <2.5 cm	Macro <1 m	Mega >1 m
Sector as a source	Indirect: fragmentation of buoys, ropes, gear, nets	Indirect: fragmentation of buoys, ropes, gear, nets	Direct source: Fishing floats, buoys, ropes	Direct source: Abandoned fishing nets and traps; rope; boats;
Examples of marine litter that could burden fisheries	e.g. microbeads from personal care products; fragmentation of existing (plastic) products	e.g. bottle caps; plastic pellets; fragments	e.g. plastic bags; food and other packaging; fishing floats, buoys; balloons	e.g. abandoned fishing nets and traps; rope; boats; plastic films from agriculture

Marine plastic size:	Micro <5 mm	Meso <2.5 cm	Macro <1 m	Mega >1 m
Impacts on fisheries and aquaculture	Potential perceived (subjective) risk from presence of microplastics and associated chemical contamination in fish and shellfish; potential impact on fitness of fish/ shellfish and hence more costly to culture	Ingestion could lead to lower quality fish and hence lesser market value	Entanglement in propellers and damage to fishing vessels; related loss of fishing time, loss of fish and associated revenues	Ghost fishing: loss of output and hence livelihoods; collision with litter affecting safety or requiring boat repair.

6.4.2 *Economic impacts of marine litter on fisheries*

The impact of marine litter on fisheries is due to the damage to fishing vessels and equipment and to the reduction of potential catches and/or sales resulting from macro- and micro-plastics. In regards to macroplastics, impacts are largely due to floating objects affecting engine cooling systems and becoming entangled in propellers (McIlgorm et al. 2011; Takehama 1990). A summary of available information on the economic impact of macro-debris on the fisheries and aquaculture sector is provided by UNEP (2016).

In regards to microplastics, there may be an impact on fish stocks due to exposure through the gills or ingestion. Microplastics can also be transferred through the food web from one trophic level to the next, increasing the risk of exposure in a diversity of fish and shellfish products. The impact of microplastics on commercial fish species is still relatively unknown (see Chapter 5).

Microplastics can be a vector of transport of chemicals into marine organisms, including additives, monomers and by-products contained in plastic particles and organic chemicals and metals from surrounding seawater. For this reason, microplastics may have an impact on wildlife and human health (see Chapter 5 for more information). Concerns about this issue may cause a reduction in demand for the seafood products (see Box 6.3 for an estimation of the related costs in the UK). For example, if people perceive a risk or are unsure of the risks associated with seafood, they will have lower intentions to consume it (Boase 2015). As such, widespread concern could have major impacts on fisheries (see discussion on risk perception in Chapter 6.8 and GESAMP 2015).

Box 6.3 Potential economic losses to the UK oyster and mussel aquaculture sector due to microplastics

A model developed by van der Meulen et al. (2014) calculated a yearly loss of up to 0.7% of the annual income for the aquaculture sector in the UK due to microplastics. These costs relate to the impacts of microplastics on the mussels and oysters (chemical and physical effects) and in turn on human health (through the consumption of seafood), which can lead to reduced consumer demand and hence socio-economic costs through loss of sales.

6.4.3 *Cost of action to address microplastics*

Marine litter produced by the fishery sector, which degrades into microplastics over time, can be reduced using a combination of preventative and clean-up measures (Macfadyen 2009). Preventative measures aim to avoid the occurrence of marine litter. Examples include marking fishing gear to identify ownership, the provision of low-cost/free and easy-to-use collection facilities in ports, schemes for fishers to collect marine litter (Box 6.4), and spatial zoning to make other marine users aware of the presence of fishing gear.

Clean-up measures aim to remove marine litter from the sea. They include the use of on-board technology to enable location of gear (e.g. side scan sonar for sea-bed surveys) and gear retrieval programmes (Macfadyen 2009). Even though it would be impractical, dangerous, and too expensive to remove all ALDFG, programmes aimed at removing it in the most sensitive areas and in areas with demonstrated high loss rates would help address the problem. The costs related to ALDFG retrieval programmes may differ considerably, depending on the specific characteristics of the geographical areas, scope and duration. For example, Wiig (2005) reports estimates ranging from $65/tonne in Taiwan Province of China, to $25,000/tonne in the Hawaii Islands. Table 6.2 summarizes some estimates of costs related to ALDFG retrieval programmes.

6.4.4 *Conclusions*

Marine litter (both macroplastic and microplastics) may translate into a loss of catches and/or sales for fishers and therefore a cost for the sector. Debris in the sea also results in costs for the fishery sector due to damage to fishing vessels. A number of policies can be used to address marine litter, including preventative and clean-up measures. The related costs will depend on local specificities, but comparing their cost to the cost of inaction will certainly provide a good argument to strengthen the policies already in place and to implement new ones. Assessing the effectiveness and cost-efficiency of such policies is not an easy task because of the global character of marine litter. However, efforts should be made to estimate the monetary costs (e.g. loss of fish sales) and non-monetary costs (e.g. potential health risks) associated with microplastics.

Unlike other sectors addressed in this study, assessing the costs of marine litter for fisheries does not need additional use of monetary valuation methodologies, as fish already have a market value. This kind of analysis needs evidence regarding the changes in wildlife popu-

lations due to microplastics. Ecological studies should be complemented with surveys of fishers in different areas, investigating the economic loss from marine debris, including due to time lost to clean-up efforts, damage to fishing vessels/aquaculture installations and loss of catches and sales.

In many cases, such analysis will show that the costs of policies addressing marine litter are outweighed by the benefits in terms of increased income/reduced costs for fishers, in addition to shipping, tourist operators and other related sectors. For microplastics there is as yet too little information on economic costs of impacts to be able to draw a definitive conclusion.

Box 6.4 Initiatives involving commercial fishers for marine litter collection

Using financial incentives, South Korea

In 2002, the city of Incheon (Korea) established a financial incentive programme that rewarded fishers for collecting marine debris with a payment of $5 per 40 litre bag. The cost was estimated to be significantly lower than the cost of collection by the authorities of derelict fishing gear, i.e. a minimum of $48 per 40 litre bag (Cho 2005). Inspired by this experience, the Korean Ministry of Land, Transport and Maritime affairs has implemented a similar incentive programme since 2003 with a budget of $5.2 million per year between 2009 and 2013, and covered 80% of the related costs, the rest being covered by local governments. The programme collects an average of 6,200 tonnes of debris per year. Both financial incentives are still in place.

Fishing for litter, Belgium

In Belgium, it is not uncommon for fishers to find items of marine litter, generally a few kilograms with each catch. In some cases, trawler nets bring in very large items such as fridges and truck tyres. Stichting voor Duurzame Visserijontwikkeling (SDVO), the Belgian foundation for Sustainable Fishery Development, has a litter campaign called Fishing For Litter. It encourages Belgian fisherman to collect the waste they pull up in on-board containers provided by SDVO. SDVO organizes the collection of this waste in all three Belgian fishing ports, and sorts the waste for recyclability. The Fishing For Litter project is a voluntary cost-sharing scheme. Uptake is 60% amongst Belgian fishers, who pay a fee depending on the size of their vessel. Although free riders exist, the project covers its costs.

[Source: Interview with representatives of SDVO and Waste Free Oceans, June 2015]

Box 6.5 Case study summary – Fishing litter in Korea

According to Jang et al. (2014c), 48% of the marine litter found in Korean seas is derelict fishing gear. It is estimated that 60% of the fishing nets used in Korea are abandoned in the sea (Jang et al 2014b). Recent surveys show that EPS represents the most abundant debris item found on Korean beaches, covering a range of sizes from microplastics to macroplastic (Hong et al. 2014; Heo et al. 2013; Jang et al. 2014a; Lee et al. 2013).

Derelict fishing gear has a great impact on wildlife (Lee et al. 2015; Hong et al. 2013), which translates into losses for the fishery and aquaculture sector (Cho 2005), as well as reduced revenues for the tourism sector (Jang et al. 2014b) and numerous maritime accidents (Cho 2005).

In order to address the problem, the Korean Ministry of Maritime Affairs and Fisheries put in place strategies in 1999 to remove marine debris. The 2nd National Plan for Marine Litter Management (2014 to 2018) includes not only a clean-up programme, but also a survey of the status of marine debris, preventative measures to reduce the discharge of debris from land-based sources to coastal areas and the development of equipment and facilities for deep-sea survey, recycling and environmentally friendly disposal of collected material (Jung et al. 2010). In addition, the Ministry of Ocean and Fisheries has provided financial support to local governments to install EPS compactors, and to fishers to buy high-density buoys which degrade less readily into microplastics (Lee et al. 2015). Also, in 2009 the Ministry of the Environment established debris management and cost-sharing agreements in the five major Korean rivers with the local governments that share the same watershed, which resulted in local governments in upstream areas transferring funds to those located downstream for clean-up (Jang et al. 2014c). Finally, the city of Incheon and the Ministry of Maritime Affairs and Fisheries have put in place incentive programmes that remunerate fishers to collect marine litter.

The level of effectiveness and cost-efficiency of these policies is still to be evaluated through regular surveys to monitor the sources, type and location of marine litter, to assess the trend over time. Also, the costs of the different programmes in place should be analysed, and if possible compared with the observed results, to assess whether the available budget is used in the most efficient way and, if not, to suggest improvements. As an example of this kind of approach, Hong et al. (2015) compared the cost efficiency of clean-up by ships, fishers' incentives and floating reception barge (FB) programmes and suggested governmental policies should mainly focus on preventative actions such as FB.

Table 6.2 Estimates of costs related to ALDFG retrieval programmes

Costs	Location	Source
\$1,300 t^{-1}	Republic of Korea	Raaymakers 2007
\$1,680 t^{-1} for removing items using ships \$1,320 t^{-1} when buying back fishing gear through the incentive programme \$194 t^{-1} when providing a floating reception barge	Republic of Korea	Hong 2015
\$25,000 t^{-1}	North-west Hawaiian Island	Raaymakers 2007
\$4.2 million, to remove 34,408 derelict blue crab pots, generating \$21.3 million in additional revenue from reduced ghost fishing	Chesapeake Bay, USA	Scheld et al. 2016

6.5 Shipping

6.5.1 *Economic impacts of marine litter on the sector*

Commercial shipping represents an important sector for marine litter. Commercial shipping is also a source of, and is impacted by marine litter (Table 6.3). Estimates suggest that shipping is responsible for between 12% (IMO 2012) and 20% (EMSA 2013) of global discharges of waste at sea. Complex international, regional and national maritime laws provide a legislative framework, which forbids the dumping of plastic waste at sea. However, both accidental and deliberate waste dumping continues to drive socio-economic impacts, which bring costs upon the sector (Newman 2015).

The process of generating marine litter and its presence in general bring costs to the commercial shipping sector. The main costs are associated with collisions with marine litter that can result in accidental loss of cargos; and indirect costs relating to operational costs, disruption of service and public image. However, these impacts generally do not tend to stem from micro or nanoplastics, but from larger items.

Table 6.3 Potential economic impacts of marine plastics on the shipping sector

Marine litter size:	Micro <5 mm	Meso <2.5 cm	Macro <1 m	Mega >1 m
Impacts on shipping	Unlikely	Potential damage to vessels (e.g. cooling systems)	Damage to vessels (propellers, cooling systems); potential loss of productivity and revenues from delays or accidents affecting supply chains	Damage to vessels (propellers, cooling systems); potential loss of productivity and revenues from delays or accidents affecting supply chains

6.5.2 *Cost of action to address microplastics*

In order to adhere to maritime laws regarding waste there are a number of actions the shipping sector can carry out on-board vessels and on land. The effectiveness of waste management on-board and at port reception facilities largely dictates the levels of marine litter originating from the shipping industry (Sherrington 2014; Seas At Risk 2011). In addition, there are sporadic accidental losses of cargo from ships. For example, after Typhoon Vicente on 24 July 2012, over 150 tonnes of plastic pellets were blown into the sea and were washed up on southern Hong Kong coasts. Operational discharges from ships may also add to microplastic abundance in the ocean. Strategies for commercial shipping vessels to collect waste may also be considered, although are yet to proceed beyond prototype experiments.[9]

[9] The creation of the world's largest solar boat and the first to circumnavigate the globe, the MS **Tûranor** represents one example in efforts to explore how ships could engage in clean-up operations (Lombardo 2013)

6.5.3 *Conclusions*

The socio-economics of marine litter for commercial shipping reflects the scale of this industry and its tendency to minimize operational costs. The data on litter originating from ships suggests that shipping continues to contribute significantly to global levels of marine litter. The costs associated with marine litter suggest that the sector should make further efforts to reduce its impact on the marine environment.

Raising awareness about the costs of marine litter to the shipping sector could support better practices. Costs are associated with loss of cargo, collisions with waste, and legal action for dumping. Due to the dependency of global supply chains on logistics from shipping, the costs of disruption to services are considerable. The shipping and ports sectors, with appropriate governance, could be encouraged to develop and utilize improved waste management infrastructures on-board, improve port reception facilities, and take steps to reduce cargo losses at sea.

6.6 Tourism and recreation

6.6.1 *Introduction*

Socio-economic impacts on the tourism sector can be significant, particularly in areas that are heavily focused on coastal tourism which relies on a clean and pristine environment to attract visitors. The increased prevalence of marine litter reduces the aesthetic value of a location and affects recreational opportunities such as beach activities, surfing, fishing and diving. This leads to reduced visitors, which in turn leads to a loss of revenue and jobs in the tourism sector. Microplastic contamination requires costly clean-up activities and may pose health and safety risks to visitors. In parallel, the tourism industry generates waste and marine litter, which is especially concerning in Small Island Developing States (SIDS) that lack the necessary infrastructure for waste management.

Table 6.4 Potential economic impacts of marine plastics on the tourism sector

Plastic litter size:	**Micro** <**5 mm**	**Meso** <**2.5 cm**	**Macro** <**1 m**	**Mega** >**1 m**
Impacts on Tourism and Recreation	Only has an impact if microplastic pollution is integrated into beach labelling that is visible to beach users	Evidence of marine litter can discourage tourism and recreation on beaches, reducing income and/or well-being	Reduction in tourist and recreation numbers and thus income / well-being. Increased costs of clean-up to maintain activities. Damage to vessels (propellers, cooling systems)	Reduced income from polluted beaches. Increased costs of clean-up to maintain activities. Damage to vessels (propellers, cooling systems)

6.6.2 *Impacts of marine litter on the sector*

The visible presence of marine litter has an impact on the aesthetic value of beaches and shorelines. This visual dis-amenity can undermine some of the benefits associated with coastal environments (e.g. improved physical health, reduced stress, improved concentration (White et al. 2013)) and may be a reason not to visit certain coastal areas (Box 6.6). There is a strong relationship between the visible presence of marine litter in the water and recreational use (Fanshawe 2002). For example, the presence of marine debris affects recreational activities such as diving and snorkelling, fouling propellers and jet intakes of recreational boaters and affecting recreational fishers in terms of the contamination of catch, restricted catch, damaged gear etc. Marine litter has also been found to be harmful to visitors' psychological well-being, as when witnessing litter on the coast people felt strong negative emotions (e.g. sadness, anger) and it was seen to diminish the restorative qualities of the environment (Wyles et al. 2015).

Box 6.6 Examples of how marine litter influences beach choice

California, USA

A study of 31 beaches in Orange County, California, USA (Leggett et al. 2014) showed that marine debris had a significant impact on how residents chose beaches to visit. The study found that a 50% reduction in marine litter could generate $67 million in benefits to residents over a three-month period. It also found that reducing marine debris by 75% from six beaches near the outflow of the Los Angeles River would benefit users by $5/trip and increase visitors by 43% leading to $53 million in benefits.

Barbados

Beach litter has potential economic costs in terms of adverse effects on the probability of tourists returning to a particular destination (Schuhmann 2011). A survey of tourists in Barbados examined the relationship between the quality and cleanliness of beaches and the probability of return visits. The results of the survey indicate that the amount of litter seen and tourist perceptions of beach quality are significantly related to the probability of return visits, particularly for first-time visitors.

In addition to being unsightly, marine debris can pose health risks and hazards for divers, recreational boaters, fishers and other coastal visitors. Medical and personal hygiene items (e.g. disposable nappies, sanitary products) contaminate some locations.

Marine litter discourages visitors from going to beaches. Reduced numbers of coastal visitors leads to lost revenues to the tourism sector. This leads to a loss of revenue and jobs in the local and regional economy. This can have short-term (e.g. where a specific natural incident such as a flood or tsunami washes up marine debris on a beach) and/or long-term impacts (e.g. where consistent levels of marine debris damages the reputation and image of the area as a tourist destination thus discouraging private sector investment in new hotel developments (McIlgorm et al. 2011). These impacts can be quite significant in certain cases, particularly where local economies are heavily dependent on tourism. Hawaii and the Maldives are facing declines

in tourist numbers and associated revenues due to marine debris, particularly plastics that threaten to affect the reputation of islands as sought-after tourist destinations (Thevenon et al. 2014). Some studies provide quantitative estimates of the costs to the tourism sector of marine litter – see Box 6.7 for some examples. There is no clear evidence yet on the impacts of microplastics specifically on tourism and recreation.

Box 6.7 Estimated costs of marine litter to the tourism sector

Goeje Island, South Korea

A period of heavy rainfall which led to marine debris washing up on the beaches of Goeje Island (South Korea) is estimated to have led to $27.7 to 35.1 million (KRW 29,217 to 36,984 million) lost revenue in 2011 from over 500,000 fewer visitors. The lost expense/revenue per visitor was estimated to be $66 (2013 $) (Jang et al. 2014a).

APEC region

Damage by marine debris to the tourism sector in the Asia-Pacific Economic Cooperation (APEC) region has been estimated to cost $622 Million (McIlgorm 2009).

South-west Sweden

The presence of beach litter on the Skagerrak coast of Bohuslan (Sweden) decreases tourism by between 1% to 5%, equating to an estimated annual loss of approximately $22.5 million and 150 man-years of work to the local community. Local clean-up efforts are estimated to cost approximately $1.4 million (GBP 937,000) per annum. Thus, the total cost to the local economy is around $24 million (GBP 16 million) per year (Fanshawe and Everard 2002).

UK

Van der Meulen et al. (2014) estimated that annual costs to the tourism sector in certain sample regions of the UK could range from $2.3 million (GBP 1.4 million) to almost $823 million (GBP 500 million) in the 2010 to 2100 period. The study identifies Devon and Norfolk as relatively vulnerable regions. Total regional beach cleaning costs are projected to range between $188,735 and $2.5 million (GBP 100,000 and 1.5 million) per year.

6.6.3 *Cost of action to address microplastics*

Addressing marine litter in the tourism sector requires preventative and responsive measures, which have associated costs and responsibilities borne by different actors. For most municipalities, the potential impact of marine litter on tourism is the main motivation for removing beach litter, often providing a more powerful incentive for action than legislation (Mouat et al. 2010). The costs of clean-up activities associated with littering by coastal visitors can fall on local actors such as municipalities or private actors such as beach managers and hotel personnel. Given the importance of the tourism sector in many economies, there is a strong incentive to both public and private actors to ensure their beaches and marine environments are kept clean (McIlgorm et al. 2009).

Clean-up costs can be expensive, and in some cases pose an undue burden on local authorities. For example, the estimated coastline clean-up cost for the Ventanillas municipality in Perú is double the annual budget of the municipality for all public cleaning (Alfaro, 2006 cited in UNEP 2009). Revenues from taxes applied on the tourism sector and other recreational users of coastal areas (e.g. car park charges near beaches, fees on recreational fishers) can contribute to the costs of coastal clean-up, waste collection and treatment, helping to alleviate pressure on the budgets of local authorities. The willingness of tourists to pay such taxes is dependent on several factors including the age and income of tourists, and whether there is a link between the tax and litter control (Oosterhuis et al. 2014). In some cases, clean-up activities are motivated by the need to uphold certain certification standards and voluntary eco-labels and awards (Box 6.8).

At the same time, certain clean-up activities can have a negative environmental impact, e.g. mechanical beach cleaning can disturb nesting areas and remove components of the food chain (Surfers Against Sewage 2014). In addition, some clean-up activities may contribute to microplastics in the environment by breaking down macro litter rather than removing it.

Box 6.8 Eco-labels and certification programmes to support prevention and clean-up activities

Blue Flag Programme

The Blue Flag Programme is a voluntary eco-label scheme, which sets standards for water quality, environmental management, information provision, safety and services. The need to maintain Blue Flag status has been an important factor motivating clean-up efforts in countries across the world. For example, a survey in the UK found that 46.3% of municipalities removed marine litter to ensure that beaches in their area meet the criteria for the Blue Flag Awards (Mouat et al. 2010). The potential impact of microplastics on water quality and potential reputational risk to Blue Flag beaches was calculated to cost between 0.09% and 3.4% of tourism revenues in selected coastal regions in the UK with a business-as-usual tourism revenue of GBP 14.75 billion per year (Van der Meulen et al. 2014).

Green Coast Award, Ireland

Some municipalities undertake beach clean-up activities to pursue awards such as Quality Coast Awards, the Green Coast Awards and the Seaside Awards, relevant for smaller coastal resorts. For example in Ireland, the Green Coast Award is awarded to beaches that have a beach management plan in place and community engagement to meet standards in the Bathing Water Directive but do not have the infrastructure to achieve Blue Flag status.

Bandera Azul Ecológica, Costa Rica

In Costa Rica, the Blue Flag Ecological Program (Bandera Azul Ecológica) engages coastal communities in protection, clean-up and maintenance efforts. The award is granted annually based on performance against certain criteria covering water quality, waste management, facilities, safety and environmental education, with monthly monitoring to ensure continued maintenance.

Different measures, other than clean-ups, are likely to attract varying degrees of public and political acceptance. For example, in a survey of beach visitors in Chile, the two most supported solutions to the problem of beach litter were community-level environmental education programmes and fines (Eastman 2013). Certain regulatory measures such as bans and fines may be politically sensitive to introduce and their enforcement challenging (i.e. requiring resources and legal capability). However, support could be built through targeted campaigns (i.e. creating peer pressure). For example, in the US, despite initial polarization of local communities to bans on smoking on beaches, people now generally support the smoking bans (Ariza and Leatherman 2012).

Box 6.9 Case study summary – marine litter in Hawaii

Marine debris is considered an important issue in Hawaii and has attracted significant attention from policy makers, private actors, NGOs, academics and the public. Land-based sources include improper waste disposal practices, tourism and recreational activities such as coastal recreational fishing.

Economic impacts include costly clean-up activities with estimates varying from an average \$589 t^{-1} to clean-up marine debris from the coastline (Lamson 2011) to \$25 000 t^{-1} to remove entangled nets from ships at sea in the NWHI (Wiig 2005). Other impacts include potential effects on the tourism industry, for example affecting recreational activities such as diving, posing a health and safety risk to coastal visitors, reducing the attractiveness of certain beaches etc., thus threatening to undermine Hawaii's reputation as a sought-after tourist destination. Although research on such linkages is limited, impacts could be significant given the importance of tourism to the Hawaiian economy.

A number of preventative and responsive measures and approaches have been adopted over the years including strategic measures such as the Hawaii Marine Debris Action Plan and pioneering legislative approaches at both State and County level such as bans on smoking on beaches and on plastic bags. The public and civil society has been very active in initiating clean-up activities, awareness raising campaigns and educational programmes and contributing to data collection, monitoring and reporting exercises.

Despite progress to date and the adoption of a range of innovative measures targeting marine litter, further action can be considered (e.g. effective preventative measures in developing countries) and additional research including on the socio-economic impacts of marine debris on specific sectors of the economy, in particular the tourism and fishing industries is needed. Such assessments can inform policy discussions and provide a further motivating factor for effective action on marine litter.

6.6.4 *Conclusions*

Whilst little is known about microplastics specifically, the tourism sector is significantly affected by marine litter and a major contributor of the debris. The presence of marine litter can discourage visitors from going to beaches, thus reducing visitor numbers, which leads to lost revenues and jobs in the tourism industry. These impacts can be quite significant in certain cases, particularly where local economies are heavily dependent on tourism. Moreover, marine debris can pose physical and mental health risks and safety hazards to recreational users of the marine environment.

The potential impact of marine litter on the tourism sector provides a powerful incentive to public and

private actors to keep beaches and marine environments clean. Responsive measures such as clean-up can have significant associated costs and in some cases can pose an undue burden on local authorities. However, the potential impact of marine litter on tourism and the need to uphold certain certification standards, voluntary eco-labels and awards provides a powerful incentive for action by municipalities. Some pressure on the budgets of local authorities can be alleviated by sharing clean-up costs with certain private actors (beach managers, hotels), supported by voluntary efforts by local community groups and NGOs, and using revenues from taxes on the tourism sector to contribute costs of coastal clean-up and waste collection and treatment.

It is also important to address tourism and recreational activities as a source of marine litter. A lot of marine litter is generated from shoreline and recreational activities (Ocean Conservancy 2010) and there is a need for various preventative measures to address the problem at source. Such measures can include regulation (e.g. smoking bans on beaches in a number of other US states), infrastructure investments (e.g. pier-side reception facilities for fishing gear in Hawaii, improved waste management practices supported by members of the Roteiros de Charme Hotel Association in Brazil and the Caribbean Alliance for Sustainable Tourism – CAST), product design requirements, targeted awareness raising and educational activities (e.g. boating safety education classes).

Assessments of the costs of marine litter on the tourism sector and assessment of impacts of tourism activities on marine litter are currently limited to small and localized studies. Further research is needed at a larger scale. It is important to measure changes in revenues to the tourism sector and identify to what extent any decline can be attributed to marine debris. There is also a need for further information on the costs (and benefits – economic, environmental and social) of prevention and clean-up activities (undertaken by both public and private actors, voluntary organizations and local community groups), information on health and safety risks from marine debris (in terms of exposure, accidents, mortality) and associated costs of hospitalisation (if relevant) as well as more intangible non-monetary costs (see Section 6.8). An analysis of the role that the tourism industry has on fighting and reducing this phenomenon is also needed.

6.7 Consumer behaviour

6.7.1 *Introduction*

The behaviours of citizens as consumers of goods and services are a major contributor to marine litter. Purchases of consumer goods such as plastic bags, plastic beverage containers, cosmetics and health care products that contain microbeads and synthetic textiles creates plastic in the waste stream that may become marine litter. Behaviour includes everyday activities at home and work and periods of recreation and tourism. This section focuses on general consumer behaviour, impacts and measures; the issues of tourism and recreation are addressed in the section above.

Citizens may also be at risk from microplastics, e.g. from microplastics in personal care products or fish and shellfish. Citizens are therefore both sources and sinks of marine plastic.

6.7.2 *Impacts of marine litter on citizens*

Marine litter can result in direct and indirect costs due to inaction (McIlgorm et al. 2009). Impacts on human health and well-being can occur through direct contact with debris which may lead to a direct cost of medical treatment (Hall 2000; ARCADIS 2012). Many of these impacts are not fully understood yet, including their magnitude on a global, regional and local scale, and the ensuing burdens and cost implications. In addition, it is difficult to identify groups that are most affected by marine litter (e.g. in terms of geographical location, age, level of wealth, level of education), due to a lack of study.

The impacts to health and safety from marine litter may result from immediate contact with litter, e.g. on beaches or in coastal waters, or indirect contact through the food chain. Microplastics enter the food chain when ingested by marine animals. Once humans consume an animal, e.g. fish, they may ingest microplastic particles and/or hazardous chemicals associated with it (Engler 2012). In addition, the presence of marine litter on the marine environment may undermine psychological benefits provided by nature, especially if the litter is seen to be from the fellow citizens (public litter) rather than a natural by-product of a marine industry such as fishing-related debris (K Wyles, personal communication).

The prevalence of discarded waste, especially plastic objects, in many communities has been linked to recent outbreaks and rapid spread of mosquito-borne diseases such as Dengue fever and Zika virus. Plastic waste provides an ideal breeding ground for mosquitoes following rainfall. SIDS in the Pacific[10] and Caribbean, countries in West Africa and many countries in South and Central America have reported sharp upturns in disease incidence (UNEP 2016).

Marine litter can also produce indirect costs to human beings. Indirect costs can occur in the form of visual impairment of littered beaches, shorelines and marine environments, which lower the recreational value of sites to visitors and local residents and result in additional costs as visitors relocate to alternative sites (McIlgorm et al. 2011; Birdir et al. 2013). Degradation of marine and coastal ecosystems through litter can lead to further disutility, having negative impact on human health by undermining some of the broader benefits associated with the recreational use of coastal areas (e.g. reduced blood pressure, tension and stress, improved level of concentration) (White et al. 2013). To address this issue, local authorities engage in costly clean-up activities.

[10] http://www.ipsnews.net/2013/05/dengue-outbreak-highlights-poor-waste-management/

6.7.3 *Cost of action to address microplastics*

It is important to address the many roles of consumers, littering contexts and littering pathways, when thinking about actions to address mitigation of microplastic litter. Ideally, activities will reduce consumption of single-use items and encourage the reuse of plastic products. The following sections provide background about informing and empowering consumers to change their behaviour, as well as looking at prevention of waste and littering in coastal and marine zones, and activities to collect litter in the environment.

Prevention of waste generation

There are a number of different approaches for reducing the generation of waste. Policymakers can address this by discouraging practices that generate litter or limiting the use of products that contribute to marine litter. Single-use products warrant attention as they are among the items most frequently found in the marine environment. Items like single-use plastic bags, food or beverage containers can be addressed via education and outreach and with economic instruments (Oosterhuis et al. 2014). Such instruments, such as fees, charges or taxes, can provide a disincentive by penalising undesirable practices. They can also be addressed by direct bans on their use (Box 6.10).

Taxes and levies generate revenues that can be used for addressing consumer behaviour, but they can also be controversial in regards to the use of revenues and their impacts on individuals' attitudes and behaviours (Section 6.8). Alternative ways of providing incentives for the desired behaviour include discounts for environmentally friendly behaviour of consumers, such as deposit schemes that give money as a reward when materials are returned for recycling.

Economic incentives can be powerful to discourage undesirable practices, especially when they are communicated as monetary losses. Some instruments such as levies also generate revenues that can be used for financing activities such as improving waste collection infrastructure and awareness campaigns. However, they come with the caveat that they actually reveal that certain practices such as uncontrolled disposal of waste are common or 'normal'. Alternatively, initiatives can also emphasize and promote behaviour that reduces littering. For example, promoting reusable cups or refillable water bottles can help establish a culture of reducing waste.

Box 6.10 Bans on plastic bags

In 2012, Hawaii introduced a county-by-county ban on plastic bags implemented over three years over the entire state. California passed the first state-wide ban in 2014, but has been challenged by industry and has been subject to a referendum in 2016. Several cities and counties in Oregon and Washington have implemented plastic bag ordinances (Stickel et al. 2012). Bans on certain types of single-use plastic bags have been introduced in several other countries across the world to varying degrees of effectiveness, for example Bangladesh, Rwanda, India, Italy and Kenya. A study on the cost of banning plastic bags in Los Angeles County concluded that the ban would cost $5.72 per capita (AECOM 2010).

Tackling less visible forms of marine litter is challenging, as the information requirements for consumers are higher than with visible litter items. For example, personal care products, such as cosmetics and toothpastes, frequently contain microbeads as an exfoliating agent (UNEP 2015). At a glance, it is not straightforward for consumers to detect plastic ingredients in personal care products and further outreach was needed to change consumer behaviour.

A tendency that has been growing during the last few years concerns the use of social media technology (e.g. Facebook™, Twitter™ and YouTube™, etc.) to reach a wider audience, see Box 6.11.

Box 6.11 Informing and empowering consumers by social media

The "Beat the Microbead" campaign quickly gained momentum in 2012. It specifically targeted microbeads used in personal care products and provided a smartphone app to identify products with microbeads. The initiative led to many manufacturers and retailers rethinking their product policy. Originally an initiative of two Dutch NGOs (the North Sea Foundation and the Plastic Soup Foundation), the initiative gained wider support by environmental and consumer groups, and is now supported by UNEP (www.beatthemicrobead.org).

The Marine Debris Tracker (MDT) is a partnership of the NOAA Marine Debris Division and the Southeast Atlantic Marine Debris Initiative (SEA-MDI). First released in 2011 and updated in 2014, this app was created by Jenna Jambeck to raise awareness about marine litter and help NOAA collect information about the position and condition of marine litter. The MDT and associate web platform aim at engaging citizens in a positive manner: they can expand their dedication to an issue, can also feel empowered by collecting and presenting data in the MDT community (Jambeck et al. 2015). The app has global coverage. There have been 12,000 downloads, over 62,400 entries with 539,700 debris items logged (Jason Rolfe, Mid-Atlantic and Caribbean Regional Coordinator, personal communication). Although the main activity of the app is linked to reporting chronic debris, it has also developed a special role in tracking marine litter reaching the US beaches after Japan tsunami and Superstorm Sandy. The app does not require users to collect litter after having reported it (and hence creates a risk of double counting) (http://www.marinedebris.engr.uga.edu/).

Prevention of littering behaviour

Information and awareness campaigns are crucial to tell consumers about their contribution to marine litter. This is especially the case for marine litter sources and pathways that are less obvious. These activities need to identify littering contexts and specifically address certain consumer groups and ages, such as school-children, outdoor travellers or interested citizens who want to engage actively, including as citizen scientists (Eastman et al. 2013).

Box 6.12 Role of ambassadors in awareness raising

Throughout the years, several celebrities have become ambassadors for the protection of the marine environment. In 2014 Lewis Pugh, United Nations Patron of the Oceans, swam in the Seven Seas to draw attention to the health of the oceans (UNEP NEWS CENTER 2014). In 2015, the swimmer Federica Pellegrini took part in the campaign "Ma il Mare non vale una cicca?" ("Isn't the sea worth a butt?"), organized by the Italian association Marevivo and the famous surfers Ben Skinner and Corinne Evans took part in the 'Save Our Seas' Marine Litter Tattoo Campaign, organized by Surfers Against Sewage (SAS, http://www.sas.org.uk/news/campaigns/save-our-seas-marine-litter-tattoo-campaign/).

In the same year, Jack Johnson was appointed Goodwill Ambassador by UNEP. The singer declared that he would focus his activity in particular on three issues; one of them being marine litter (http://www.unep.org/gpa/news/JackJohnsonGWA.asp).

Ambassadors for raising awareness need not be internationally known celebrities. In fact, their effectiveness as a voice for taking action against marine litters stems from their role as model and inspiration for the local community or region of interest. For example, Mama Piru, a native Rapa Nui woman from Easter Island, has become famous for her commitment to cleaning up the coast every day. She has been fulfilling her promise for the last 15 years.

For the most problematic types of marine litter, bans on certain products and activities are conceivable. These include restricting smoking on beaches, banning plastic bags with certain product characteristics, or banning products containing microbeads. For example, the US has signed legislation to ban microbeads. A study by Environment Canada (2015) recommends classifying microbeads as a toxic substance under the Canadian Environmental Protection Act.

Collecting litter from the marine environment

While efforts to reduce sources of marine litter are important, clean-up activities remain necessary. Local or regional authorities are often in charge, but a number of initiatives exist that explicitly involve volunteers (Table 6.5). The role of such engagement is two-fold. Clean-ups can help reduce the physical amount of litter entering streams, waterways and oceans. In addition, they are an important tool for bringing together communities and stakeholders to generate a common sense of action, raise awareness and create ownership.

Table 6.5 Selected examples for clean-up activities involving volunteers

Initiative	Regional coverage	Remarks
International Coastal Cleanup	International	Organized by Ocean Conservancy, takes place all around the world.
Clean-up SA Month	South Africa	Its aim is to increase awareness by educating the community about the social, environmental and economic benefits of recycling.
Marine Litter Project	Greater Caribbean region	The objective of this project was to assist in the environmental protection and sustainable development of the Wider Caribbean region through the implementation of the "Regional Action Plan on the Sustainable Management of Marine Litter in the Wider Caribbean" (RAPMaLi).
Clean Up Australia Day	Australia	The campaign started in 1989 when Ian Kiernan decided to clean up the Sydney Harbour. Starting from that, the campaign has kept growing until becoming the nation's largest community-based environmental event.
Beachwatch	UK	It is a popular national beach cleaning and litter surveying programme organized by the Marine Conservation Society to help people all around the UK to care for their coastline.
Clean Up Arabia	UAE (United Arab Emirates)	Organized by EDA, Clean Up Arabia is an annual voluntary campaign that aims to clean up the dive sites and beaches of the UAE and surrounding regions. At the end of the activity, the participants receive a certificate attesting their participation to the event.

Although activities carried out by volunteers are fundamental because they help local municipalities and alleviate costs, it is important to underline that their participation to a clean-up event constitutes an opportunity cost. It has been estimated that the participation of volunteers in two of the largest clean-up schemes in the UK, MCS Beachwatch and Keep Scotland Beautiful National Spring Clean, is worth approximately $173,500 (EUR 131,000) (Mouat et al. 2010).

6.7.4 *Conclusions*

Engaging consumers is crucial to addressing microplastics. Drawing links to the risks to human health and well-being can be an important step to raise awareness regarding the role that consumers play. It is important to empower consumers, show potential solutions and support concrete action.

The prevention of waste generation and littering are important and widespread awareness raising activities are needed. Consumers require information about product sustainability. Modern technology like smartphone apps and social media help reach wider audiences, though traditional channels of reaching out to consumers such as print media remain important. Equally, engaging consumers in clean-up activities is an important element to create awareness and interest in helping address the issue, especially when targeting different groups and demographies such as schoolchildren, coastal residents and visitors.

Ideally, behaviour change would be based on intrinsic motivation (see Section 6.8). For this reason, it is important to promote desirable practices to establish the default behaviour that would be in line with addressing marine plastic litter. One example is facilitating and removing barriers to the reuse of everyday items such as cups, bottles or plastic bags. Providing easy-to-use waste disposal facilities for waste is another. Such activities are not only important in coastal zones, but also further inland, as a high share of plastic litter occurs on land.

Economic instruments and incentives can play a role in addressing marine plastic litter. They are, for example, suitable to limit the amount of plastic items in use. Decision-makers need to be aware of mechanisms that address marine litter, such as fees, charges, taxes and deposit refund schemes (ten Brink et al. 2009). Sometimes, such instruments will focus on the price mechanism, i.e. making undesirable practices more expensive to discourage them. In other cases, the focus might be on generating revenue to finance activities related to marine litter, such as improving collection infrastructure or awareness-raising. In practice, these two mechanisms might be used in combination. However, decision-makers also need to be aware of the limits of economic instruments, as some evidence exists that external economic drivers of behaviour can have unintended and unwelcome consequences (see Section 6.8).

6.8 Waste management and recycling

6.8.1 *Introduction*

Waste management practices and infrastructure are critically important to address marine litter (Jambeck et al. 2015). Where there is a lack of such infrastructure, there is a high risk of microplastics entering the marine environment. Figure 6.2 presents an overview of the coverage of solid waste management globally.

While there are reports of management and mismanagement of plastics within national borders, there is a mismanagement of plastics through the trade and transport of plastic as a commodity. The discussion of land-to-sea plastic leakage has omitted the land-to-land leakage. In many cases, plastic products and packaging that have few or no markets in the US and Europe are exported to countries, like China or India, that have less stringent or no controls on environmental contamination and worker health and safety. In 2012 China implemented the "Green Fence"[11] in an effort to reduce the import of low-quality plastic products and packaging, mostly originating from the west coast of the United States, that were becoming marine litter after China had little use for them as well. What the Green Fence has done for US exporters of plastic waste, mostly from recycle centres that find it cheaper to export waste than pay landfill tipping fees, is to force US cities to revisit the full lifecycle of plastic products and packaging.[12] The global trend is to 'clean-up' waste exports, which may further catalyse EPR for upstream design for end-of-life recovery.

The wastewater treatment and water supply sectors are important – both as means of reducing marine litter and as sources thereof (e.g. small plastic biofilters that provide a physical structure to support bacteria in water purification plants[13]). Wastewater treatment plants can capture significant amounts of plastic waste and the existence of a water supply infrastructure that provides citizens with safe drinking water reduces the demand for (plastic) bottled water. Recycling of plastic can both avoid the generation of marine litter and reduce plastic already in the sea when collected and recycled. This section focuses in particular on recycling initiatives.

6.8.2 *Benefits of action*

Well-developed waste infrastructures can help reduce marine litter. In parts of the world where such infrastructures do not exist or are inadequate, some initiatives are being introduced to provide waste management at a very local or community level, as a means of tackling litter problems. A small number of examples are included in Table 6.6.

[11] http://www.plasticsnews.com/article/20150519/NEWS/150519925/recyclers-expect-more-china-green-fence-actions

[12] http://www.pri.org/stories/2014-02-18/chinas-green-fence-cleaning-americas-dirty-recycling

[13] http://www.keepersofthecoast.com/biofilters-lake-geneva-investigation/

Table 6.6 Examples of socio-economic benefits/value generated from collecting and recycling marine litter

Activity	Socio-economic benefits/value generated
Plastic recycling industry	*Employment:* Over 6,000 formal jobs and over 47,000 informal jobs in South African plastics recycling (Motsoai 2015)
Small-scale/local waste collection initiatives	*Value to citizens:* Points/money gathered by individuals to be spent on household items, food, clothing, mobile phone credit (e.g. TrashCash in Ghana, Wecyclers in Nigeria, Recycle Swop Shop in South Africa).
Trash for treasure initiatives	*Employment created:* 100 people employed by Ocean Sole (Kenya); Goal to create 100 direct and 500 indirect jobs through EcoPost Ltd (Kenya); 20 people trained in craft skills through Kriki4Shore (South Africa)

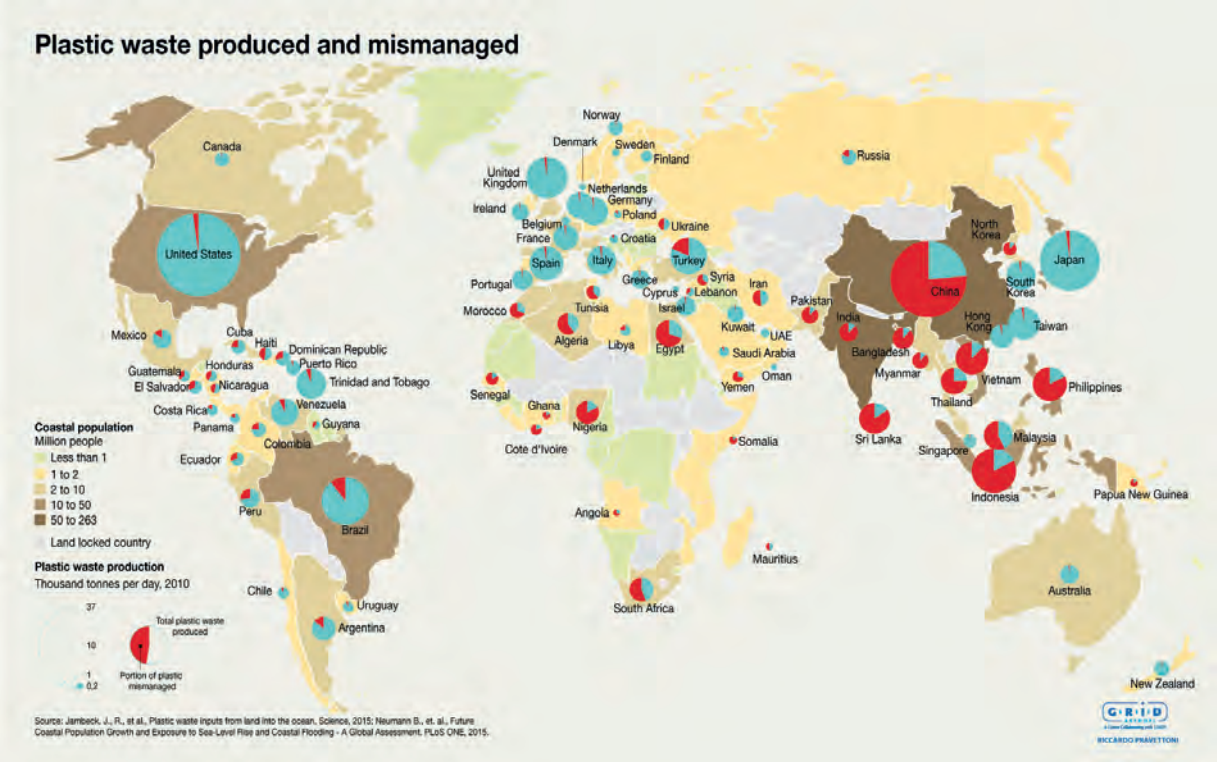


Figure 6.2 *Estimated quantities of plastic waste produced and mismanaged, based on Jambeck et al. (2015) (image courtesy of GRID-Arendal)*

Box 6.13 Wastewater treatment and microplastics

Wastewater streams transport microfibres, for example from textiles, into the sea (Browne 2011), since they cannot be retained by all existing wastewater treatment systems or traditional washing machine filters. Some modern systems are capable of capturing even above 90% of microfibres, though this still allows some significant leakage of microfibres into the aquatic environment.

Options to address microfibres could include attaching filters to washing machines, and innovative technologies such as additives for laundry detergents or textile finishing treatments to reduce the release of plastic microfibres during the washing process (Life-Mermaids Project, 2015).

Recycling captures value from used materials as well as offering the potential to create jobs. Plastics SA reported that 1,084,400 tonnes of plastic waste was landfilled in South Africa in 2014, whilst 1,400,000 tonnes (22.5% of plastic waste produced) were recycled; 32.9% of plastic packaging material was recovered. The company has set up over 450 fishing line collection bins across Cape Town to facilitate recycling.[14] According to

[14] Personal communication, UNEP

Plastics SA, the informal sector of the plastic recycling industry employs 47,420 people, whilst 6,037 people are employed in the formal sector; this represented an 11.4% increase in jobs in the industry in 2014 compared with the previous year. The majority of the 221 plastics recycling companies and estimated 1,800 converters in the industry in South Africa are small, medium and micro enterprises (SMMEs) (Motsoai 2015).

6.8.3 *Conclusions*

There are a number of initiatives around the world where socio-economic benefits and/or value are generated from collected marine litter, whether in terms of employment created, collected materials (e.g. plastics), or profit from recycled products sold. Some of these values are summarized in the table below; these should not be taken as comprehensive, but they do give a snapshot of several examples where marine litter is converted into socio-economic value.

6.9 Moving towards effective long-term behaviour change

6.9.1 *Introduction*

Within this chapter, the general costs of microplastics and marine litter have been summarized for a range of sectors. To fully understand the cost-effectiveness of initiatives and campaigns that aim to reduce the amount of waste entering and remaining in the marine environment, it is fundamental to acknowledge the underlying, influential factors. Understanding determinants of behaviour and behaviour change – identifying these necessary factors that are fundamental for interventions to be effective and long lasting – is a large and continually growing research field. Whilst it is not within the scope of this report to comprehensively review all influential factors, we have described some of the factors relevant for the initiatives described. Please see Darnton (2008) for a more detailed review. Below, we focus on five factors that seem relevant in the present context: the importance of risk perception, perceived responsibility and behavioural control, social norms, motivation and demographic factors.

6.9.2 *Risk perception*

The first assessment report summarized research on risk perception principles and applied these to marine litter and microplastics. In many cases, a perception of some degree of risk is required to engage people in the issue and trigger behaviour change. In this subsection, we provide an update on media reporting since the first report, which is a common source of information that individuals use to develop their risk perceptions. From this, we introduce key findings from the risk communication literature; especially the mental models approach used in interdisciplinary research (Morgan 2002) and we draw on research examining risk perception regarding nanoparticles and nanotechnology.

The first report showed a growing trend over time for the terms 'microbeads' and 'microplastics' to be mentioned in UK newspaper reporting. Monitoring July 2004 to July 2014, we found 29 articles over ten years, with 26 appearing since 2012. Updating this exact analysis, the most recent year (July 2014 to July 2015) already contained 25 media articles in total; a further four have appeared between July and October 2015. In addition, we coded whether these 29 new articles focused on the problem ($n=9$), the solution ($n=9$) or contained both ($n=10$).[15] Thus, the trend for more exposure on the topic in the media is continuing, with coverage of both on the problem and solutions. To promote behaviour change, this is especially important, as individuals need to perceive the relevance of the issue and how their actions can help (Tanner and Kast 2003). Still, empirical research on public risk perception of microplastics and nanoplastics is still lacking.

Most risk perception research is purely descriptive, monitoring perceptions in certain stakeholder groups (most often the public). Granger Morgan et al. (2002) mental models approach is different in that it starts with risk perception, but assesses this for the explicit purpose of risk communication. The goal of this approach is to make recommendations for empirically informed and targeted messages. The approach is designed for interdisciplinary risk contexts and explicitly deals with potentially disparate 'expert' and 'non-expert' views. The model contains three broad steps combining qualitative and quantitative social research: a) create and compare expert and non-expert mental models to identify discrepancies, b) test non-expert models in representative large-scale surveys, c) develop and evaluate risk communication based on insights from the first two steps. Applications of this approach show that expert and non-expert views often differ, that communication materials benefit from repeated piloting in target audiences and that targeted communications can improve understanding. For example, Boase (2015) investigated perceived risks and benefits of consuming shellfish using the mental models approach. He showed that the more uncertain people were about shellfish facts, the less likely they were to consume shellfish. A targeted mental models communication improved knowledge, reduced uncertainty and increased consumption intentions compared to alternative communications. Note that microplastics did not emerge as a theme in this work, which concentrated on health risks and benefits. This means that at this time, microplastics were not seen as a salient threat to seafood consumers.

Another aspect that was not covered in the first report is nanoplastics. Nanoplastics are an emerging issue because monitoring methods have not been developed yet and the scale of industrial production is unclear. However, there has been a literature around the perception of "nanotechnologies" in the social sciences since the early 2000s. As opposed to other contested issues in new technology development (e.g. GM foods), public opinion on nanotechnologies appears to be largely positive, with "discussion of risk issues […] relatively limited so far" (Pidgeon and Rogers-Hayden 2007). Satterfield et al. (2009) provides a meta-analysis of recent studies about public perceptions of nanotechnology. Their key findings are that the majority of people surveyed in the US, UK and Canada believe the benefits outweigh the risks of nanotechnologies,

[15] One article did not contain enough information for coding.

but more than 40% are unsure. This uncertainty is still present in recent work and has been linked to high fragility and mobility of attitudes (Satterfield et al. 2012). This is a considerable societal risk because new information or a future risk event has the potential to change public opinion rapidly in the case of such unstable attitudes. Pidgeon and Rogers-Hayden (2007) report on the outcome of a public engagement process ('NanoJury UK') that led to recommendations, which include more testing of new materials, communication in plain English and clear, plain-language labelling of products that contain nanotechnologies. At the international level, the International Risk Governance Council has produced a White Paper (2006) and Policy Brief (2007) including recommendations (IRGC 2015). It is striking that no such research or debate exists to date for microplastics let alone nanoplastics specifically.

6.9.3 *Perceived responsibility & behavioural control*

In many theories of behaviour change, two key factors are noted as important: perceptions of responsibility and perceived control or efficacy (Steg et al. 2012). Out of two people who have limited control over an issue, the one who has higher *perceptions* of control is more likely to act. For example, marine litter initiatives that provide individuals the ability to dispose of waste (e.g. floating reception barges, Section 6.3) or recycle their fishing lines (e.g. *Reel in and Recycle)* will strengthen the perception of control and thus encourage positive behaviour (Steg and Vlek 2009).

Perceived responsibility is also important in the context of marine litter. Large-scale surveys within the MARLISCO project show that general public respondents perceived sectors to vary widely in responsibility. Industry, decision makers (such as government and policy makers) and commercial users of the coast were viewed to have high responsibility. However, the respondents also held themselves responsible. This indicates that people were not discarding their own responsibility (Hartley et al. in preparation). This is fundamental, as a precedent of action is the perceived obligation or responsibility of the individual (Steg and Vlek 2009). A second MARLISCO study showed that a targeted educator training on marine litter was able to increase perceptions of own responsibility significantly, even in an already interested, selective sample (Hartley et al. in preparation). Given the many sectors and actors involved in the issue of marine litter, another promising example is the programme Amigos del Mar (Friends of the Sea) in Ecuador, led by the Comisión Permanente el Pacífico Sur (CPPS), which targets students, fishers and tour operators as key influencers. It is clearly necessary to engage all sectors, emphasize their responsibility (e.g. by illustrating the cost of action and inaction) and work cooperatively to help address the problem of marine litter (see Section 6.9).

6.9.4 *Social norms*

Behaviour change can be influenced by a number of social factors, including social norms. These can be what the individual thinks is common practice and is widely accepted (descriptive norms) and what ought to be done in society (injunctive norms). Marine litter in itself can be seen to indicate a descriptive norm: the presence of the items imply that it is normal practice and thus acceptable to litter, which will likely lead to more littering behaviours (Keizer et al. 2008). A number of psychological studies in terrestrial environments have repeatedly shown that people are more likely to litter if a) the setting is littered (descriptive norm) and/or b) if they witness someone litter (injunctive norm). On the other hand, these social norms can also encourage desired actions, as littering reduces and removal of rubbish increases if people are in a clean setting and/or if they witness someone picking up and throwing away rubbish (Keizer et al. 2008).

A number of the initiatives referred to within this chapter can be seen to include components of social norms. Social media campaigns can have both injunctive and descriptive norms. For instance, the Beat the Microbead campaign explicitly focused on the problem and what ought to be done (see Section 6.3) but it could be argued that the social media activities set social norms. The site has more than 1200 Twitter followers and more than 5500 Facebook likes (Plastic soup 2015). In these social media communities the use of microplastics in cosmetics is not seen as acceptable. Providing a forum for like-minded people has no doubt helped strengthen the campaign. A range of other marine litter apps are summarized in Table 6.7 that all include plastics, although this list is not comprehensive.

6.9.5 *Intrinsic and extrinsic motivation*

The motives for undertaking particular behaviours play an important role in whether the behaviour will be long lasting. Behaviours that are personally rewarding (thus have an intrinsic motive) are more likely to re-occur than those that are motivated for a reward or punishment (extrinsic motivation, De Young 1993). For example, charges on plastic bags or fines on littering (e.g. Section 6.6) can encourage an extrinsic motivation to avoid a financial punishment, and similarly deposit schemes that pay for rubbish and schemes that remove fees have the extrinsic motivation to gain a reward. These and similar schemes have been found to be effective, but the behaviour change will typically only last as long as the duration of the incentive. A further key question is what exactly is incentivized. For example, programmes that incentivize less quantity of rubbish in bins may have side effects such as illegal dumping or use of public bins. There is also a risk that by focusing on extrinsic motives, negative behaviours may be encouraged. For example, the fishing for litter schemes that pay commercial fishers to catch and land litter may increase the rubbish being thrown into the sea, as it becomes financially viable to litter then collect the rubbish (see Box 6.14).

Table 6.7 Examples of social media campaigns

Name	Aims	Includes micro-plastics explicitly?	Funding	Geographical scope	Functions
Clean swell http://www.beaconfire.com/blog/2015/05/engaging-environmentally-minded-volunteers-with-a-new-app/#.VkS4gLntlBc	Improve volunteer engagement by going mobile, streamline data collection, extend data gathering year-round, empower people to help keep the seas trash free.		Ocean Conservancy	Worldwide	Location tagging, summary of clean-ups, haptic feedback, social network integration, learn scientific facts, 16 litter categories
Beat the microbead http://www.beatthemicrobead.org/en/	To increase awareness of microbeads and pressure manufacturers into removing them from cosmetics	Yes	North Sea Foundation, Plastic Soup, UNEP and Fauna and Flora	Worldwide	Social media integration, enables users to search or scan barcode of a product and see whether microbeads are present.
Marine Litter Watch (MLW) http://www.eea.europa.eu/themes/coast_sea/marine-litterwatch/at-a-glance/european-citizens-to-help-tackle	Citizen science based app that aims to help fill data gaps in beach litter monitoring required by the Marine Strategy Framework Directive		European Environment Agency	EU + Iceland, Liechtenstein, Norway, Switzerland, and Turkey	Social media integration, location tagging, monitoring events following the MSFD framework through communities, clean-up events, supports approx. 200 items that fit into 9 main litter categories, users able to download data, top 10 items displayed within the app
Marine Debris Tracker https://outlook.office365.com/owa/?realm=students.plymouth.ac.uk - _msocom_1 http://www.marinedebris.engr.uga.edu/	To use innovative technologies and unique expertise to add culturally relevant outreach tools and information to the current NOAA Marine Debris Division, spread awareness and offer a simple tool for data collection.	Yes	NOAA, Southeast Atlantic Marine Debris Initiative	Worldwide	Enables users to log marine litter they have removed from beaches, streams, rivers etc. List of top contributors, location tagging, website integrates Twitter and lists recent debris activity, 7 main litter categories, users awarded 11 different badges in relation to how much debris they clear progressing from starfish to whale, enables users to flag objects that may be associated with Japan's tsunami, supports 8 tracking lists including ICC
Coastbuster http://www.ocean-networks.ca/learning/citizen-science/coastbuster	To aid reporting of large, unusual and potentially hazardous marine debris, especially items that may have been associated with Japan's March 2011 tsunami	No	Ocean Networks Canada	Canada (west coast)	World map, photo uploads, location tagging, link to Coastbusters Flickr stream, data linking to NOAA Marine Debris program and BC Ministry of Environment

Box 6.14 Voluntary vs. paid initiatives

The different fishing for litter schemes in Section 6.3 illustrate how two similar initiatives with different approaches can relate to intrinsic and extrinsic motivation. In Box 6.4, two approaches were described, one that paid fishers to collect marine litter in Korea and one that requires voluntary involvement by commercial fishers by providing the facilities and collection of waste for free in Belgium. During a study examining KIMO's Fishing For Litter scheme in Southwest England and Scotland, which adopts the voluntary approach (Wyles et al. in preparation), the risks and benefits of these approaches were discussed. It was evident that paying fishers could be seen to encourage the unintended attitudes and behaviour of only doing it for the money, whereas KIMO's approach focused on providing the facilities (linking to perceived behavioural control noted above) and thus raising awareness, interest and concern over the topic (linking to risk perception) in addition to removing the immediate cost of landing litter to the fishers (a less extreme financial incentive). This could be seen to address a more intrinsic motivation, potentially leading to a sustained behaviour change.

In contrast, initiatives that focus more on encouraging an intrinsic motivation to do the behaviour will more likely be long-lasting. These include awareness raising campaigns that aim to get individuals emotionally involved and discuss how they can be part of the solution (e.g. Section 6.5). When individuals volunteer to do pro-environmental activities, they are likely to feel good about themselves and have achieved something personally meaningful, which in turn increases the chance of repeating the behaviour (Asah and Blanhna 2013; Clary et al. 1998).

Halvorsen (2012) reviewed the effects of norms and policy incentives on recycling across ten OECD countries. The largest predictor of recycling efforts was the belief that it is beneficial for the environment, and to a lesser extent that it was a civic duty. Households with either weight- or unit-based pricing recycled significantly less than those without monetary incentives, with Halvorsen concluding that monetary incentives might crowd out morally motivated behaviour. Miafodzyeva (2013) provides a meta-analysis of 63 recent studies and found that moral norms, information and convenience were the most important predictors of recycling behaviour, followed by environmental concern. Sociodemographic predictors only made a "poor" contribution according to the authors, and pricing did not contribute. This evidence suggests that incentive policies need to be considered carefully and may not be the best approach to targeting behaviour related to marine litter, such as recycling behaviours here. This is linked to a broader movement within the behaviour change literature that suggests changing values and self-identity (associated with intrinsic motivations) is associated with more sustainable and long-term behaviour change than extrinsic motivations (e.g. Poortinga et al. 2013).

6.9.6 *Influence of demographic factors*

Why demography matters

Demography involves the study of the composition of populations. Human populations can be classified in many different ways, including in terms of ethnicity, religious background, social status/caste, degree of poverty or wealth, level of education, age structure, birth and death rates, and gender differences. Those demographic factors contributing to human well-being may be measured using individual descriptors or by using a collective indicator such as the Human Development Index (HDI).[16] Many aspects of human society are linked to where individuals fit into the demographic structure of their community. For example, those involved in the informal recycling industry in India or West Africa are often associated with particular demographic groups, based on age, gender, income and social status. They may be more exposed to risk as a result, including suffering significant human health consequences from handling plastics associated with electronic goods (UNEP 2016). Countries with a high HDI (e.g. OECD) tend to generate more waste per capita but have more effective waste management systems, with less leakage to the environment (Jambeck et al. 2015). Countries with low HDIs may generate less waste per capita but tend to have poorly developed waste management infrastructure, a lack of funding for improvements and less effective governance structures (UNEP 2016). In addition, there is a legal and illegal trade in waste from North America and western Europe to Asia and West Africa, as it is often cheaper to transport waste from a high-cost country to a lower-cost country, where education levels, governance, environmental standards and compliance may all be lower.[17]

It is important to include demographics when analysing the generation of microplastics, the sectors of society which are affected by microplastics, and when seeking to change behaviours and promote effective reduction measures for microplastics. This has been recognized by many individuals and groups seeking to raise awareness about issues though campaigns and educational initiatives.

Demographics and behaviour

Individual consumption of goods and services, personal habits (e.g. use of reusable bags and packaging) and waste practices (such as littering) are key drivers of marine litter. Consumer behaviour derives both from individual preferences and tastes, and from corporate strategies and marketing. Microbeads, for example, were introduced into consumer goods as a top-down corporate strategy, not in response to bottom-up consumer demand.

[16] http://hdr.undp.org/en/content/human-development-index-hdi

[17] Basel Convention on the Control of Transboundary Movements of Hazardous Waste and their Disposal; http://www.basel.int/

Little is known about the demographic factors influencing perceptions and behaviours of relevance to marine litter, but it seems to assume there will be effects in particular circumstances. For example, a recent study in the USA on the purchase of bottled water indicated that age and income were stronger predictors of consumption than gender. In some countries it is the unavailability of safe potable water that drives the demand for bottled water, irrespective of other factors. Littering behaviours are demographically variable, although cross-national comparisons have not been made and it is not clear to what extent gender is relevant (KAB, 2009; Lyndhurst 2013; Curnow and Spehr 2005). Clearly, sustained and comparative research is needed to understand the demographic drivers of such behaviours, and thus the possible levers for change. Further research into the demographics of consumer behaviour specific to marine plastic pollution, and willingness to change those behaviours, is urgently needed. To extend on the brief overview of demographic differences outlined in the first report (GESAMP 2015), potential gender-based aspects are described more fully as an illustrative example.

Gender-based aspects

Gender is one of several key factors to consider when assessing the societal response to microplastics. However, its importance may be hidden if social categories in an environmental assessment are not differentiated by gender. The influence of gender on the frame of reference for environmental inquiry can be demonstrated using a general model of environmental gender analysis (Table 6.8). This approach could be adapted to take account of other societal characteristics.

Table 6.8 UNEP model of integrated environmental assessment: modification of the foundational questions using gender as an example (based on Seager 2014)

Foundational questions in the UNEP model of integrated environmental assessment[18]	Gender-sensitive version
1. What is happening to the environment and why?	1. What social forces are producing the changes we see in the environment and why? Are those social forces 'gendered'?
2. What are the consequences for the environment and humanity?	2. What are the ecological changes produced, and what are the consequences for social systems and human security? In what ways are those consequences gender-differentiated? What are the larger social consequences of gender-differentiated impacts?
3. What is being done and how effective is it?	3. Who are the actors involved in responding (at many levels) and are men and women equally engaged? Equally effectively engaged? Are there gender differences in weighing what 'should' be done and in weighing the effectiveness of possible actions and solutions?
4. Where are we heading?	4. Where are we heading and will there be different outcomes for women and men? Are there gender-differentiated perceptions of where we're heading?
5. What actions could be taken for a more sustainable future?	5. What actions could be taken for a more sustainable future that will position men and women as equal agents in taking such actions? What socio-economic factors will shape different outcomes and responses for men and women?

[18] http://www.unep.org/ieacp/iea/

The extent to which gender per se is the main factor in influencing an outcome will depend on other individual, situational and demographic factors, and these are likely to vary widely on a variety of spatial and temporal scales. For example, an increase in relative wealth or educational attainment may alter the relative importance of other demographic factors for individuals or communities.

Gender and fisheries, an example

Commercial fisheries and aquaculture are key economic activities in many coastal regions, and artisanal fishing (i.e. traditional, small-scale) may be vital for food security. It is a sector that both generates and is impacted by marine plastics. Many roles in this sector are differentiated by gender. Women participate throughout most parts of the fishing cycle; including post-capture processing, inland-waters and onshore aquaculture, net-mending, processing, and selling. Women fish in the coastal zones, inshore reefs and mangroves, they glean at low tide, and cultivate fish fry in the shallows (Lambeth et al. 2014; FAO 2015), but very few participate in open-sea capture fishing. Open-sea, commercial and large-boat fishing is almost entirely a male domain. In many cultures around the world there are taboos, prohibitions, superstitions and cultural norms about femininity that keep women off the boats and on the shore. This renders women's fishing contributions largely invisible – it is left out of most data collection efforts, as well as overlooked in conventional government or aid programmes that support fishing and fishers (Siason et al. 2010). If there are to be remediation programmes, financing to cope or reduce plastics pollution, or education programmes about plastics, a concerted effort to make these gender-inclusive will be essential.

Because of these spatial differences in women's and men's fishing, there may be significant gender differences in the experience of, knowledge of, and impacts of plastics pollution. Debris build-up in littoral and coastal zones can be severe and is different in character than open-sea plastics pollution, as analyses discussed elsewhere in this report demonstrate. This will have a different impact on women's fishing activities in the near-shore zone than on men's fishing in open oceans. Consequently, loss of economic activity, damage to well-being, and mental health aspects of impacts from degraded environments (see Table 6.10) may be associated with gender, in particular circumstances.

6.9.7 *Conclusion*

In order to make the most of the initiatives and schemes that aim to help tackle microplastics in the marine environment, it is necessary to consider individual and demographic factors. We briefly reviewed a subset of factors known to influence behaviour and behaviour change.

Understanding people's perceptions of the risks associated with microplastics is important, as this can result in direct impacts on different sectors (e.g. to the fishing industry if people start to avoid fish fearing they may be at risk of taking in microplastics) and on the issue at hand (e.g. if they do not perceive microplastics as a problem, individuals will be less willing to adopt behaviours to address the problem). Whilst scientifically little is currently known on the public's perceptions of microplastics, a simple media analysis shows that there is a sudden rise in interest in microplastics in the UK, so we can expect risk perceptions to become more prominent. More research is urgently needed, ideally with systematic methodologies, such as the mental models approach. Similarly, with nanoplastics as an emerging issue, social research exists on perceptions of nanotechnologies more broadly, which could be examined in greater detail.

As well as individuals' perception of the problem and associated risks, people's perceived responsibility, behavioural control and social norms all play an important role on behavior. Similarly, these perceptions and behaviours can also vary with demographic positionalities, e.g. health effects, personal and community disruption caused by environmental degradation, and understanding the pathways to solutions, may vary with age, class, and gender, among other demographic variables. Whereas different sectors are seen to vary in responsibility, all sectors seem to accept that it is everyone's duty to address marine litter. Thus, it is necessary to collaborate on solutions. Emphasizing what ought to be done and is being done can be a powerful tool in promoting positive behaviour change. Understanding what motivates a particular behaviour is necessary to sustain good acts, as intrinsic motivations (such as benefit for the environment, feelings of moral duty) are key predictors of recycling behaviour likely to last long-term in contrast to extrinsic motives (such as incentives or fines) that are often short-lived and can have adverse effects on behaviour.

To paint a fuller portrait of the social dimensions of plastics pollution and to map potential transformative pathways towards solutions, serious, cross-national and sustained research – as well as fundamental data collection – is needed. For example, considerable work is needed on the gender dimensions of marine-based livelihoods, on health and environmental impacts of environmental change in these environments, and on risk perceptions.

6.10 Collaborating on solutions

6.10.1 *Conclusions on actor roles – multi-level governance*

Since marine litter occurs on a global scale and knows no geographic boundaries, but also has impacts down to the local level, action is needed by many different stakeholders. The capacity and responsibility to address marine litter is spread across a range of stakeholders and innovative collaborations are needed. Engagement is necessary by those who are responsible for and those impacted by marine litter if the problem is to be addressed effectively:

- Need for international collaboration to reach solutions. International collaboration such as the Global Partnership on Marine Litter (GPML), the UNEA Resolution on marine plastic debris and microplastics, and the June 2015 commitment by the G7 group of nations can help highlight the problem and catalyse solutions;
- National governments can invest in infrastructure, set incentives, legislate, inspect and enforce, support research and development (R&D), and encourage greater producer and consumer responsibility;
- Municipalities/local governments can invest further in waste, wastewater treatment and port reception infrastructure that can help prevent marine litter;
- Private sector can invest in innovative product design (e.g. improved durability, recyclability and green chemistry) and embrace producer responsibility more widely. In addition, industries should reduce as much as possible the loss and disposal of products at sea;
- NGOs and voluntary organizations can motivate changes in consumer habits (e.g. the Beat the Microbead campaign/app) and norms and encourage producer responsibility;
- Local communities can engage in awareness-raising and clean-up activities, as well as participating in small- or larger-scale projects to generate value from collected marine litter;
- Consumers and individuals, including tourists, can make responsible choices regarding purchases and take responsible actions regarding waste disposal; and
- Academia should prioritize research on improving understanding of the impacts of marine litter, designing optimum communi-

cations and behaviour change interventions, the costs of action and inaction and governance solutions to marine litter.

6.10.2 *Tools and opportunities*

There are a range of tools and opportunities for addressing macro and microplastic throughout the supply chain – see Figure 6.1 for an illustration of the supply chain and key sectors. Given the longevity of plastic and its value as a resource, some argue that an overarching circular economy approach should be adopted, encouraging greater reuse, repair, remanufacture and recycling, so as to minimize the risk of marine plastic waste, and keep as much of plastics' value in the economy.

There is a wide range of solutions to address marine litter, from upstream prevention to downstream clean-up. These broadly follow a hierarchy for marine litter management that builds on the concept of the waste management hierarchy, which is widely accepted in waste management policy and legislation. The typical waste hierarchy prioritizes prevention as the preferred method of waste management, followed by reuse, material recycling, energy recovery and disposal. Figure 6.3 uses this order and applies it to marine litter to create a suggested ideal hierarchy for the management of marine litter.

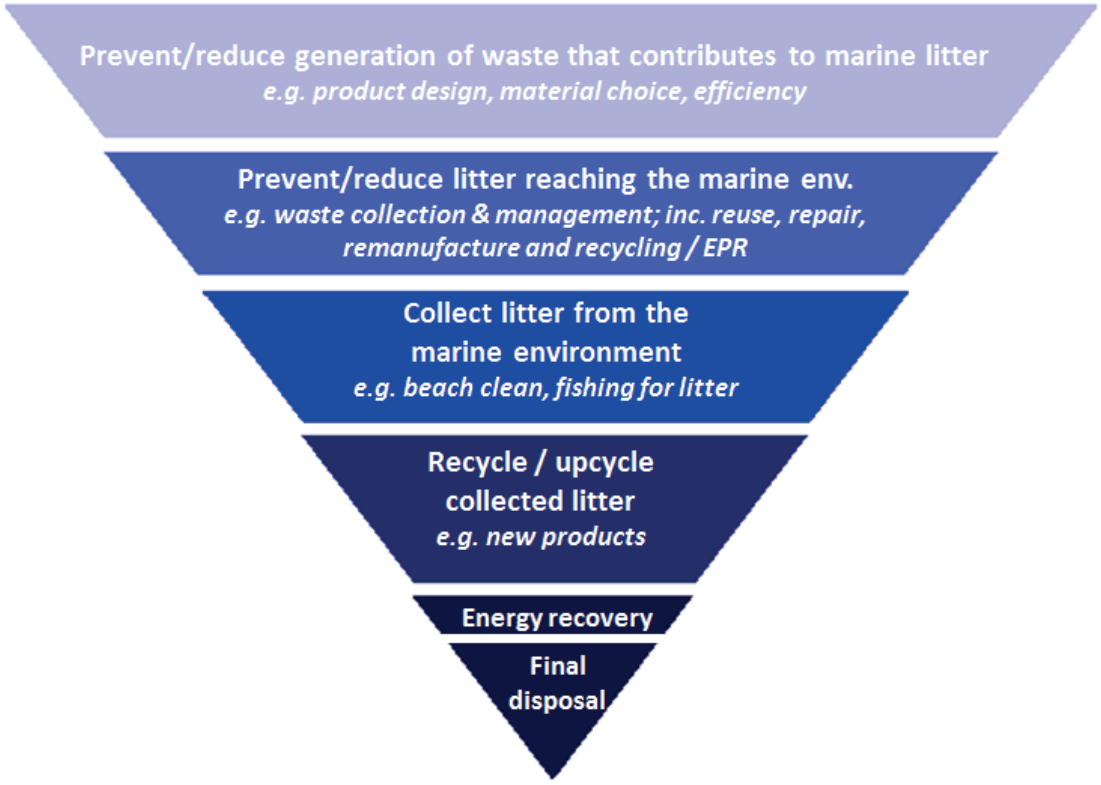


Figure 6.3 *A hierarchy for marine litter management (Source: Emma Watkins, IEEP)*

Within this hierarchy, various actions to address marine litter can be grouped as follows:

Preventing/reducing waste that contributes to marine litter:

- Product design changes and extended producer responsibility can help to prevent the generation of marine litter and avoid certain impacts which are more difficult to address after litter has been created.
- Target group-specific awareness raising, including consumers to reduce the generation of waste (e.g. reusable items for everyday use or identifying microbeads in personal care products, economic incentives such as discounts for reusable packaging).

Preventing/reducing litter reaching the marine environment:

- Invest in new and improved waste management infrastructure, including to avoid waste being blown from landfills (e.g. perimeter netting), riverine, port and beach infrastructures (e.g. litter traps, booms and bins).
- Economic incentives such as deposit refund schemes and plastic bag charges can help influence consumer choice and/or encourage different habits (e.g. return bottles; choose multi-use bags) which can reduce the incidence of marine litter. Similarly littering fees and fines for illegal disposal of waste can be useful incentive tools.
- Bans (e.g. plastic bag bans, smoking bans on beaches, bans on plastic blasting in shipyards) can provide a cost-effective solution. However, feasibility will depend on various

factors including the availability of viable substitutes, political considerations etc.

- Carefully designed campaigns and awareness activities can help avoid the generation of marine litter through improved habits and social norms and can spur changes in product design. This is an essential upstream preventative measure.

Collecting litter from the marine environment:

- Fishing for litter can be a useful option in the hierarchy of means to address marine litter (although this can only address certain types of marine litter) and can be combined with economic incentives to encourage action.
- Marine litter clean-ups that engage volunteers in clean-up activities can help reduce costs (although the time of volunteers also has an economic value) and improve awareness, which in turn can be an upstream action – by raising awareness, individuals are less likely to litter.

6.11 Conclusions, knowledge gaps and priorities

6.11.1 *Conclusions*

While we have a fair and growing understanding of the costs of macroplastic litter, at present we lack an adequate understanding of the costs of microplastic litter. Similarly, there is little information on the cost-effectiveness of actions related to microplastics. Given the physical links between macro and microplastics, the motivations for action on marine plastic cannot be fully separated in terms of macro and microplastics. The argument for keeping the value of plastic within an increasingly circular economy, and avoiding environmental, health and social and economic burdens associated with marine litter, is clear when looking at the macro scale. It is less clear if looking only at the microplastic scale, given information gaps. This underlines both the importance of research into impacts and costs, and the importance of looking at both macro- and microplastics when considering costs, policy responses and specific actions.

At present we lack an adequate understanding of the costs of marine litter, the cost-effectiveness of actions, motivations for action and hence the potential of measures to tackle marine litter. This weakens the argument for keeping the value of plastic within an increasingly circular economy, and avoiding environmental, health and social and economic burdens associated with plastic become marine litter.

6.11.2 *Knowledge gaps*

Economics

- The core knowledge gap as regards economics is the difficulty in estimating the costs of impacts of microplastics with the current level of (lack of) knowledge on the health and ecosystem impacts of microplastics. A fair amount is known of macroplastics and the benefits of acting on macroplastics include both avoided impacts of macroplastics and microplastics (as the macro breaks down into micro). The economics of the two cannot be separated easily.
- It is therefore critical to address the scientific knowledge gaps on microplastic impacts to be able to build a nuanced and robust economic analysis for microplastics. The economics of macroplastics are already well enough understood (though of course data gaps remain) to warrant action on macro plastics that will also have an impact on microplastics. A key area of impacts will be linked to how public perception changes with knowledge related to the level of ingestion of microplastics in fish as this can in principle be expected to lead to a lower demand and lower price. Research into the likely price and demand effects related to perceptions of quality will be important.

Social aspects

- There is a lack of information about level of knowledge that the general public, the many sectors involved, international bodies and policymakers have about microplastics.
- There is a lack of knowledge about the level of understanding of risk in general by the public, and microplastics in particular.

6.11.3 *Research priorities*

Research priorities are noted below for: science, economics and social aspects in turn. In each case, research needs related to understanding the problem and related to the solutions are included.

Science related priorities are:

- Understanding microplastic impacts on human health via fish/shellfish ingestion, how social perceptions respond to uncertainty/knowledge, and how these risks translate into consumer demand and thus economic impacts;
- An understanding of the perceived and actual nature and extent of ecosystem impacts of smaller marine debris (e.g. nanoparticles and microfibres) and leachates/uptake from all debris (e.g. chemicals that can be endocrine disruptors); and
- Understanding the nature and scale of plastic footprints – of a person, a product, company, sector, of a nation.

Economics of action and inaction

Research to understand the problem

- Improve understanding of the costs of inaction and how it relates to costs of action to underline where early action is particularly important, beneficial or effective. This could be done at the macro, sector, product and type of marine litter scales to give different evidence base for different decision frameworks and governance processes;
- Improve understanding by policy-makers of the cost of action and the benefits of action to highlight cost-effective solutions; and
- Determine the value of plastics (cost, benefit) to help underline the potential benefits of circular economy activities and the economic inefficiencies of letting plastic become waste – this needs to be done for plastics as a whole to be able to understand measures that affect both macro and microplastics.

Research on measures/solutions

Many of the measures that will affect the level of microplastics in the seas will focus on addressing macroplastic marine litter. Research priorities include:

- Research into the likely elasticity of demand for: a) plastic products – i.e. how is demand likely to change with price (e.g. for plastic bottles, plastic bags); and b) fish – i.e. how is demand likely to change with perception of quality and potential health impacts;
- Explore the economics of recycling for plastic waste – values of recycling of waste before it becomes marine litter, the values of different plastic types that have become marine litter and hence incentives for recycling;
- Additional information on the costs of litter prevention and clean-up activities;
- Assessment of the long-term effectiveness and cost-efficiency of existing actions and initiatives on microplastics, to provide information to support and justify action;
- Information on the costs of action that have been, or could be, taken by producers (e.g. more environmentally-friendly design, participation in extended producer responsibility and/or voluntary initiatives);
- Information on the costs of inaction and action taken by the fisheries and aquaculture sector, for example through ecological studies and surveys of fisherman to identify economic losses; and
- Further data on marine-litter related costs to the shipping sector, and the effectiveness and cost-efficiency of actions taken by the sector.

Perceptions and behaviour

Related to the problem

- Research consumer perception about plastic in seafood – i.e. how they would likely react to knowledge of plastic levels in their food and health risks – and demographic differences, including gender, in these perceptions;
- Further research on the impacts (including demographic factors) of marine litter on resident and visitor beach choice, as well as scrutiny on changes in tourism revenues and how these might be linked to marine litter;
- Study the difference in public perception and established science on impacts of marine debris; and
- Research into why many people do not take responsibility for their waste and what motivates others who do take responsibility.

Related to solutions/measures

- Greater understanding of different stakeholders' (especially consumers') perceptions of the issue and risks surrounding microplastics in order to take appropriate action;
- Research the effectiveness of citizen-science campaigns;
- Understand what would drive behaviour change away from single-use plastic;
- Research the most effective messaging to encourage responsible use; and
- Study how media campaigns cover risk and actions on marine debris and how to make better and more effective campaigns.

Economic measures

This additional research, if successful, would allow a more comprehensive understanding, through an improved evidence base, of the costs of marine litter, the cost-effectiveness of actions, motivations for action and hence the potential of measures to tackle marine litter. This will keep the value of plastic within an increasingly circular economy and avoid environmental, health and social and economic burdens associated with plastic becoming marine litter.

7 METHOD DEVELOPMENT AND HARMONIZATION

Key points

1. A number of factors may affect the representativeness of data on microplastics, including spatial and temporal variability, types of particles, proximity to rivers, variety of approaches, sampling methods, size limits, extraction methods, characterization and reporting units.
2. In many cases, environmental levels of microplastics may be difficult to interpret due to the lack of consistency in the assays used and technical challenges.
3. As sampling, extraction, detection methods and techniques are developed worldwide, a harmonization and standardization of techniques and protocols is urgently needed to better assess risk in a reproducible manner.
4. Further research on methods needs to consider sampling design and analytical methods capable of characterizing and quantifying small sized particles, e.g. 20 to 30 µm and nano-sized particles.

7.1 Lessons from the first assessment

Many decisions are made in the process of designing and implementing sampling plans that can affect the accuracy, reliability and representativeness of the results. The first GESAMP report on microplastics discusses the diversity of methods used to extract, quantify and characterize microplastics from environmental matrices. The analysis of environmental samples is a multi-step process that includes sample preparation, extraction of microplastics, further purification ('clean-up'), detection and quantification of particles, and identification of polymer types. The significant heterogeneity in the distribution of microplastics at sea and in sediments or beaches, emphasizes the need to harmonize sampling methodologies. Still, it can be difficult to choose a "best practice" when, for example, mass may be useful from an overall waste management perspective and number of particles may be of greater significance ecologically.

Whether sampling at the sea surface, on the seabed, in the intertidal or in biota, it is important to note that today a variety of methods have become available. At sea, towed nets with variable 330 µm net mesh, variable net aperture and net length are commonly utilized to filter large volumes of water in situ. Sampling sediments can require significantly more effort and resources, with finer-grained sediments usually requiring more elaborate, laboratory-based separation techniques. In biota, microplastics are measured in several species of fish, bivalves, crustaceans and birds, with greatest focus on stomach content analysis.

In addition to differences in matrices, the diversity of plastic material has created methodological challenges, especially for targeted, quantitative analyses of microplastics. Most studies have focused on large microplastics (1–5 mm), including pre-production resin pellets, which are visible to the naked eye and can be picked out. However, when smaller particles are targeted for analysis, they are harder to identify. Initial separation is a necessary step but it becomes increasingly difficult to distinguish plastic from non-plastic particles with decreasing size. Raman and/or FTIR spectroscopy are then required to confirm the identification of plastics, and their synthetic polymer for particles.

This chapter will discuss the many methods that are used today with hopes to help facilitate harmonization of methods in the future.

7.2 Introduction

Microplastics comprise a heterogeneous assemblage of plastic particles that vary in size, shape, colour, specific density, chemical composition and other characteristics. Most studies focus on quantifying their abundance in the marine environment and have applied a wide variety of methods for detecting, identifying and quantifying the contamination in many different types of aquatic habitats.

Individual analyses are complicated by the spatial and temporal variability of microplastics and the types of matrices they are found in. Many surveys focus on open waters, shorelines and more recently estuaries (reviews in Moore 2008; Hidalgo-Ruz et al. 2012; Cole et al. 2011; Van Cauwenberghe et al. 2013; Lusher et al. 2015; Rocha-Santos and Duarte 2015). They measure microplastics in surface waters, sediments, animals and even sea ice. A number of factors may modify the representativeness of data, including the proximity to a source of microplastic such as a large river system (Moore et al. 2002) and/or the size, shape and type of particles that are included in the analyses (e.g. plastic fibres; Browne et al. 2010).

Datasets are further complicated by the wide variety of methodological approaches that are applied by different researchers to extract, identify, quantify and characterize microplastics. This makes comparison of reported microplastics difficult among studies without additional calculations based on assumptions (e.g. volume calculation, sediment densities, etc.). The majority of these method inconsistencies can be related to: (i) differences in the lower and upper size limit examined; (ii) the sensitivity of the applied extraction technique; and, (iii) differences in sampling technique, all leading to a wide variety of efficiencies and reporting units (Lusher et al. 2015; Van Cauwenberghe et al. 2013).

For some time, the majority of sampling and extraction techniques were similar. Studies often relied on volume reduction and visual or density separation. But, as the

field has become more complex, a large assortment of variations has been developed. The large diversity in techniques applied for extraction, detection and quantification of microplastics primarily derived from: i) the need to investigate and/or monitor a variety of matrices (ice, water, sediment, animal gut content, whole animals, etc); and, ii) the fact that measuring abundances of different sizes of microplastics (e.g. <1 mm or 1 to 5 mm) require different methods (Figure 7.1). Recent reviews and critiques of the field have called for an improvement in the methods to yield more comparable, precise and accurate results (Rocha-Santos and Duarte 2015). In order to achieve this goal, several studies have been carried out to analyse method development and/or comparison for sampling (Norén 2007, 2011; Song et al. 2014), separation (Imhof 2012; Claessens et al. 2013), identification (Vianello, Boldrin et al. 2013) and clean-up (Claessens et al. 2013).

This chapter discusses sampling, extraction and analytical methods to characterize quantities, types, sizes and chemical properties of microplastics. These methods all have a given degree of specificity in what is targeted which depends on how the microplastics are extracted from the environmental matrix, such as seawater, sediment and biota.

7.3 Sampling and observations

The observed variations in environmental samples are largely due to many factors, including a large diversity in the type and size of particles, the locations examined (e.g. proximity to sources), the sample matrix, the patchy distribution of microplastics and sampling conditions (e.g. weather conditions that affect sea-state). Sampling microplastics in the marine environment requires different approaches for different matrices (sea surface, water column, sediment, organisms). Defining a consistent sampling strategy for many of the different sample matrices (i.e. sediment, water, biota) is of high importance to achieve robust and comparable datasets. Statistical methods are also important in the development of monitoring protocols for harmonization, which will improve our ability to assess the risk of contamination.

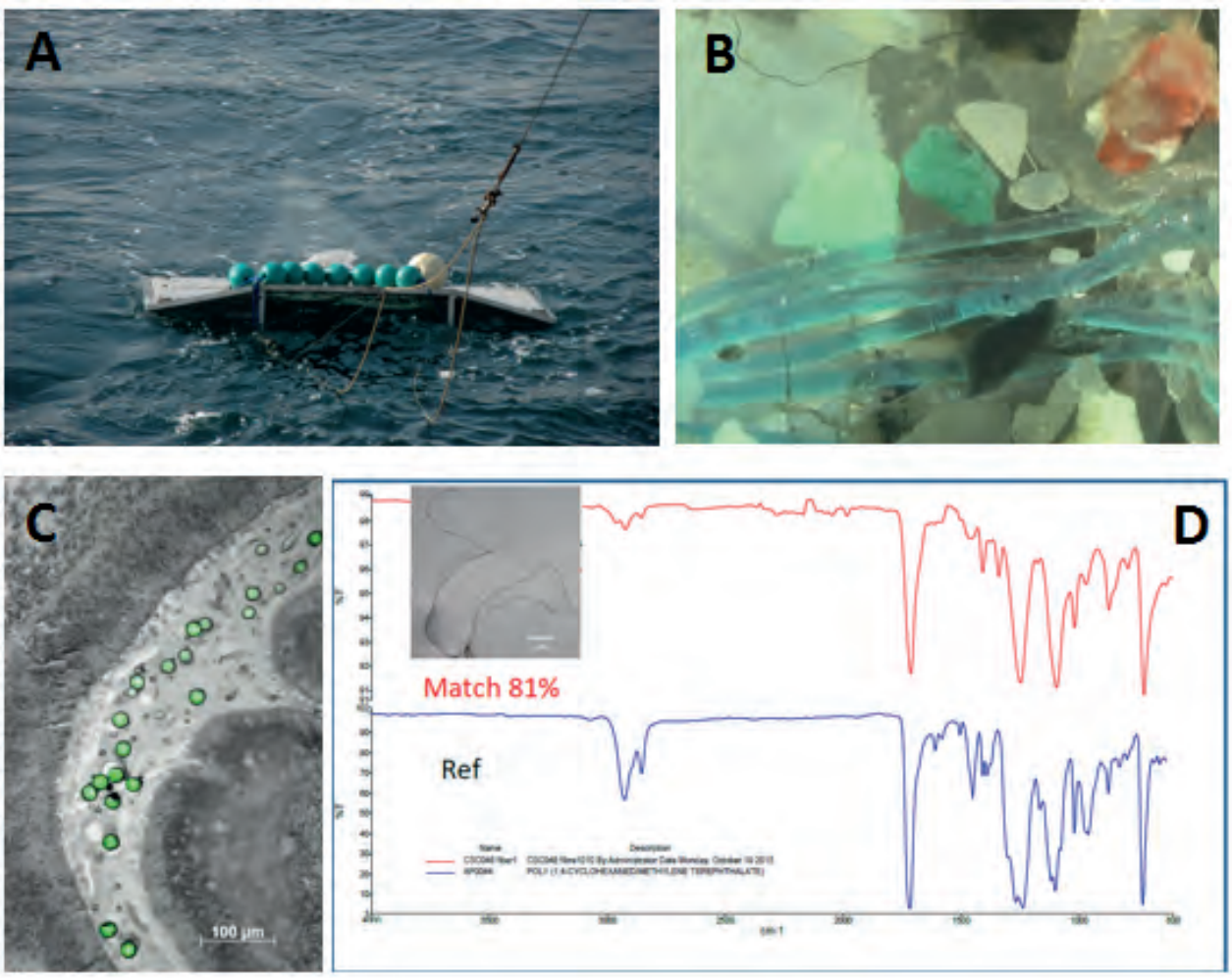


Figure 7.1 *Examples of sampling methods for sea surface and sediments or beaches (A: surface "Manta" trawl for sampling microplastics at the sea surface, credit F. Galgani), standard protocol for visual observation (B: microplastic assemblage as observed by microscope, Credit J.H. Hecq & F. Galgani), ecologically relevant laboratory experiments (C: experimental ingestion by Mussels of dye-coated microparticules, credit A. Huvet) and reliable and fast characterization of all types of plastic microparticles (D: Raman spectral analysis of fibres for identification and characterization of plastics type, credit P. Sobral)*

7.3.1 *Microplastics at sea*

In seawater, surface layers are generally sampled, since many of the most mass-produced polymers (e.g. polyethylene and polypropylene) initially are buoyant and accumulate at the surface. Density values range from 0.85 to 0.94 g cm^{-3} for polypropylene and from 0.92 to 0.97 g cm^{-3} for polyethylene (Leslie 2011). Many other polymers (e.g. PVC, polycarbonates) are denser and likely to sink (Hidalgo-Ruz et al. 2012). As a consequence of sampling methods, denser microplastics may be underrepresented in open ocean datasets (Woodall et al. 2014). Even for buoyant polymers, the condition of the sea affects suspension of microplastic particles and can skew results. During rough weather, researchers have found that buoyant plastics are mixed below the surface, causing an underestimate in the quantity of microplastics. Correction factors during strong winds have been developed and can be applied for sampling surface layers at sea during rough weather (Kukulka et al. 2012). A complicating factor is that many plastics designed for more durable applications can contain a wide variety of additive chemicals that can alter the initial polymer density.

The majority of sampling methods used to examine the spatial distribution, abundance, mass, type, and/or size of microplastics in seawater are based on volume-reduced samples, i.e. filtering large volumes of seawater with nets, and preserving only portions of interest for further processing. Surface water sampling techniques mainly include manta trawls and neuston nets that sampled the top 10 cm of water. Few have used bongo nets and opening-closing nets for mid-water. Mesh sizes of the nets range from 0.053 to 3 mm, with a majority of the studies using 330 µm aperture mesh. Units commonly used for abundance estimates are number of particles per km^2, m^2 or m^3, using flow metres to estimate the volume of water sampled.

The size ranges of microplastics obtained from bulk seawater are limited by the pore size of the mesh on the net (Ng and Obbard 2006). Net apertures, or the size of the mouth of the net, vary from 0.03 to 2 m^2, depending on the type and shape of the net. Of course, there are limitations and benefits to different nets. While smaller mesh sizes increase net resistance and clogging, the length, area or aspect ratio of the net may also vary, enabling higher-speed tows in some cases. Sampling techniques using 80µm mesh nets with a deeper vertical sampling range have been proposed to quantify microplastic in the smaller size range and below the surface (Norén 2011; Dris et al. 2015). Another method for sampling microplastics from seawater is using long-term data from Continuous Plankton Recorders (CPRs) using a 280 µm mesh on regular and fixed routes. This is now considered a routine part of on-going CPR analysis (Cole et al. 2011) and has the advantage of allowing retrospective evaluation of microplastic abundance in archived samples. The method is restricted to sub surface (down to 10 m) collection. Overall, it is important to consider the range of methods that can be used for sampling microplastics from seawater. Using mesh with different apertures can cause large variations in the quantity of microplastics collected, and it is important to note that the common method currently used, the manta trawl, does not capture smaller sized particles.

After sampling in the field, further processing is needed to sieve and separate particles (i.e. extraction and analysis). Mesh sizes used in the laboratory often range from 38µm to 5 mm and often include 330 µm, 1 mm and 2 mm. To avoid degradation, plastics separated from the sample have been dried and kept in the dark. This step is probably unnecessary if samples are examined within a few months of collection. When necessary, digestion methods are used to clean up the organic matter (see Chapter 7.3.3). In other cases, density separation in NaCl(aq) is used to isolate the plastic debris through flotation. Last, the samples may be counted directly and/or weighed to calculate the mass of the sample. To calculate mass, the sieved material must be dried.

7.3.2 *Microplastics in sediments*

A wide range of sampling techniques are used for monitoring microplastics in sediments (reviewed in Hidalgo-Ruz et al. 2012; Van Cauwenberghe et al. 2013 and Rocha-Santos and Duarte 2015). These methods include density separation, filtration and/or sieving (Hidalgo-Ruz et al. 2012; Rocha-Santos and Duarte 2015). To facilitate the extraction of microplastics from organic components, such as organic debris (shell fragments, small organisms, algae or sea grasses, etc.) and other items such as pieces of tar, other methods can be applied, such as enzymatic, carbon tetrachloride (CCL_4) or Hydrogen peroxide (H_2O_2) digestion of organic materials (Galgani et al. 2011; Hidalgo-Ruz et al. 2012; Cole et al. 2014).

The most common approach is to extract plastic particles from sediment using a density separation based on the difference in density between plastic and sediment particles. Typically, this is achieved by agitating the sediment sample in concentrated sodium chloride (NaCl) solution. However, as the density of the NaCl solution is only 1.2 g cm^{-3}, only low-density plastics float to the surface and are extracted. Different authors have addressed this issue by using different salt solutions such as 1.4 g cm^{-3} polytungstate (Corcoran et al. 2009), using zinc chloride (ZnCl2, 1.5-1.7 g cm^{-3}; Imhof et al. 2012 or sodium iodide (NaI, 1.6 -1.8 g cm^{-3}; Dekiff et al. 2014; Van Cauwenberghe et al. 2013) to obtain higher densities. These modifications result in an increased extraction efficiency for high-density microplastics such as polyvinylchloride (PVC, density 1.14 to 1.56 g cm^{-3}) or polyethylene terephthalate (PET, density 1.32 to 1.41 g cm^{-3}). As these high-density plastics make up over 17% of the global plastic demand (PlasticsEurope 2013), not including these types of microplastics can result in a considerable underestimation of microplastic abundances in sediments, especially as these high-density plastics have a negative buoyancy and thus are much more likely to sink.

The choice of sampling strategy and sampling approach (reviewed by Hidalgo-Ruz et al. 2012) will eventually determine the unit in which observed abundances will be reported. While a simple conversion can sometimes be made to compare among studies (Lusher et al. 2015), comparison is often impossible or requires assumptions that lead to biased results. Studies sampling an area (using quadrants) will often report abundances per unit of surface (m^{-2}; e.g. Martins and Sobral 2011). If bulk samples from the surface to a specific depth are

taken, the reporting unit is m^3 (e.g. Turra et al. 2014). Conversion between these types of abundances is possible, if sufficient information is available on sampling depth. Yet, for 20% of the studies this is not the case as reported sampling depths can range from 0 to 50 cm. Other widely used reporting units for sediment samples are volume (mL to L; e.g. Noren, 2007) or weight (g to kg; e.g. Claessens et al. 2011; Ng and Obbard 2006). Conversion between these two types of units is not straightforward. Detailed information on the density of the sediment is required. As this is never (as far as we could establish) reported in microplastic studies, assumptions have to be made (e.g. Claessens et al. 2011). Additionally, within studies reporting weight, a distinction must be made among those reporting wet (sediment) weight and those reporting dry weight. This adds to the constraints of converting from weight to volume units, or vice versa. Sediment samples from different locations or even different zones on one beach have different water content. Therefore, some authors choose to express microplastic abundance per sediment as dry weight to eliminate this variable (Claessens et al. 2013; Dekiff et al. 2014; Ng and Obbard 2006; Van Cauwenberghe et al. 2013; Vianello et al. 2013).

7.3.3 *Microplastics in biological samples*

Biological sampling that involves the examination and characterization of microplastics consumed by marine organisms has been used for vertebrates (e.g. Lusher et al. 2013; Choy and Drazen 2013; Avio et al. 2015), invertebrates (e.g. Browne et al. 2008; Murray and Cowie 2011; Desforges et al. 2015; Van Cauwenberghe et al. 2015) and birds (e.g. van Franeker et al. 2011). In general, the research question addressed will greatly influence which sampling and extraction technique to use. For example, the size range of microplastics overlaps the size range of micro- and macroplankton, highlighting the potential for microplastic ingestion by a wide variety of organisms (Hidalgo-Ruz et al. 2012). Thus, the sampling scale and methodology will depend on the size of the particle and the size of the studied organisms. Harmonization of sampling and extraction techniques should be adopted for monitoring purposes.

Gut contents can be analysed for the presence of microplastics, which can then be identified and quantified. This approach has now become one of the ecological quality assessment markers used by OSPAR to assess both the abundance of microplastic debris at sea and regional differences and trends over time (van Franeker et al. 2011). In terms of monitoring and with regard to in situ experiments, one of the most important aspects is the choice of target species. It is important to consider (i) the exposure to plastics, especially for the species that are living at the surface or in the sediments, (ii) the ingestion rate, especially for filter feeders such as bivalves, (iii) the significance of results, which will vary depending on whether environmental impact or human health is considered, (iv) the biological sensitivity of certain species, such as the high retention rate in birds of the procellariform family, and finally (v) a large distribution and easy sampling of the target species.

The methodological difficulties for isolating particles from biota partly explain why only a few studies specifically addressed the occurrence of microplastics in marine organisms. The extraction and quantification of microplastics from organisms is especially challenging because the plastic pieces may be masked within biological material and tissues. Protocols have been proposed on the extraction of microplastics from marine invertebrates after a pre-digestion of organic matter (Claessens et al. 2013), indicating the importance of solvent properties and pH for sample treatment, affecting both the estimation and the characterization of the polymers by FT-IR. The enzymatic digestion of organic matter with proteinase k is a reliable method to extract microplastics from plankton samples (Cole et al. 2014), but at higher costs when considering large-scale field sampling and monitoring.

More recently, Avio et al. (2015) optimized a new protocol allowing an extraction yield of microplastics from fish tissues ranging between 78% and 98%, depending on the polymer size. This protocol integrates previously used extraction methods with slight modifications. Each sample was added to 250 ml NaCl hypersaline solution (1.2 g cm^{-3}), stirred and decanted for 10 min; the filtration step was carried out twice in order to obtain a better extraction performance. The membranes with retained materials were then transferred in a petri dish with a 15% H_2O_2 solution for the partial digestion of residual organic matter and allowed to dry in an oven (50°C, overnight), before observation using a microscope. FT-IR spectra for particles analysed before and after the new extraction procedure were comparable, with a similarity of approximately 93% for polyethylene profiles, and greater than 87% for polystyrene.

7.4 Detection and analytical techniques

Visual examination is the most common method used to assess size and quantities of microplastics, although it can have a relatively high error rate (Loder et al. 2015). Various imaging approaches, such as zooscan (Gilfillan et al. 2009) or semi-automated methods (flow/cytometer, cell sorter, coulter counters) may be practical for the visualization or counting of microplastic particles, with the potential to enable large numbers of samples to be analysed rapidly. For a better identification of plastics, specific criteria can be applied, such as the presence of cellular or organic structures, the constant thickness of fragments or fibres, homogeneous colours and plastic brightness. However, the reliability of such approaches has not been evaluated. Other analyses based on visual examination with light, polarized or not, or electron microscopy may provide higher resolution but cannot be used to determine polymer type.

Different characteristics of microplastics may indicate possible sources (e.g. from type). It is thus important to use methods that identify the type (pellets, filaments, plastic films, foamed plastic, granules, extruded polystyrene foam), shape (cylindrical, disks, flat, ovoid, spheroids etc.), condition (degraded, rough, eroded, broken, presence of fractures) and colour (opaque, clear, pigmented, etc.).

A detailed analysis of microplastics in various environmental samples requires the identification of chemical compounds and polymers. Such methods can confirm that a material is actually plastic. The identity of small pieces of debris is usually confirmed by an additional step, such as Fourier transform infrared (FT-IR) or Raman spectroscopy. FTIR compares the Infra Red spectrum of a sample with spectra of known polymers. Infrared spectrophotometry and near-infrared spectrometry enable the identification of common polymers including PP, PE, and polyester. Raman spectroscopy gives information about the crystalline structure of the polymer and may be combined with imaging techniques to identify microplastics in the µm range, and to perform polymer analysis at multiple points on the surface of a sample (Leslie 2011). Other methods such as differential scanning calorimetry, smoke characterization after combustion, calculation of specific density, attenuated total reflectance (ATR) FT-IR or "deep Raman" spectroscopy and colour have also been considered, with compromises between simplicity and precision. Other analytical techniques, such as pyrolysis-gas chromatography-mass spectrometry (Pyr-GC-MS), SEM-EDS and ESEM-EDS, FTA based FT-IR /imaging and thermogravimetry (TGA) have also been used to identify microplastics polymers (Frias et al. 2010; Claessens et al. 2013; Dekiff et al. 2014; Fries et al. 2013; Lenz et al. 2015; Nuelle et al. 2014; Van Cauwenberghe et al. 2013; Tagg et al. 2015; Dumichen et al. 2015) and some can also characterize inorganic and organic additives in microplastics fragments (Fries et al. 2013) (Table 7.1).

Table 7.1 Analytical techniques used to assess the surface morphology, composition and concentration of microplastics (modified from Rocha-Santos and Duarte, 2015)

Method	Approach and information obtained	Sample preparation (excluding separation)	Advantages/limitations
Scanning electron microscopy (SEM)	Interaction of an electrons beam/sample producing a sample image	Requires coating under vacuum	• High-resolution image • May require coating • Charge effects
Fourier Transform Infrared Spectroscopy (FT-IR)	Spectra collected in Transmittance, Reflectance or Attenuated Total-Reflectance (ATR) mode.	No sample preparation required other than clean-up	• Possible visualization of samples, spectra and map samples • Need a dust free environment for the microscope
Pyr-GC-MS	Mass spectrometry of microplastics by analysing their thermal degradation products	Sampler equipped with a thermal desorption system	• Analyse polymer type and organic plastic additives in one run, avoiding background contamination • Destructive
Raman spectroscopy	Laser excitation, informs about bonding within the material, and about molecule and networking structures	No sample preparation required other than clean-up	• No contact and non-destructive • Apply to very much different materials • Interference with colour/pigment spectras
SEM-EDS[19]	Diffraction and reflection of emitted radiation from microplastics surface	No requirement of coating due to work in low vacuum	• Chemical and morphological characterization of particles
Environmental (E) -SEM-EDS[19]	Diffraction and reflection of emitted radiation from microplastics surface	No sample preparation required	• Elemental composition and surface morphology of microplastics • No charge effects
FTA based FT-IR	Focal Plane Array-Based Reflectance Micro-FT-IR Imaging	30% hydrogen peroxide (H2O2) pre-treatment	• Works in organic-rich waste water samples
Thermal decomposition method	thermogravimetry (TGA) with TDS-GCMS detection. Identify and quantify polymer particles	No sample preparation required	• Works in organic-rich wastewater samples • Destructive?

[19] Scanning Electron Microscopy – Energy-Dispersive X-ray Spectroscopy.

7.5 Dealing with uncertainty

Regrettably, environmental levels of microplastics may be difficult to compare due to the lack of consistency in the assays used and technical challenges. The most common discrepancies can be related to environmental conditions during sampling, the matrix examined, the extraction protocol (e.g. digestion of tissues), quality analysis/quality control (e.g. procedural contamination of airborne fibres), the particle size-range assessed, the reporting unit and the analytical method used for identification of plastics (Song et al. 2015). It has been demonstrated that analytical methods may be improved in some laboratories. For example, recording colour, width and length of microfibre airborne contamination, Torre et al. (2015) minimized the flow of airborne contamination by 95% using a plastic sheet covering a stereo microscope.

Ecological significance of results is also important for consideration. Estimates of the impacts of microplastics are usually conducted in laboratories with only one type or size of microspheres at concentrations much higher than environmental levels and based on short- to mid-term (hours to days) exposures. In addition, many experiments are done with clean plastic that does not have a biofilm. This is not environmentally realistic. As a result, effects demonstrated may not be environmentally relevant. Long-term chronic exposures under controlled conditions with environmentally relevant microplastics concentrations, types and exposure scenarios are required for a realistic assessment of microplastic-associated risks.

Selecting suitable and comparable quantification and identification methods for microplastics is crucial for evaluating concentrations of and risks due to microplastic pollution. The appropriate methods must therefore be determined (Song et al. 2015). For example, techniques such as FT-IR and Raman spectroscopy should be adopted globally to help determine particle composition. Visual identification alone is inappropriate for studies on particles below 100 µm. Standardized methodology that is most environmentally relevant will make it possible to better understand contamination and impact.

7.6 Conclusions, knowledge gaps and priorities

7.6.1 *Conclusions*

Management and reduction measures dedicated to microplastics should be based on correct evaluations and consistent monitoring. They must also be based on sound scientific and technical basis and strategies. To date, several studies have demonstrated widespread contamination of microplastics in aquatic habitats and organisms, but limitations and inconsistencies in methods have complicated large-scale assessment. As microplastics are increasingly measured in a greater quantity and diversity of environmental matrices, much effort is required to evaluate and improve methods and develop new products and initiatives, such as reference materials, proficiency testing schemes, ring tests, intercalibration exercises and standard operating protocols (SOPs). These approaches should be developed in the context of dedicated research and will help to ensure that the quality of the data produced meets predefined performance criteria, which may lead to some form of accreditation. Still, the existing data can still be of use for determining the relative state of the environment and informing decisions on possible management measure.

7.6.2 *Knowledge gaps*

One of the main difficulties for assessing microplastics is due to the lack of standardization of sampling and extraction methods for microplastic particles. As sampling, extraction and detection methods and techniques are being developed worldwide, a harmonization and standardization of techniques and protocols is urgently needed. This will help achieve quality control.

7.6.3 *Research priorities*

Further research on methods needs to consider sampling design in terms of (i) the number and the size of replicates, (ii) the spatial area and the frequency of sampling, (iii) the method used for sampling (i.e. type of net for aquatic samples or core for sediment samples), and (iv) methods used for identification of microplastics (Rocha-Santos and Duarte 2015).

Although some methods have been proven useful techniques for monitoring (Galgani 2014; Masura 2015) and identifying the composition of microparticles (Dumichen et al. 2015), the following points have become critical:

- There is still a lack of analytical methods capable of characterizing and quantifying small sized particles, $<$20 to 30 µm diameter, including nanoparticles from environmental samples and consequently assessing their concentration.
- There is a need to harmonize procedures to mitigate airborne contamination. Only in this way the correct levels of microplastic contamination in biota (which is essential for risk assessment) can be determined in a reproducible and science-based manner. In addition, effort should be directed at how to convert legacy data sets to the new harmonized units. This will allow decadal scale comparisons and analysis of trends that cannot be achieved in any other way.
- Better understanding of degradation processes will enable researchers to define chemical indicators, not only for the timespan of polymers at sea, but also to evaluate the rates of degradation and the leachability of pollutants.
- More generally, research will have to focus on developing new tools and strategies in order to optimize sampling effort (considering spatial and temporal variability), and adequately count and characterize microplastics particles.
- Working at oceanic scale requires assessments to be relevant, supporting the development of automated sensors and real time measurements. This will support in situ analysis in a wide range of environmental compartments.

8 AN INITIAL RISK ASSESSMENT FRAMEWORK

Key points

1. Adopting a risk-based approach provides a more robust basis for estimating the impact of microplastics and deciding on an appropriate response
2. Risk is typically placed into one of five categories, from negligible to very high/catastrophic
3. A risk assessment framework includes analysis of the context in which the hazard occurs, the risk assessment, an evaluation of options for treating the risk, communication with relevant stakeholders throughout the process and monitoring and assessment
4. The assessment can be focused on a single protection goal or widen to include several ecosystem components

8.1 Risk, consequence and likelihood

In simple terms risk is defined as the likelihood (or probability) that a consequence (or hazard) will occur. Terms such as likelihood and consequence may be more familiar to a non-technical audience, whereas probability and hazard are terms that may be preferred by specialists. It is an approach that is routinely applied in every aspect of human activity, ranging from formal risk assessments, for example in major construction projects, to informal decision-making by individuals, for example on when to cross a busy road.

Risk = likelihood/probability x consequence/hazard

In the context of marine microplastics, the hazard is the potential impact of plastic particles and the likelihood is the extent or rate of encounter, otherwise referred to as the exposure. The earlier sections of this report describe the source and distribution of the hazard (and microplastics), and the potential impact. Estimating the degree of risk provides a more robust basis for decisions on whether or how to act to reduce the risk, if it is considered unacceptable.

Box 8.1 Definitions under the risk assessment framework

Definition of risk

Risk can be defined as the characteristic of a situation or action in which two or more unknown outcomes are possible, one of which is undesirable (after Covello and Merkhofer 1993).

Definition of hazard

Hazard can be defined as an agent, medium, process, procedure or site with the potential to cause an adverse effect (EC 2000). A hazard produces a risk only if an exposure pathway exists and if exposures create the possibility of adverse consequences (Covello and Merkhofer 1993).

Definition of probability

Probability is a measure of the likelihood of an event occurring. In statistical analysis probability is given a value between 0 and 1, with higher values indicating higher probability of occurrence of the event. In the present context the event represents a hazard, and the assigned probability may be qualitative (e.g. based on expert judgement) rather than fully quantitative, due to a lack of empirical evidence. This introduces an additional *uncertainty* in the risk assessment.

The risk of a significant impact occurring will vary depending on the ecosystem component being assessed, the nature of the hazard and the likelihood of the hazard occurring. GESAMP carried out a risk assessment and risk communication study for coastal aquaculture, in which potential hazards associated with water quality were described in some detail (GESAMP 2008). Hazards were ranked from negligible to catastrophic, and accompanied by a description of the effects (Table 8.1).

Table 8.1 Description of hazards in relation to aquaculture (adapted from GESAMP 2008)

Degree of hazard	Description of hazard
Catastrophic	• Irreversible change to ecosystems performance at the faunal-province [regional] level; or • The extinction of a species or rare habitat
High	• High mortality for an affected species or significant changes in the function of an ecosystem • Effects would be expected to occur at the level of a single coastal or oceanic body • Effects would be felt for a prolonged period after the culture activities stop (greater than the period which the new species was cultured or three generations of the wild species whichever is the lesser time period) • Changes would not be amenable to control or mitigation
Moderate	• Change in ecosystem performance or species performance at a regional or sub-population level, but they would not be expected to affect whole ecosystems • Changes associated with these effects would be reversible • Changes that have a moderately protracted consequence • Changes may be amenable to control or mitigation at a significant cost or their effects may be temporary
Low	• Changes are expected to affect the environment and species at a local level but would be expected to have a negligible effect at the regional or ecosystem scale • Changes would be amenable to mitigation or control • Effects would be of a temporary nature
Negligible	• Changes expected to be localized to the production site and to be of a transitory nature • Changes are readily amenable to control or mitigation

The hazard descriptions can be adapted readily for other ecosystem components, for example:

- Injury or death to endangered species following ingestion of microplastics
- Injury or death to rare or iconic species following ingestion of microplastics
- Injury or death to indicator species following ingestion of microplastics
- Population-level effects due to physical impacts of ingested microplastics
- Population-level effects due to chemical contamination of commercial and non-commercial species following ingestion of microplastics (seafood security)
- Chemical contamination of commercial species (seafood safety)
- Microplastics as a vector for nuisance species
- Loss of biodiversity, resilience and ecological functioning

Similar tables can be developed for a variety of maritime sectors or ecosystem components (i.e. species, habitats, functional groups) and for a wide range of potential hazards.

8.2 Risk assessment

Risk assessment is a useful tool for systematically evaluating and organizing information, and the associated assumptions/uncertainties, to facilitate the understanding of relationships between stressors and ecological effects. Environmental risk assessments focus on evaluating the likelihood of adverse environmental or ecological effects resulting from one or more anthropogenic environmental stressors (GESAMP 2008). Risk assessments generally follow a similar set of steps, and a variety of conceptual frameworks have been proposed to illustrate this process. These tend to have a number of common features, beginning with problem identification and formulation (risk identification), followed by a characterization of exposure and effect (risk analysis). This allows a societal decision to be made as to whether the risk is considered acceptable or unacceptable (risk evaluation) (GESAMP 2008). If the risk is deemed unacceptable then options for reducing the risk can be considered (risk treatment or risk management). This forms part of the Response component of the DPSIR conceptual framework (Section 1.3).

More formally, McVicar (2004) described risk analysis as:

> … a structured approach used to identify and evaluate the likelihood and degree of risk associated with a known hazard. It leads to the implementation of practical management action designed to achieve a desired result regarding protection from the hazard. Actions taken should be proportionate to the level of the risk. This provides a rational and defendable position for any measures taken to allow meaningful use of resources and for the focus to be on the most important areas that can be controlled. Risk management requires that all possible major hazards to the matter of concern should be identified.

A risk matrix can be developed to illustrate the strength of a hazard (or effect) and the probability or likelihood of the occurrence of the effect (or concentration), for a particular environmental stressor and ecosystem component (Figure 8.2); for example, the impact of PBDE flame-retardants associated with ingested microplastics on the reproductive success of oceanic seabirds. Transfer of PBDEs from ingested plastics into adipose tissue of the short-tailed shearwater *(Puffinus tenuirostris)* has been supported by analysis of PBDE cogeners (Tanaka et al. 2013).

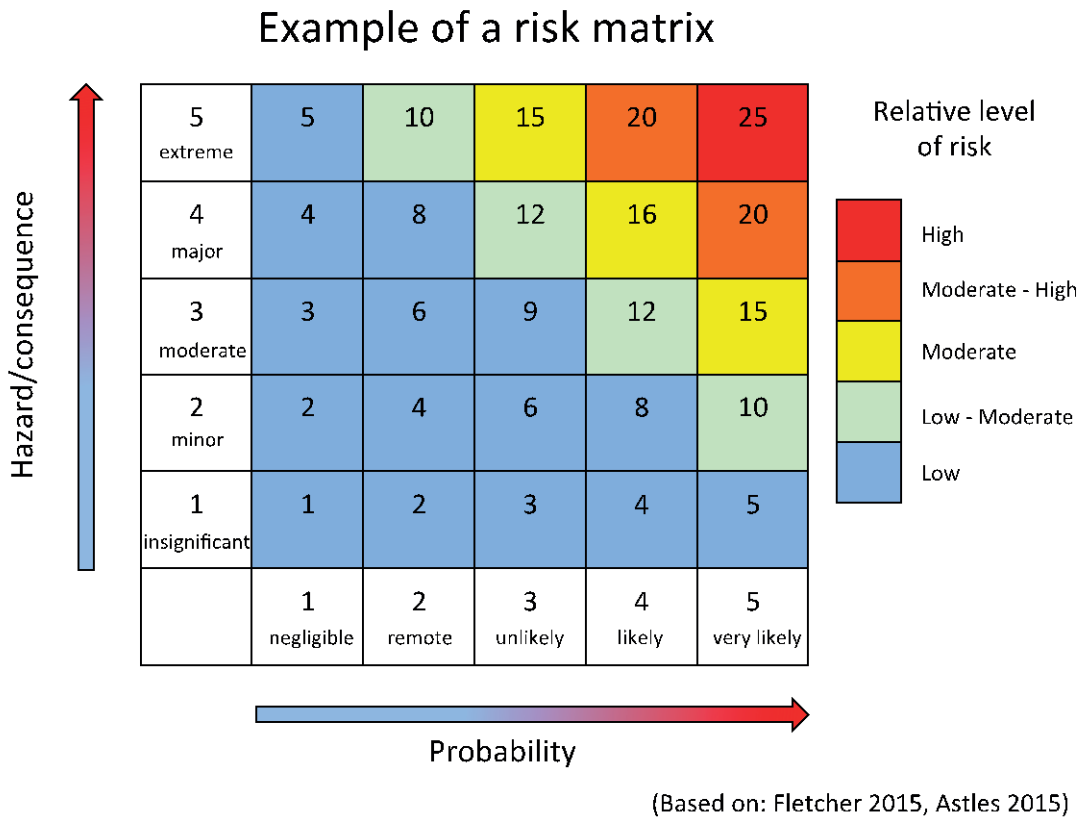


Figure 8.2 *Risk matrix linking probability/likelihood of occurrence to degree of potential hazard/consequence (based on Fletcher 2015, Astles 2015)*

From the above discussion it can be seen that it is not appropriate to consider the risk from microplastics as a single entity. The hazards associated with microplastics will have physical, chemical and biological aspects, and their potential impact on the ecosystem will be species dependent, as well as varying spatially and temporally. For illustrative purposes, Table 8.2 presents a summary of *hypothetical* potential hazards associated with the ingestion of microplastics containing potentially hazardous chemicals, added during manufacture, by a species of finfish. In this case the hazard is characterized by the known endocrine-disrupting properties of some additives, with the potential to affect the viability of the commercial fish stock and the availability of seafood that is safe to eat. It is possible to produce hazard tables for a range of ecosystem components and microplastic characteristics, based on a combination of observations, laboratory experiment and expert elicitation. But, it is very difficult to assign probabilities of occurrence with the present level of knowledge.

Table 8.2 Description of the hypothetical risk level, from Insignificant to Severe, due to chemical contamination (= Hazard) resulting from the ingestion by biota of microplastics containing additive chemicals, for five societal objectives. This table is for illustrative purposes only – there is no evidence that any chemical additives in marine plastics are presenting a significant risk at present

Objective	Insignificant	Minor	Moderate	Major	Severe
Target species	No measureable effect	Very unlikely to effect fish stocks	Some minor effect on fish stocks	Significant effect on fish stocks	Significant depletion of stock
Food security	No measureable effect	Very unlikely to effect fish stocks	Possibility of some minor effect on fish stocks but no discernible affect on market availability	Some market shortages	Widespread market shortages

Objective	Insignificant	Minor	Moderate	Major	Severe
Food safety	No measureable effect	Very unlikely to experience neurological damage to unborn and developing children	Unlikely to experience neurological damage to unborn and developing children	Potential for significant neurological damage for unborn and developing children of high seafood consumers	Significant neurological damage to unborn and developing children – all seafood consumers
Ecosystem effects/ food chain	No measureable effect	Very unlikely to experience depletion of prey species	Unlikely to experience depletion of prey species	Some depletion of prey species	Significant depletion of prey species for commercial species
Consumer choice (perceived risk)	No measureable effect	Very unlikely to influence consumer choice	Some concern may be expressed by potential consumers	Some rejection of affected seafood by susceptible consumers	Large-scale rejection of affected seafood

In theory a risk matrix could be developed to cover all probable occurrences and severities, in combination with a description of the hazard (Table 8.2), and used to illustrate what level of risk is considered unacceptable (Figure 8.3). In practice, as demonstrated in Sections 2 to 6 above, we lack much of the information required, but the matrix is still a useful tool for exploring the probability of effects.

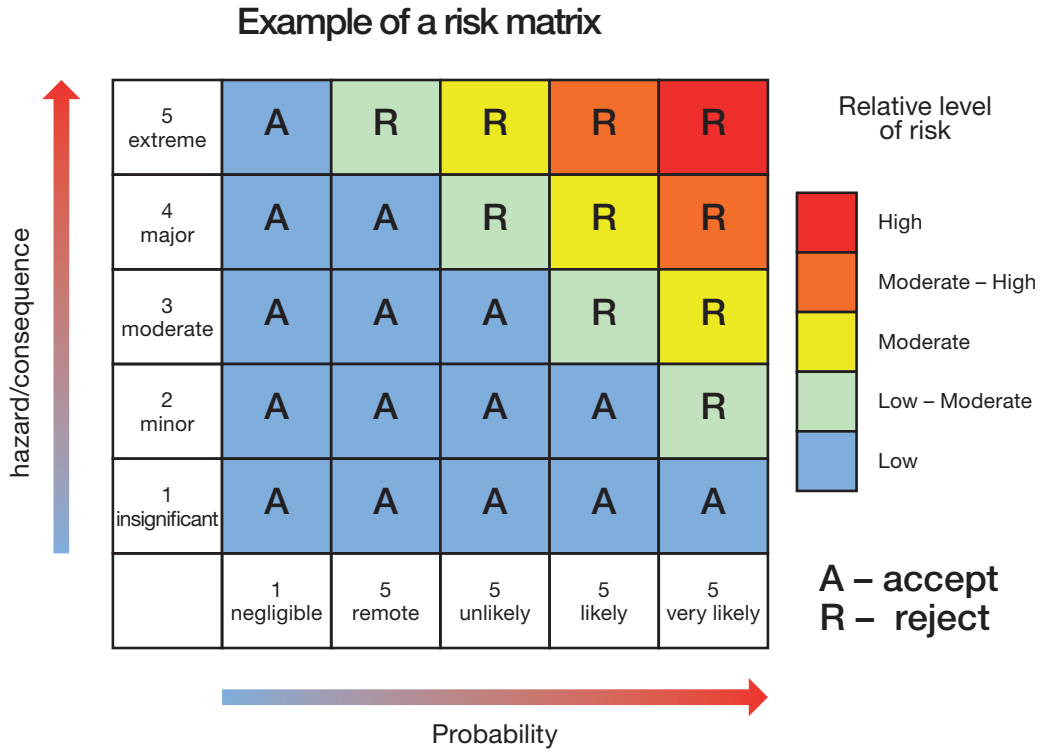


Figure 8.3 *Risk matrix indicating the acceptable level of risk for a hypothetical hazard, illustrated in Figure 8.2 (based on GESAMP 2008)*

8.3 Conceptual framework

Risk Assessment Frameworks provide a means of formalizing the process of examining a system in context, describing possible consequences if a failure in the system occurs and predicting the likelihood of a failure occurring (Figure 8.4). Evaluating the context is an essential first step (Fletcher 2015). This requires communication and consultation with those individual or organizational stakeholders who may be directly or indirectly affected, a process which should be maintained throughout. The risk assessment consists of three stages: risk identification, risk analysis and risk evaluation. A decision can then be made on the best way to treat this risk. The overall environmental system and risk assessment process need to be monitored and kept under review so that adjustments can be made as new information becomes available. The description of the approach, and the examples given in Figure 8.5, were developed as part of the UNEP report on marine plastics and microplastics for the UNEA-2 (UNEP 2016). This material is reproduced here to provide context although the reader is encouraged to consult the UNEA report.

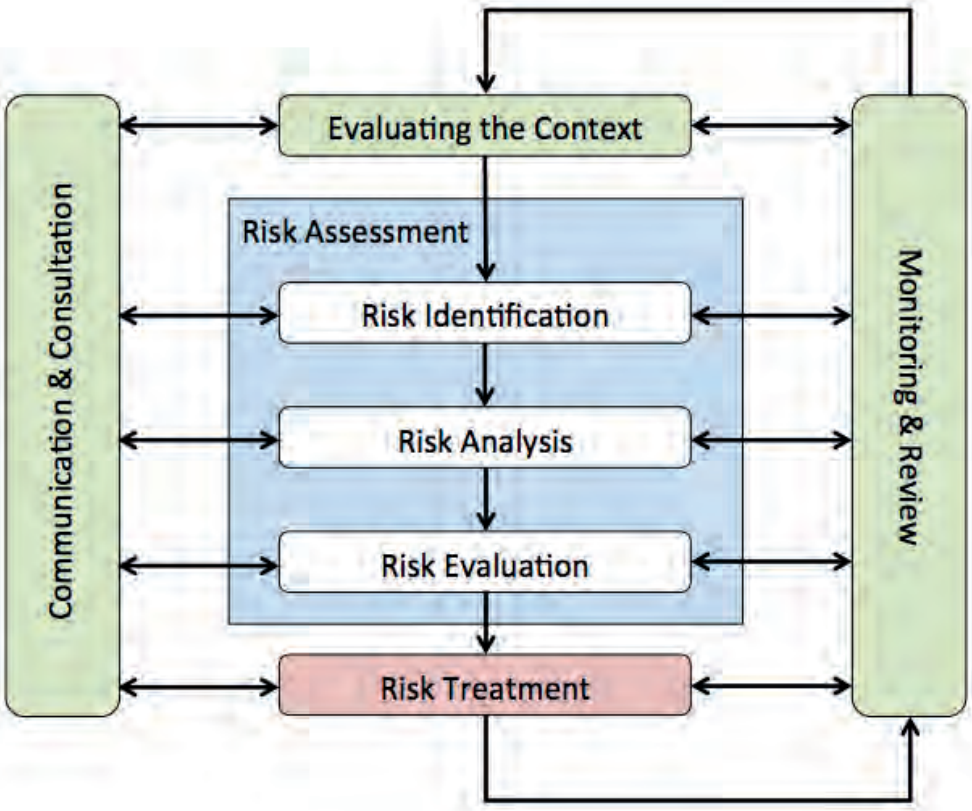


Figure 8.4 *Risk Assessment Framework proposed by Fletcher (2015). Risk treatment is also referred to as risk management (taken from UNEP 2016)*

One difference between assessing the impacts of macro- and microplastic debris is that the former tends to be easier to demonstrate when it occurs. A turtle suffering injury of death as a result of becoming entangled in ALDFG needs little further explanation to illustrate the nature of the hazard and the degree of risk. Public perceptions of the degree of risk from microplastics are more likely to be influenced by more general considerations of potential hazards which are less readily understood; for example, levels of chemical or radioactive contamination which are declared 'safe' by politicians, industry spokespeople or 'experts', but where there is a lack of confidence in official assurances. This disparity between perceived and actual risk is well illustrated by the case of radioactive contamination of seafood following the Fukushima Dai-ichi nuclear accident in 2011, as a consequence of the Tōhoku earthquake and tsunami.

8.4 Case studies

As yet there appear to be no published studies of microplastic impacts that have used a formal risk assessment approach. However, an illustration of how the framework can be applied more generally to floating plastic is provided in Figure 8.5, using information provided in a publication by Wilcox et al. (2015) in which the risk to turtles was assessed from floating ALDFG in the Gulf of Carpentaria, northern Australia.

Although the research on microplastics is increasing rapidly many data gaps exist that prevent the completion of a full risk assessment (Van der Meulen et al. 2014). However, we can learn a great deal by using the framework to outline potential risks and to identify the most pressing research needs. The application of the framework to a hypothetical case of bivalve aquaculture is presented in Figure 8.6. The risk is to the human population from consumption of shellfish contamination by chemicals associated with ingested microplastics. In this example it is assumed that contaminant levels will be elevated over 'background' levels but within national or international (FAO/WHO Codex Alimentarius) guidelines; i.e. the potential health risk to human consumers is considered within acceptable limits, as defined by more familiar risk assessment methods used for hazardous chemicals. However, in this case risk treatment is considered justified by the probability that consumers will change their purchasing preferences because of a perception that the seafood is 'unsafe'.

There are costs and benefits associated with most food consumption. It is necessary to balance the assured benefits of fish consumption with the potential risk due to seafood contamination. However, misguided perceptions that exaggerate the likelihood of harm may result in a cost to the consumer in terms of removing a wholesome source of protein and energy. This emphasizes the need for clear, trustworthy, objective and unambiguous communication of the potential risks and benefits involved (FAO 2014).

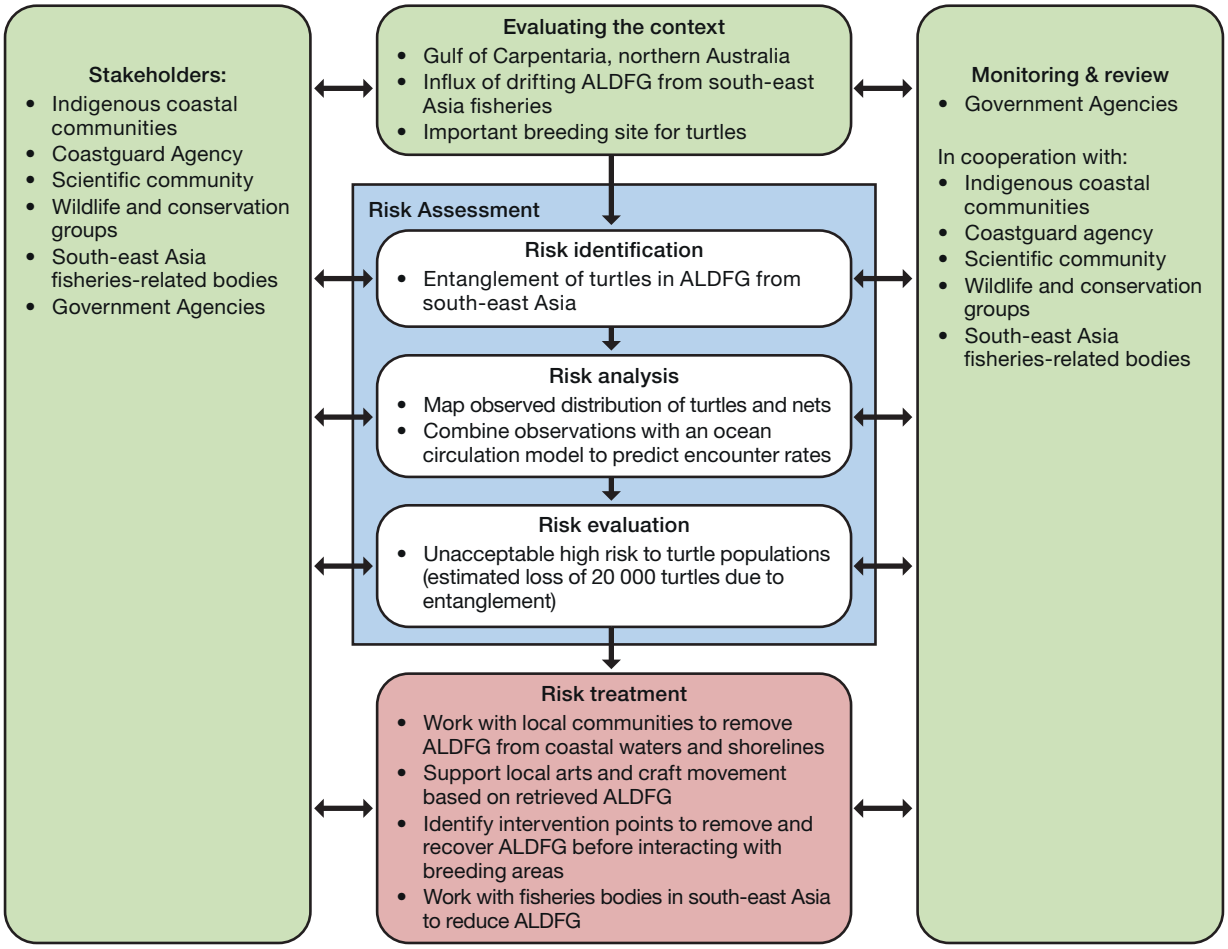


Figure 8.5 *Application of the Risk Assessment Framework to a real-life example of the impact of macroplastic debris, assessing the risk to marine turtles in the Gulf of Carpentaria from floating ALDFG, using information published by Wilcox et al. (2015) (taken from UNEP 2016)*

8.5 Conclusions, knowledge gaps and priorities

8.5.1 *Conclusions*

The field of risk assessment is quite mature. However, it has not been systematically applied to assess the impact of microplastics in the marine environment. It is important to acknowledge the challenges in conducting a risk assessment of this nature. Microplastic impacts are likely to be sub-lethal (i.e. they will compromise individual fitness rather than cause death) and will interact with other stressors (e.g. other pollutants, climate change) creating cumulative effects which could be additive or synergistic. This will make it difficult to tie observed organismal and ecosystem impacts to one stressor. However, development of a framework for both organismal and human risk, that includes all of the factors affecting exposure and impact, would still be valuable to identify the most urgent research needs and management actions moving forward. It has been argued that the precautionary approach should be applied in the case of marine plastics and microplastics; i.e. there is sufficient information available to warrant taking action to reduce inputs and exposure, even though we lack thorough quantitative evidence (UNEP 2016). In the present context, there are several aspects to assessing the risk of an effect that include ecological, social (including human health) and economic components.

8.5.2 *Knowledge gaps*

There is a lack of quantitative information about the physical and chemical impacts of microplastics, and their associated chemicals, on marine organisms. The presence of microplastics in a wide range of taxa has been demonstrated but it is not clear to what extent this compromises individuals or populations. Without such information it is very challenging to ascribe the level of risk associated with a particular loading of microplastics.

The degree to which chemicals associated with ingested plastics transfer across the gut and contaminate the organisms is largely unknown. This is a critical gap that prevents the understanding of the extent to which plastics add to the body burdens of such chemicals in organisms, and hence add to the risk of ingestion of contaminants in seafood. There have been attempts to apply theoretical approaches to bridge this gap (Koelmans et al. 2016) but uncertainties remain about the assumptions that such approaches require, especially given differences in the physiology, anatomy and metabolism of different organisms at varying trophic levels.

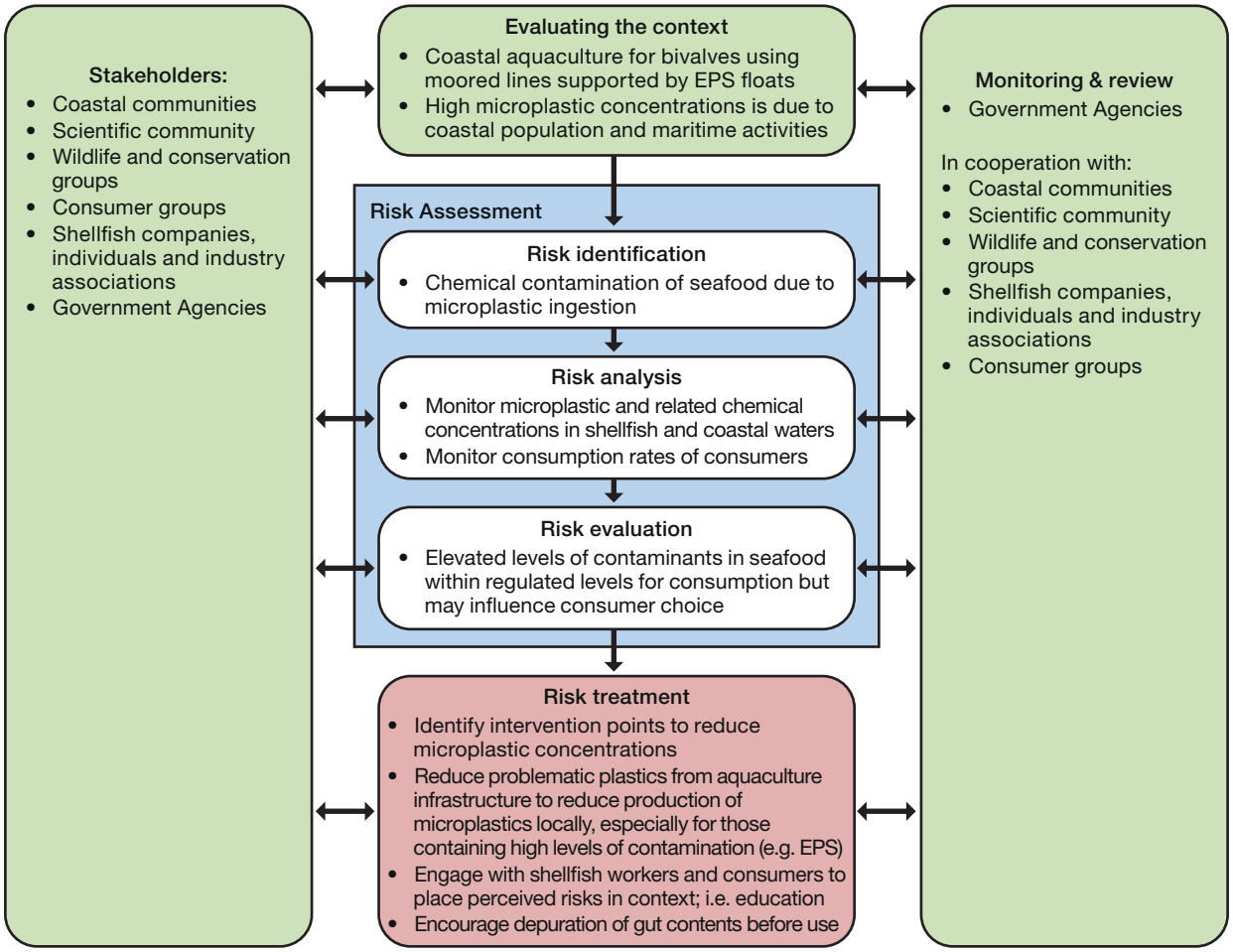


Figure 8.6 *Application of the Risk Assessment Framework to a hypothetical example of the impact of microplastics ingestion by bivalves in aquaculture (UNEP 2016)*

Further uncertainty is introduced by a lack of harmonization of measurement and assessment approaches, and the need for improved quality standards (see Section 7).

8.5.3 *Research priorities*

In order to improve the reliability of risk assessments for microplastics a number of key research priorities need to be addressed in order to:

1. better define the range of potential effects, in terms of ecological, social and economic aspects;

2. better define the probability of such effects occurring;

3. develop risk tables and risk matrices for a range of taxa;

4. include commercially important species, key prey species and sentinel species for monitoring and assessment purposes;

5. include examples of additive chemicals and absorbed POPs and PBTs;

6. include samples of potential transport, survival and consequence of alien species on microplastics;

7. address physical effects for an appropriate range of particle size and shape, including nano-size plastics and fibres; and

8. further develop the risk assessment framework and apply it to a range of case studies, covering different taxa and regional concerns.

9 KEY CONCLUSIONS

There has been increasing attention paid to the occurrence and impacts of microplastics within the past decade, by NGOs, researchers, policy makers, international agencies, regional seas organizations, funding bodies and the media. This has been accompanied by a marked increase in the number of publications in both the peer-reviewed and 'grey' literature. In this report we have reviewed the literature in an attempt to synthesize the current weight of evidence to make our understanding regarding microplastic debris available to key stakeholders, including managers and policy makers. Below are some key conclusions from our report.

Sources

There are many sources of microplastics to the marine environment, including terrestrial and maritime, and there is evidence that microplastics are littered into the environment at all steps in the lifecycle of a plastic product, from production to waste management. Microplastics can enter the marine environment via riverine systems, coastlines, directly at sea from vessels and platforms or by wind-induced transport in the atmosphere. Broadly, sources of microplastics are categorized into two types: primary and secondary. The distinction 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). Fragmentation and degradation plays an essential role in the formation of secondary microplastics, but the processes remain poorly understood.

Distribution, fate and 'hot-spots'

Understanding the sources, fate and transport of microplastics in the marine environment is a growing field and increasingly important to guide management decisions. The fate and transport of microplastics is complex and driven by myriad factors including: weathering and fragmentation, winds, buoyancy (plastics properties), local and large-scale currents, wave action and biofouling. Understanding fluxes of microplastics and hot-spots of microplastics distribution requires understanding movement between these compartments. Microplastics are distributed between the ocean surface, the water column, the seafloor, the shoreline and in biota. The physical, chemical and biological processes acting on the microplastics within each reservoir or compartment differ. Due to lack of data for most of the compartments, the risks and opportunities for mitigation are poorly understood at present. Harmonizing the multiple existing approaches to sampling, measuring and quantifying microplastics will improve local, regional and global understanding and support much-needed, large-scale syntheses.

Ecological impacts

Microplastics have been documented in a diversity of habitats and in over 100 species of biota. Microplastics can impact an organism at many levels of biological organization. Still, the majority of the evidence is for sub-organismal effects (e.g. changes in gene expression, inflammation, tumour promotion) or effects on individual organisms (i.e. death). Microplastics can present a physical hazard, but can also be a source of hazardous chemicals to organisms. The importance of microplastics as a source of chemicals relative to others (e.g. water, sediment, diet) remains under investigation. Microplastics can also act as a vector for invasive species, including harmful algal blooms and pathogens. Nano-sized plastics are probably as common as micro-sized plastics, yet the hazards are less understood and may be more complex.

Commercial fish and shellfish

Capture fisheries and aquaculture sectors provide an important protein source that may be negatively affected by microplastic pollution. Microplastics have been documented in finfish, shellfish and crustaceans, which are consumed by humans. The impacts of the consumption of microplastics by food fish are unknown; however, studies on non-commercial species suggest microplastics have the potential to negatively affect organism health, and hence food security although at current observed concentrations this appears to be unlikely. It is possible that microplastics may increase the chemical contamination of seafood, but there is little evidence to suggest that this represents a significant increase in risk to human health at the current observed microplastic concentrations.

Socio-economic aspects

There is growing concern, globally and by sector, about the increasing cost both of inaction and action needed across the value chain. Whilst the benefits of action against macroplastics often outweigh their costs, downstream clean-up actions for microplastics are unlikely to be cost-effective. It is in the interests of those employed in many sectors of the economy to find strategies to reduce marine litter, as this can help reduce social and economic burdens. Examples include: tourism and recreation, aquaculture and fisheries, and shipping. In parallel, citizen consumption of goods and services, personal habits (e.g. use of reusable bags and packaging) and waste practices (littering, waste separation) are a key driver of marine litter.

Mitigating the effects of marine litter can benefit communities (e.g. through awareness raising, education), support long-term livelihoods (e.g. links to fisheries or tourism), well-being (e.g. linked to recreation) and social cohesion (e.g. sense of belonging to a clean environment). A range of factors influence perceptions and behaviour, such as: cultural norms, gender, social standing, education level and economic status. Accounting for these in the design and implementation of measures to encourage behaviour change may result in longer lasting, more effective and lower-cost solutions. The overarching need is for plastic and its value to be kept within the economy and out of the seas, via a range of circular economy measures. This will help avoid the costs of their impacts on health, environment, society and the economy

Method development and harmonization

In many cases, environmental levels of microplastics may be difficult to interpret due to the lack of consistency in the assays used and technical challenges. A number of factors may effect observations of the distribution of microplastics in the environment, including spatial and temporal variability, types of particles, proximity to rivers, variety of approaches, sampling methods, size limits, extraction methods, characterization and reporting units. Obtaining a 'representative' sample can be problematic. As sampling, extraction, detection methods and techniques are developed worldwide, a harmonization and standardization of techniques and protocols is urgently needed to better assess risk in a reproducible manner, and assist in data comparisons.

An initial risk assessment framework

Adopting a risk-based approach provides a robust basis for estimating the impact of microplastics and deciding on an appropriate response. Risk is typically placed into one of five categories, from negligible to very high/catastrophic. A risk assessment framework includes analysis of the context in which the hazard occurs, the risk assessment, an evaluation of options for treating the risk, communication with relevant stakeholders throughout the process and monitoring and assessment.

10 KEY RECOMMENDATIONS

Although key recommendations for management were not part of our terms of reference, below we list what our group agreed may be useful for consideration. Recommendations are arranged by chapter.

Chapter 2: Sources

- Find the intervention point to stop debris at the source.
- Target mitigation in local waste streams.
- Phase out plastics that are designed to be littered (e.g. microbeads).
- Create incentives for recycling.
- Design and produce plastics that have a more recoverable end-of-life strategy.
- Reduction of single-use items.
- Build more infrastructure for waste management in the rapidly developing world.
- Raise awareness by teaching others where marine debris comes from and ultimately goes.

Chapter 3: Distribution, fate and 'hot-spots'

- Focus source reduction and clean-up efforts in locations with heavier sources of marine litter.
- Target hot-spots that overlap with Marine Protected Areas for mitigation.
- Raise awareness about the issue in regions that are considered hot-spots.
- Use government intervention to fund large-scale clean-up in regions with large concentrations of marine litter.

Chapter 4: Ecological impacts

- Developing educational and awareness programmes that describe the most up to date scientific research regarding the impacts of microplastic on ecosystems to industry, non-governmental organization and government agencies.
- Developing educational and awareness programmes for the public and students at all levels to increase motivations for actions that help mitigate the pollution (e.g. behavioural changes, policy engagement).

Chapter 5: Commercial fish and shellfish

- Put identification markers on fishing and aquaculture nets to keep track of lost gear.
- Redesign fishing and aquaculture equipment to be more environmentally sustainable (e.g., phase out expanded polystyrene buoys).
- Include microplastic contamination as a criterion for aquaculture site selection.
- Reduce practices that could increase microplastic generation around farms (e.g. pressure washing of nets)
- Integrate microplastic into seafood guidelines for sustainability and food safety.
- Fishery gear recapture schemes that provide incentives for recovering lost gear.
- Increase port facility infrastructure for waste removal and recovery.

Chapter 6: Socio-economic aspects

- Create a cost for plastic polluters, e.g. through application of extended producer responsibility.
- Increase the cost of plastic, e.g. by internalizing external costs of end-of-life waste management for plastics, and/or cost of addressing littering/marine litter.
- Make plastic more valuable to encourage reuse, repair, remanufacture and recycling.
- Increase the level of encouragement for separate waste collection by households.
- Put taxes/deposit-refund fees on (plastic) bottles and bags.
- Pay fishermen to collect litter.
- Invest in new and improved waste management infrastructure, riverine, port and beach infrastructures.
- Increase awareness campaigns and engage more stakeholders.
- Encourage positive changes in behaviour.

Chapter 7: Method development and harmonization

- Record the amount of litter removed from beaches globally in standardized units.
- Create global standardized technology for monitoring.
- Create a rapid method for assessing microplastic.
- Design cost-effective methods.
- Harmonize sampling and quantification methodologies.

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Section 3 and Annex III

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ANNEX I – MEMBERSHIP OF THE WORKING GROUP 40

Member	Affiliation
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ANNEX II – WG40 WORK PLAN

To use a combination of workshops, teleconferencing and inter-sessional work, including the WG40 Basecamp site administered by IMO

ToR	Approach
1	Identify probable 'hot-spots' of land- and sea-based sources for plastic and microplastics, using a combination of targeted modelling, knowledge of actual and potential sources (e.g. coastal tourism, aquaculture, fisheries, riverine inputs, urban inputs), environmental and societal data. This can help to inform the development of effective measures in other regions
2	Include coastal-open ocean scaling, 3D circulation, varying particle properties (e.g. size, density). To contribute to a modelling workshop involving a wider group of modellers to improve current assessment methodologies
3	Combine field observations of commercial species, trophic transfer, laboratory experiments of uptake and effects, engagement with the shellfish and fisheries industry (regional cases), consumer preference, potential human health implications
4	Combine collation and analysis of field observations with modelling and sources
5	Critically review laboratory-based experiments examining the behaviour and potential effects of nanoplastics and assess their relevance to the natural environment. Improve sampling and detection methods for nano-sized plastic particles, particularly in biota. Include expertise on human and non-human physiology, biomedicine and toxicology; nano-sciences and nano-engineering in future assessments.
6	Compare information from laboratory-based experiments of organism-chemical behaviour with field-based observations. Include expertise on animal behaviour and physiology for target species, including important commercial species. Take account of gut retention times and the conditions inside the gut when assessing risk. Include a consideration of particle size and shape when assessing risk of damage.
7	Review the published evidence on NIS introductions and potential vectors (e.g. ship hull transfer, ballast water transfer), to estimate the relative importance of plastics and microplastics as a transport vector. Review epidemiological evidence for the occurrence of outbreaks of pathogenic disease associated with NIS. Undertake a targeted risk assessment based on existing data on NIS introductions and disease outbreaks, and utilize existing circulation models to identify key transport routes for pathogenic organisms and the conditions favourable for growth.
8	Review existing and developing standards, assess common usage, identify weaknesses in current definitions and methodologies, propose new guidelines for review by the wider scientific community.
9	Compile regionally resolved socio-economic data relating to sources and effects of plastics and microplastics, including individual and organizational attitudes. Work with local/regional public and private sector organizations, NGOs, special interest and citizens groups including using the GPML.
10	Engage with communications experts within the sponsoring and participant's organization, as well as external experts, to help define and develop appropriate channels for effective communication of all aspects of the study to a range of target groups (policy/governance, private sector, media, public, research community)

Workshops

Rome

Paris

Guayaquil

ANNEX III – LABORATORY STUDIES AND FIELD OBSERVATIONS OF MICROPLASTICS IN MARINE ORGANISMS

Table AIII.1 Laboratory studies exposing fish, shellfish and other marine organisms to microplastics

Organisms which have a commercial interest have a * after the species name. Table includes all published studies until 11 November 2015.

Species	Common name	Size of ingested material	Polymer	Exposure concentration	Length of exposure	Particle endpoint	Effect	Source
Phylum Dinoflagellata								
Oxyrrhis marina		7.3 µm	PS	3000 per mL	1 hr	Digestive tract	Ingestion	Cole et al. 2013
Phylum Chlorophyta								
Tetraselmis chuii		1 – 5 µm	PE	0.000046 – 0.0015 per mL	96 hrs	Cellular	No significant effect on growth, did not interact with toxicity of copper	Davarpanah and Guilhermino 2015
Scenedesmus spp.		20 nm	PS	1.6 – 40 mg per mL	2 hrs	Cellular	Absorption, ROS increased, photosynthesis affected	Bhattacharya et al. 2010
Phylum Haptophyta								
Isochrysis galbana		2 µm	PS	9 x 104 per mL	6 hrs	External	Microspheres attached to algae, no negative effect observed	Long et al. 2014
Phylum Dinophyta								
Heterocapsa triquetra		2 µm	PS	9 x 104 per mL	6 hrs	External	Microspheres attached to algae, no negative effect observed	Long et al. 2014
Phylum Cryptophyta								
Rhodomonas salina		2 µm	PS	9 x 104 per mL	6 hrs	External	Microspheres attached to algae, no negative effect observed	Long et al. 2014
Phylum Ochrophyta								
Chaetoceros neogracilis		2 µm	PS	9 x 104 per mL	6 hrs	External	Microspheres attached to algae, no negative effect observed	Long et al. 2014
Phylum Ciliophora								
Strombidium sulcatum		0.41 – 10 µm	-	5% to 10 % ambient bacteria concentration	1 hr	Digestive tract	Ingestion	Christaki et al. 1998
Tintinnopsis lobiancoi		10 µm	PS	1000, 2000, 10,000 per mL	3 hrs	Digestive tract	Ingestion	Setälä et al. 2014

Species	Common name	Size of ingested material	Polymer	Exposure concentration	Length of exposure	Particle endpoint	Effect	Source
Phylum Cnideria								
Obelia sp.		20.6	PS	2240 per mL	1 hr	Digestive tract	Partial ingestion	Cole et al. 2013
Dipsastrea pallida	Coral	10µm – 2mm	PP	0.395 mg per mL	48 hrs	Mouth and mesenteries of polyps	Ingestion	Hall et al. 2015
Phylum Rotifera								
Synchaeta spp.		10 µm	PS	2000 per mL	3 hrs	Digestive tract	Ingestion	Setälä et al. 2014
Phylum Annelida								
Arenicola marina	Lugworm	20 – 2000 µm	-	1.5 mg per mL	several days	Digestive tract	Ingestion	Thompson et al. 2004
Arenicola marina	Lugworm	130 µm	U-PVC	0% to 5% by weight	48 hour, 4 weeks	Digestive tract	Ingestion, reduced feeding, increased phagocytic activity, reduced available energy reserves, lower lipid reserves	Wright et al. 2013b
Arenicola marina	Lugworm	230 µm	PVC	1,500 g of sediment	10 days	Digestive tract	Ingestion, oxidative stress	Browne et al. 2013
Arenicola marina	Lugworm	<5mm	HDPE, PVA, PA	0.02, 0.2, 2% of sediment	31 days	Digestive tract	Concentration in sediment had significant effects on the metabolic rate of lugworms (increase mp = increase metabolic rate)	Green et al. 2015
Arenicola marina	Lugworm	400 – 1300 µm	PS	0, 1, 10, 100 mg per mL	28 days	Faeces	Ingestion, reduced feeding, weight loss	Besseling et al. 2013
Galeolaria caespitosa	Fan worm	3 – 10 µm	-	5000 per mL	20 mins	Digestive tract	Ingestion	Bolton and Havenhand 1998
Marenzelleria spp.		10 µm	PS	2000 per mL	3 hrs	Digestive tract	Ingestion	Setälä et al. 2014
Phylum Mollusca								
Bivalvia (larvae)		7.3 µm	PS	3000 per mL	24 hrs	Digestive tract	Ingestion	Cole et al. 2013
Mytilus edulis*	Blue mussel	30 nm	PS	0, 0.1, 0.2, 0.3mg per mL	8 hrs	Digestive tract	Ingestion, pseudofaeces, reduced filtering	Wegner et al. 2012.

Species	Common name	Size of ingested material	Polymer	Exposure concentration	Length of exposure	Particle endpoint	Effect	Source
Mytilus edulis*	Blue mussel	0 – 80 µm	HDPE	2.5 mg per mL	<96 hrs	Digestive tract, Lymph system	Ingestion, retention in digestive tract, transfer to lymph system, immune response	von Moos et al. 2012 and Köhler 2010
Mytilus edulis*	Blue mussel	0.5 µm	PS	50 µL per 400 mL seawater	1 hr	Digestive tract	Ingestion, trophic transfer to Carcinus maenas	Farrell and Nelson 2013
Mytilus edulis*	Blue mussel	3, 9.6 µm	PS	0.51 mg per mL	12 hrs	Digestive tract, Lymph system	Ingestion, retention in digestive tract, transferred to lymph system	Browne et al. 2008
Mytilus edulis*	Blue mussel	10 µm	PS	2 x10 4 per mL	45 mins	Faeces	Ingestion, egestion	Ward and Tagart 1989
Mytilus edulis*	Blue mussel	10 µm	PS	1,000 per mL	45 mins	Faeces	Ingestion, egestion	Ward and Kach 2009
Mytilus galloprovincialis*	Mediterranean mussel	<100 µm	PS, PE	1.5 mg per mL	7 days	Gills, Digestive tract, Lymph system	presence in haemolymph, gills and digestive gland	Avio et al. 2015
Mytilus galloprovincialis*	Mediterranean mussel	50 nm	PS	1, 5, 50 µg per mL	-	Haemocytes	only the haemocytes were exposed, signs of cytotoxicity	Canesi et al. 2015
Mytilus trossulus*	Bay mussel	10 µm	PS	/	0.5 to 1.5 hr	Digestive tract	Ingestion	Ward et al. 2003
Placopecten magellanicus*	Atlantic Sea scallop	15, 10, 16, 18, 20 µm	PS	1.05 per mL	1 hr	Faeces	Ingestion, retention, egestion	Brillant and MacDonald 2000
Placopecten magellanicus*	Atlantic Sea scallop	15, 10, 16, 18, 20 µm	PS	1.05 per mL	1 hr	Faeces	Ingestion, retention, egestion	Brillant and MacDonald 2002
Crassostrea virginica*	Eastern oyster	10 µm	PS	1,000 per mL	45 mins	Faeces	Ingestion, egestion	Ward and Kach 2009
Crassostrea gigas*	Pacific oyster	2, 6 µm	PS	1,800 per mL for the 2µm size; 200 per mL for the 6µm size	2 months	Digestive tract	Increased filtration and assimilation, reduced gamete quality, slower larval rearing for larvae from MP exposed parents.	Sussarellu et al. 2014
Phylum Echinodermata								
Apostichopus californicus	Giant Californian sea cucumber	10, 20 µm	PS	2.4 per µL	-	Digestive tract	Ingestion, retention	Hart 1991

Species	Common name	Size of ingested material	Polymer	Exposure concentration	Length of exposure	Particle endpoint	Effect	Source
Thyonella gemmate	Stripped sea cucumber	0.25 – 15 mm	PVC, PA	11 g PVC shavings, 60g resin pellets, 2 g nylon line, to 600 mL of silica sand	20 to 25 hrs	Digestive tract	Selective ingestion	Graham and Thompson 2009
Holothuria (Halodeima) grisea	Grey sea cucumber	0.25 – 15 mm	PVC, PA	As above	20 to 25 hrs	Digestive tract	Selective ingestion	Graham and Thompson 2009
Holothuria floridana	Florida sea cucumber	0.25 – 15 mm	PVC, PA	As above	20 to 25 hrs	Digestive tract	Selective ingestion	Graham and Thompson 2009
Cucumaria frondosa *	Orange footed sea cucumber	0.25 – 15 mm	PVC, PA	As above	20-25 hrs	Digestive tract	Selective ingestion	Graham and Thompson 2009
Paracentrotus lividus*	Sea urchin	40 nm	PS	<25 µg per mL	48 hr	Digestive tract	accumulation and embryo toxicity	Della Torre et al. 2014
Lytechinus variegatus	Green sea urchin	3-5 mm	PE	2 mL per 8 mL	24 hr	External	toxic effects, increasing anomalous embryonic development	Nombre et al. 2015
Tripneustes gratilla*	Collector urchin	32 – 35 µm	PE	1, 10, 100, 300 per mL	1, 6 hrs, 9 days	Faeces	Ingestion, egestion	Kaposi et al. 2014
Dendraster excentricus	Eccentric sand dollar	10, 20 µm	PS	2.4 per µL	-	Digestive tract	Ingestion, retention	Hart 1991
Strongylocentrotus sp*	Sea urchin	10, 20 µm	PS	2.4 per µL	-	Digestive tract	Ingestion, retention	Hart 1991
Ophiopholis aculeate	Crevice brittle star	10, 20 µm	PS	2.4 per µL	-	Digestive tract	Ingestion, retention	Hart 1991
Dermasterias imbricate	Leather star	10, 20 µm	PS	2.4 per µL	-	Digestive tract	Ingestion, retention	Hart 1991
Phylum Arthropoda								
Semibalanus balanoides	Barnacle	20 – 2000 µm	-	1 mg per mL	several days	Digestive tract	Ingestion	Thompson et al. 2004
Tigriopus japonicas	Copepod	0.05 µm	PS	9.1 × 1011 per mL	24 hrs	Faeces	Ingestion, egestion, mortality, decreased fecundity	Lee et al. 2013
Tigriopus japonicas	Copepod	0.5 µm	PS	9.1 × 108 per mL	24 hrs	Faeces	Ingestion, egestion, mortality, decreased fecundity	Lee et al. 2013
Tigriopus japonicas	Copepod	6 µm	PS	5.25 × 105 per mL	24 hrs	Faeces	Ingestion, egestion, mortality, decreased fecundity	Lee et al. 2013
Acartia (Acanthacartia) tonsa	Copepod	7 – 70 µm	-	3,000 – 4,000 beads per mL	15 mins	Digestive tract	Ingestion, size selection	Wilson 1973

Species	Common name	Size of ingested material	Polymer	Exposure concentration	Length of exposure	Particle endpoint	Effect	Source
Acartia spp.	Copepod	10 µm	PS	2000 per mL	3 hrs	Faeces	Ingestion	Setälä et al. 2014
Acartia clausi	Copepod	7.3, 20.6, 30.6 µm	PS	635, 2240, 3000 beads per mL	24 hrs	Digestive tract	Size based selection: Ingestion at 7.3 µm, no ingestion at 20.6 µm, partial ingestion at 30.6 µm	Cole et al. 2013
Eurytemora affinis	Copepod	10 µm	PS	1000, 2000, 10, 000 per mL	3 hrs	Faeces	Ingestion, egestion	Setälä et al. 2014
Limnocalanus macrurus	Copepod	10 µm	PS	1000, 2000, 10, 000 per mL	3 hrs	Digestive tract	Ingestion	Setälä et al. 2014
Temora longicornis	Copepod	1.7, 3.8, 7.3, 20.6, 30.6 µm	PS	635, 2240, 3000 beads per mL	24 hrs	Digestive tract	Ingestion	Cole et al. 2013
Temora longicornis	Copepod	20 µm	PS	100 per mL	overnight	Digestive tract	Ingestion 10.7 $\pm$ 2.5 beads per individual	Cole et al. 2014
Calanus helgolandicus	Copepod	20 µm	PS	75 per mL	23 hrs	Faeces	Egestion, ingestion	Cole et al. 2015
Calanus helgolandicus	Copepod	7.3, 20.6, 30.6 µm	PS	635, 2240, 3000 beads per mL	24 hrs	Digestive tract	Ingestion, size-based selection	Cole et al. 2013
Centropages typicus	Copepod	7.3, 20.6, 30.6 µm	PS	635, 2240, 3000 beads per mL	24 hrs	Digestive tract	Ingestion	Cole et al. 2013
Idotea emarginata	Isopod	10 µm	PS	0.3 mg/g – 120 mg/g	3 days	Faeces	Ingestion, presence in stomach, faeces, no evidence of assimilation, no absorbance, no adverse effect on life history	Hamer et al. 2014
Orchestia gammarellus	Amphipod	20 – 2000 µm	-	1 g per individual (n = 150)	several days	Digestive tract	Ingestion	Thompson et al. 2004
Talitrus saltator	Amphipod	10 – 45 µm	PE	10% weight food (0.06 – 0.09 per g dry fish food)	24 hrs	Faeces	Ingestion, egestion after 2 hours	Ugolini et al. 2013
Allorchestes compressa	Amphipod	11 - 700 µm	PE	0.1 per g	72 hrs	Faeces	Ingestion, egestion within 36 hours	Chua et al. 2014
Neomysis integer	Shrimp	10 µm	PS	2000 spheres per mL	3 hrs	Digestive tract	Ingestion	Setälä et al. 2014
Mysis relicta		10 µm	PS	2000 spheres per mL	3 hrs	Faeces	Ingestion, egestion	Setälä et al. 2014

Species	Common name	Size of ingested material	Polymer	Exposure concentration	Length of exposure	Particle endpoint	Effect	Source
Carcinus maenas*	Shore crab	8 – 10 µm	PS	4.0 x 104 per L ventilation 1.0 x 106 per g	16 hrs, 24 hrs, 21 days	Faeces	Ingestion through gills and gut, retention and excretion, no biological effects measured	Watts et al. 2014
Carcinus maenas*	Shore crab	250 – 500 µm	-	180 mg per 9 cubes of feed	3 weeks	Digestive tract	Ingestion, MP presence did not affect PAH uptake	Msc thesis: Zoeter Vanpoucke Mechtild
Uca repax	Fiddler crab	180 – 250 µm	PS	108 mg/kg – 1000mg/kg	2 months	Gills, Digestive tract, Lymph system	2 month exposure, 100% with MP found in gills, stomach, hepatopancreus. More MP exposure, more MP in crab. Not sure of effect	Brennecke et al. 2015
Nephrops norvegicus*	Norway lobster	5 mm	PP	10 fibres per cm^3 fish	24 hrs	Digestive tract	Ingestion	Murray and Cowie 2011
Porcellanidae (zoea)	Decopoda	30.6 µm	PS	635 beads per mL	24 hrs	Digestive tract	Partial Ingestion	Cole et al. 2013
Paguridae (zoea)	Decopoda	20.6 µm	PS	2240 beads per mL	24 hrs	Digestive tract	Partial Ingestion	Cole et al. 2013
Caridea (larvae)	Decopoda	20.6 µm	PS	2240 beads per mL	24 hrs	Digestive tract	Ingestion	Cole et al. 2013
Barchyura (megalopa)	Decopoda	20.6 µm	PS	2240 beads per mL	24 hrs	Digestive tract	Ingestion	Cole et al. 2013
Artemia franciscana	Brine shrimp	40 and 50 nm	PS	5 – 100 µg per mL	48 hrs	Digestive tract	ingestion, no mortality, possible effect on motility, some excretion	Bergami et al. 2015
Nephrops norvegicus*	Norway lobster	500 – 600 µm loaded with 10 µg of PCBs	PE	150 mg microplastics in gelatine food	3 weeks	Faeces	Ingestion, 100% egestion. Increase of PCB level in the tissues. Same increase for positive control. No direct effect of microplastics	Devriese et al. in prep.
Phylum Chordata								
Doliolidae	Tunicata	7.3 µm	PS	3000 beads mL	1 hr	Digestive tract	Ingestion	Cole et al. 2013
Pomatoschistus microps	Common goby	1 – 5 µm	PE	18.4 and 184 µg per L	96 hrs	External	Abnormal swimming behaviour and lethargy, ACHe activity affected	Oliveria et al. 2013
Pomatoschistus microps	Common goby	420 – 500 µm	PE	<30 per fish	3 mins	Digestive tracts	ingestion, significant decrease in predatory performance	de Sa et al. 2015

Species	Common name	Size of ingested material	Polymer	Exposure concentration	Length of exposure	Particle endpoint	Effect	Source
Pomatoschistus microps	Common goby	1 – 5 µm	PE	0.216 mg per L	/	Digestive tracts	the toxicological interaction between MP and Cr(VI) at conc >3.9 mg/l decreased predatory performance (67%) and caused significant inhibition of ACHe activity (<31%)	Luis et al. 2015
Gadus morhua*	Atlantic cod	2, 5 mm	PE	/	/	Faeces	Ingestion, egestion, 5mm held for prolonged periods, emptying of plastics improved by food consumption additional meals.	dos Santos and Jobling 1991
Oryzias latipes* (freshwater species)	Japanese medaka	<0.5 mm	LDPE	Ground up as 10% of diet	1 to 2 months	Digestive tracts	Liver toxicity, pathology, hepatic stress	Rochman et al. 2013a
Oryzias latipes* (freshwater species)	Japanese medaka	<0.5 mm	LDPE	Ground up as 10% of diet	1 to 2 months	Digestive tracts	Altered gene expression, decreased choriogenin regulation in males and decreased vitellogenin and choriogenin in females	Rochman et al. 2014a
Dicentrarchus labrax*	Seabass (larvea)	10 – 45 µm	PE	0-105 per g incorporated with food	8 dph – 26 dph	Digestive tract	Ingestion, no significant increase in growth, effect on survival of larvae. Possible gastric obstruction.	Mazurais et al. 2014
Halichoerus grypus	Grey seal	3 mm	PE	2818 beads (99% recovery)	96 hours	Faeces		Grellier and Hammond 2006
Calonectris leucomelas	Streaked shearwater	3 – 5 mm	PE	1 g of beads exposed to PCBs ~ 97 ng per g	1st day exposed, studied for 42 days	Chemicals in preen oil	Ingestion, chemical transfer	Teuten et al. 2009

Table AIII.2 Field evidence of microplastic ingestion by fish, shellfish and other marine organisms

If mean not available range is reported. Standard deviation is reported where possible. * represents percentage ingestion by total number of individuals, not separated by species. * species which are commercially important

Scientific name	Common name	Number of individuals	% with microplastic	Mean particles per individual (SD)	Range	Polymer	Type of microplastic	Size ingested (mm)	Study location	Source
Phylum Mollusca										
Dosidicus gigas	Humboldt squid	30	26.7	/	0 – 11	/	Nurdles	3 – 5 mm	British Columbia, Canada	Braid et al. 2012
Mytilus galloprovincialis*	Mediterranean mussel	17	/	total: 0.08 (0.09) – 0.34 (0.22 sd) per gram	/	/	fibres, particles	<5 mm	Tagus Estuary, Portugal	Vandermeersch et al. 2015
Mytilus galloprovincialis*	Mediterranean mussel	17	/	mean: 0.11 (0.12) – 0.15(sd 0.33) per gram	/	/	fibres, particles	<5 mm	Ebro Delta Coastal Embayment, Spain	Vandermeersch et al. 2015
Mytilus galloprovincialis*	Mediterranean mussel	5	/	mean: 0.25 (0.26 sd) per gram	/	/	fibres, particles	<5 mm	Goro, Italy	Vandermeersch et al. 2015
Mytilus galloprovincialis*	Mediterranean mussel	5	/	mean: 0.04 (0.09 sd) per gram	/	/	fibres, particles	<5 mm	Amposta, Ebro Delta, Spain	Vandermeersch et al. 2015
Mytilus galloprovincialis*	Mediterranean mussel	18	100	4.33 (2.62)	/	PET, PA, PE	fibres, fragments, pellets	<5 mm	Fish market, china	Li et al. 2015
Mytilus galloprovincialis*	Mediterranean mussel	17	/	mean: 0.05 (0.11)- 0.16 (0.11 sd) per gram	/	/	fibres, particles	<5 mm	Po estuary, Italy	Vandermeersch et al. 2015
Mytilus edulis*	blue mussel	5	/	mean: 0.06 (± 0.13) particles per gram	/	/	fibres	<5 mm	Baie de Saint Brieux, France	Vandermeersch et al. 2015
Mytilus edulis*	blue mussel	5	/	mean: 0.32 (± 0.22) per gram	/	/	fibres, particles	<5 mm	Inschot, The Netherlands	Vandermeersch et al. 2015
Mytilus edulis*	blue mussel	45	/	3.5 per 10 g		/	Fibres	300 – 1,000 µm	Belgium and The Netherlands	De Witte et al. 2014
Mytilus edulis*	blue mussel	36	/	0.36 (±0 .07) per g		/	/	5 – 25 µm	North Sea, Germany	Van Cauwenberghe and Janssen 2014

Scientific name	Common name	Number of individuals	% with microplastic	Mean particles per individual (SD)	Range	Polymer	Type of microplastic	Size ingested (mm)	Study location	Source
Mytilus edulis*	blue mussel	20	/	170 – 375 particles per 5 mussels		/	fibres	/	Nova Scotia, Canada	Mathlon and Hill, 2014
Scapharca subcrenata*	Ark shell	6	100	45 (± 14.98)	/	PET, PA, PE	fibres, fragments, pellets	<5 mm	Fish market, China	Li et al. 2015
Tegillarca granosa*	Blood cockle	18	100	5.33 (± 2.21)	/	PET, PA, PE	fibres, fragments, pellets	<5 mm	Fish market, China	Li et al. 2015
Patinopecten yessoensis*	Yesso Scallop	6	100	57.17 (± 17.34)	/	PET, PA, PE	fibres, fragments, pellets	<5 mm	Fish market, China	Li et al. 2015
Alectryonella plicatula*	fingerprint oyster	18	100	10.78 (± 4.07)	/	PET, PA, PE	fibres, fragments, pellets	<5 mm	Fish market, China	Li et al. 2015
Sinonovacula constricta*	Chinese razor clam	6	100	14.33 (± 5.35)	/	PET, PA, PE	fibres, fragments, pellets	<5 mm	Fish market, China	Li et al. 2015
Ruditapes philippinarum*	carpet shell	24	100	5.72 (± 2.86)	/	PET, PA, PE	fibres, fragments, pellets	<5 mm	Fish market, China	Li et al. 2015
Meretrix lusoria*	orient clam	18	100	9.22 (± 0.46)	/	PET, PA, PE	fibres, fragments, pellets	<5 mm	Fish market, China	Li et al. 2015
Cyclina sinensis*		30	100	4.82 (± 2.17)	/	PET, PA, PE	fibres, fragments, pellets	<5 mm	Fish market, China	Li et al. 2015
Crassostrea gigas*	Pacific oyster	12	30	0.6±0.9	0 – 2	/	fibres		USA	Rochman et al. 2015a
Crassostrea gigas*	Pacific oyster	11	/	0.47(± 0.16) per g		/	/	5 – 25 µm	Atlantic Ocean	Van Cauwenberghe and Janssen 2014
Phylum Arthrapoda										
Lepas spp.*	Gooseneck barnacle	385	33.5	/	1 – 30		/	<5 mm	North Pacific	Goldstein and Goodwin 2013
Neocalanus cristatus	calanoid copepod	960	/	1 particle per 34 zoop			fibre, fragment	556 (149) µm	North Pacific	Desforges et al. 2015
Euphausia pacifica	Euphausid	413	/	1 particle per 7 euph			fibre, fragment	816 (108) µm	North Pacific	Desforges et al. 2015
Nephrops norvegicus*	Norway lobster	120	83	/			/		Clyde, UK	Murray and Cowie 2011

Scientific name	Common name	Number of individuals	% with microplastic	Mean particles per individual (SD)	Range	Polymer	Type of microplastic	Size ingested (mm)	Study location	Source
Crangon crangon*	Brown shrimp	110	/	11.5 fibres per 10 g			95% fibres, 5% films	300 – 1,000 µm	Belgium	Devriese et al. 2015
Phylum Annelida										
Arenicola marina	lugworm			1.2 +- 2.8 g/ w.w				>5 µm	Belgium, NL, France	Van Cauwenberge et al. in Devriese et al. 2015
Phylum Chaetognatha										
Parasagitta elegans	arrow worm	1	100	/		PS	spheres	0.1 – 3 mm	New England, USA	Carpenter et al. 1972
Phylum Chordata										
Phoca vitulina	harbour seal	100 stomachs, 107 intestines	S:11.2, I:1	max: 8 items (s), 7 items (i)			fragments	>0.1	The Netherlands	Bravo Rebolledo et al. 2013
Mesoplodon mirus	True´s beaked whale	1	100	88			fibres, fragment	mean 2.16mm	Connemara, Ireland	Lusher et al. 2015
Megaptera novaeangliae	Humpback whale	1	100	45 items			fragments	1 – 17cm	The Netherlands	Besseling et al. 2015
Arctocephalus spp.	fur seal	145	100	1 – 4 per scat			fragments, beads	4.1 mm	Macquarie Island, Australia	Eriksson and Burton 2003
Chelonia mydas*	green turtle	24	/	total: 11 pellets			pellets	<5 mm	Rio Grande do Sul, Brazil	Tourinho et al. 2010
Menidia menidia	Atlantic silversides	9	33	/			PS	0.1 – 3mm	New England, USA	Carpenter et al 1972
Atherinopsis californiensis*	jacksmelt	7	28.57142857	1.6 ± 3.7			fibres, fragments	0 – 10	USA	Rochman et al. 2015a
Alepisaurus ferox	Longnosed lancetfish	144	24	2.7 (± 2.0)			fragments	68.3 (± 91.1)	North Pacific	Choy and Drazen 2013
Cololabis saira	Pacific saury	52	*35	3.2 (± 3.05)			fragments	2 – 2.79	North Pacific	Boerger et al. 2010
Clupea harengus*	Atlantic herring	2	100	1		PS	ps	0.1 – 3mm	New England, USA	Carpenter et al 1972

Scientific name	Common name	Number of individuals	% with microplastic	Mean particles per individual (SD)	Range	Polymer	Type of microplastic	Size ingested (mm)	Study location	Source
Clupea harengus*	Atlantic herring	566	2		1 to 4		fragments	0.5 – 3	North Sea	Foekema et al. 2013
Clupea harengus*	Atlantic herring	3	100	/			/	/	North Sea	Collard et al. 2015
Sprattus sprattus *	European sprat	111	38.74%	0.88 (0.88)			fibres, granual, film	0.1 – 4.9 mm	Belgium, North Sea	Msc thesis: Zoeter Vanpoucke Mechtild
Spratelloides gracilis*	silverstripe round herring	4	40	1.1 ± 1.7	0-5		0-5 fragments		Indonesia	Rochman et al. 2015a
Alosa fallax *	Twait shad	1	100	1			fragment	<5 mm	North Eastern Atlantic	Neves et al. 2015
Sardina pilchardus*	European pilchard	3	100%	/			/	/	North Sea	Collard et al. 2015
Sardina pilchardus*	European pilchard	99	19%	1.78 ± 0.7				<1 mm	Adriatic sea	Avio et al. 2015
Sadinella longicxeps*	Oil sardine	10	60%	/			fibres	0.5 – 3 mm	Mangalore	Sulochanan et al. 2014
Stolephorus commersonnii*	Anchovy	16	37.5	/			fragments	1.14 – 2.5	Alappuzha, India	Kripa et al. 2014
Engraulis encrasiscolus*	Anchovy	3	100%	/			/	/	North Sea	Collard et al. 2015
Engraulis mordax*	Pacific anchovy	10	30	0.3 ± 0.5	0 to 1		fibres and film		USA	Rochman et al. 2015a
Pollachius virens*	Saithe	1	100	1		PS	PS	0.1 – 3 mm PS	New England, USA	Carpenter et al. 1972
Ciliata mustela	Five-bearded rocklings	113	0 – 10	/		PS	PS	2 mm	Severn Estuary, UK	Kartar 1976
Merlangius merlangus*	Whiting	105	6	1 – 3				1.7 (± 1.5)	North Sea	Foekema et al. 2013
Merlangius merlangus*	Whiting	50	32	1.75 (± 1.4)			fragment, fibres, beads	2.2 (± 2.3)	English Channel	Lusher et al. 2013
Melanogrammus aeglefinus*	Haddock	97	6	1			fragments	0.7 (± 0.3)	North Sea	Foekema et al. 2013

Scientific name	Common name	Number of individuals	% with microplastic	Mean particles per individual (SD)	Range	Polymer	Type of microplastic	Size ingested (mm)	Study location	Source
Gadus morhua*	Cod	80	13	1 – 2			fragments	1.2 (± 1.2)	North Sea	Foekema et al. 2013
Micromesistius poutassou*	Blue whiting	27	51.9	2.07 (± 0.9)			fragment, fibres, beads	2.0 (± 2.4)	English Channel	Lusher et al. 2013
Trisopterus minutus*	Poor cod	50	40	1.95 (± 1.2)			fragment, fibres, beads	2.2 (± 2.2)	English Channel	Lusher et al. 2013
Merlucius merlucius*	Hake	3	100%	1.33 ± 0.57				<1 mm	Adriatic sea	Avio et al. 2015
Merlucius merlucius*	Hake	12	25%	0.33 ± 0.65			4 fibres	<5 mm	North Eastern Atlantic	Neves et al. 2015
Lampris sp. (big eye)		115	29	2.3 (± 1.6)			fragments	49.1 (± 71.1)	North Pacific	Choy and Drazen 2013
Lampris sp. (small eye)		24	5	5.8 (± 3.9)			fragments	48.8 (± 34.5)	North Pacific	Choy and Drazen 2013
Lophius piscatorius*	Monkfish	2	50	0.5			1 fibre	<5 mm	North Eastern Atlantic	Neves et al. 2015
Hygophum reinhardtii		45	*35	1.3 (± 0.71)			fragments	1 – 2.79	North Pacific	Boerger et al. 2010
Loweina interrupta		28	*35	1			fragments	1 – 2.79	North Pacific	Boerger et al. 2010
Myctophum aurolaternatum		460	*35	6.0 (± 8.99)			fragments	1 – 2.79	North Pacific	Boerger et al. 2010
Symbolophorus californiensis		78	*35	7.2 (± 8.39)			fragments	1 – 2.79	North Pacific	Boerger et al. 2010
Diaphus anderseni	Anderson's lanternfish	13	15.4	1			fragments		North Pacific	Davison and Asch 2011
Diaphus fulgens		7	28.6	1			fragments		North Pacific	Davison and Asch 2011
Diaphus phillipsi	Boluin's lanternfish	1	100	1			fragments	0.5	North Pacific	Davison and Asch 2011
Lobianchia gemellarii	Coco's lanternfish	3	33.3	1			fragments		North Pacific	Davison and Asch 2011
Myctophum nitidulum	Pearly lanternfish	25	16	1.5			fragments	5.46	North Pacific	Davison and Asch 2011
Morone americana	White perch	12	33	/		PS	PS	0.1 – 3mm	New England, USA	Carpenter et al. 1972

Scientific name	Common name	Number of individuals	% with microplastic	Mean particles per individual (SD)	Range	Polymer	Type of microplastic	Size ingested (mm)	Study location	Source
Tautogolabrus adspersus	Bergall	6	<83	/		PS	PS	0.1 – 3 mm	New England, USA	Carpenter et al. 1972
Pomatoschistus minutus (As Gobius minutus)	Goby	200	0 – 25	/		PS	PS	2 mm	Severn Estuary, UK	Kartar 1976
Argyrosomus regius*	Meagre	5	60	0.80 (± 0.8)			2 fragments, 2 fibres	<5 mm	North Eastern Atlantic	Neves et al. 2015
Stellifer brasiliensis		330	9.2	0.33 – 0.83			fragments	<1	Goiana Estuary, Brazil	Dantas et al. 2012
Stellifer stellifer		239	6.9	0.33 – 0.83			fragments	<1	Goiana Estuary, Brazil	Dantas et al. 2012
Eugerres brasilianus		240	16.3	1 – 5			fragments	1 – 5	Goiana Estuary, Brazil	Ramos et al. 2012
Eucinostomus melanopterus		141	9.2	1 – 5			fragments	1 – 5	Goiana Estuary, Brazil	Ramos et al. 2012
Diapterus rhombeus		45	11.1	1 – 5			fragments	1 – 5	Goiana Estuary, Brazil	Ramos et al. 2012
		7	71	5 ± 5.2	0-24				Indonesia	Rochman et al. 2015a
Trachurus trachurus*	Horse mackerel	100	1	1			fragments	2.52	North Sea	Foekema et al. 2013
Trachurus trachurus*	Horse mackerel	44	7	0.07 ± 0.25			2 fragments; 1 fibre	<5 mm	North Eastern Atlantic	Neves et al. 2015
Trachurus trachurus*	Horse mackerel	56	28.6	1.5 (± 0.7)			fragment, fibres, beads	2.2 (± 2.2)	English Channel	Lusher et al. 2013
Trachurus picturatus*	Blue jack mackerel	29	3.00%	0.03 ± 0.18			1 fibre	<5 mm	North Eastern Atlantic	Neves et al. 2015
Seriola lalandi*	Yellowtail amberjack	19	10.5	1			fragments	0.5 – 11	North Pacific	Gassel et al. 2013
Decapyerus macrosoma	Shortfin scad	17	29	2.5 ± 6.3	0 – 21		Fragments and PS		Indonesia	Rochman et al. 2015a
Callionymus lyra	Dragonette	50	38	1.79 (± 0.9)			fragment, fibres, beads	2.2 (± 2.2)	English Channel	Lusher et al. 2013
Cepola macrophthalma	Red band fish	62	32.3	2.15 (± 2.0)			fragment, fibres, beads	2.0 (± 1.9)	English Channel	Lusher et al. 2013

Scientific name	Common name	Number of individuals	% with microplastic	Mean particles per individual (SD)	Range	Polymer	Type of microplastic	Size ingested (mm)	Study location	Source
Morone saxatilis	striped bass	7	28.57142857	0.9 ± 1.2	0 – 3		fibre, film, foam		USA	Rochman et al. 2015a
Mullus barbatus*	red mullets	11	64%	1.57 ± 0.78				<1 mm	Adriatic sea	Avio et al. 2015
Mullus surmulletus*	striped red mullet	4	100%	1.75 ± 0.5			7 fibres	<5 mm	North Eastern Atlantic	Neves et al. 2015
Boops boops*	Bogue	32	9	0.09 (±0.3)			1 fragment, 2 fibres	<5 mm	North Eastern Atlantic	Neves et al. 2015
Dentex macrophthalmus*	Large-eye dentex	1	100	1			1 fibre	<5 mm	North Eastern Atlantic	Neves et al. 2015
Brama brama*	Atlantic pomfret	3	33	0.67±1.2			2 fibres	<5 mm	North Eastern Atlantic	Neves et al. 2015
Thunnus thynnus*	bluefin tuna	34	32.40%	/				>0.63 mm	Mediterranean	Romeo et al. 2015
Thunnus alalunga*	albacore tuna	2	50.00%			PE		<3 cm	Arabian Sea	Sajikumar et al. 2013
Thunnus alalunga*	albacore tuna	131	12.90%	/				>3.60 mm	Mediterranean	Romeo et al. 2015
Rastrelliger kanagurta*	Indian Mackerel	10	50.00%	/			fibres	0.5 - 3 mm	Mangalore	Sulochanan et al. 2014
Rastrelliger kanagurta*	Indian Mackerel	9	56	1 (± 1.1)	0-3		fragments, pellets		Indonesia	Rochman et al. 2015a
Scomber japonicas*	Chub mackerel	35	31	0.57 ± 1.04			14 fragments; 6 fibres	<9.42 mm	North Eastern Atlantic	Neves et al. 2015
Scomber scombrus*	Atlantic mackerel	13	31	0.46 ± 0.78			3 fragments; 3 fibres	<5mm	North Eastern Atlantic	Neves et al. 2015
Siganus argenteus	streamlined spinefoot	2	50	0.5 ± 0.7			0 to 1 fragments		Indonesia	Rochman et al. 2015a
Siganus canaliculatus	rabbitfish	3	29	0.3 - 0.6			0 to 1 fragments		Indonesia	Rochman et al. 2015a
Xiphias gladius*	swordfish	56	12.50%	/				>3.69 mm	Mediterranean	Romeo et al. 2015
Pagellus acarne*	Axillary seabream	1	100	1			1 fibre	<5 mm	North Eastern Atlantic	Neves et al. 2015
Citharichthys sordidus*	Pacific sandab	5	60	1 ± 1.2	0-3		fibre and dilm		USA	Rochman et al. 015a

Scientific name	Common name	Number of individuals	% with microplastic	Mean particles per individual (SD)	Range	Polymer	Type of microplastic	Size ingested (mm)	Study location	Source
Pseudopleuronectes americanus*	Winter Flounder	95	2.1	/		PS	ps	0.1 – 3 mm	New England, USA	Carpenter et al. 1972
Platichthys flesus*	Flounder	/	/	/		PS	ps	1 mm	Severn Estuary, UK	Kartar 1973
Platichthys flesus*	Flounder	1090	0 – 20.7	/		PS	ps	1 mm	Severn Estuary, UK	Kartar 1976
Buglossidium luteum	Solenette	50	26	1.23 (± 0.4)			fragment, fibres, beads	1.9 (± 1.8)	English Channel	Lusher et al. 2013
Microchirus variegatus	Thickback sole	51	23.5	1.58 (± 0.8)			fragment, fibres, beads	2.2 (± 2.2)	English Channel	Lusher et al. 2013
Oncorhynchus tshawytscha*	Chinook salmon	4	25	0.25 ± 0.5	0 – 1		fibre		USA	Rochman et al. 2015a
Myoxocephalus aenaeus	Grubby	47	4.2	/		PS	PS	0.1 – 3 mm	New England, USA	Carpenter et al 1972
Ophiodon elongates*	Ling cod	11	9.090909091	0.1 ± 0.3	0 – 1		0-1 film		USA	Rochman et al. 2015a
Liparis liparis liparis	Sea snails	220	0 – 25	/		PS	PS	1 mm	Severn Estuary, UK	Kartar 1976
Sebastes flavidus*	yellowtail rockfish	1	33	0.3 ±0.6	0 – 1		fibres		USA	Rochman et al. 2015a
Sebastes mystinus*	blue rockfish	10	20	0.2 ± 0.4	0 – 1		fibres		USA	Rochman et al. 2015a
Chelidonichthys cuculus*	Red gurnard	66	51.5	1.94 (± 1.3)			fragments	2.1 (± 2.1)	English Channel	Lusher et al. 2013
Chelidonichthys lucernus*	Tub Gurnard	3	0.67	1 ± 0				<1 mm	Adriatic sea	Avio et al. 2015
Trigla lyra*	Piper gurnard	31	19	0.26 ± 0.57			1 fragment; 7 fibres	<5 mm	North Eastern Atlantic	Neves et al. 2015
Prionotus evolans	Striped searobin	1	100	1		PS	PS	0.1 – 3 mm	New England, USA	Carpenter et al 1972
Cathorops spixii	Madamago sea catfish	60	18.3	0.47	1 – 4				Goiana Estuary, Brazil	Possatto et al. 2011
Cathorops spp		60	33.3	0.55	1 – 4				Goiana Estuary, Brazil	Possatto et al. 2011

Scientific name	Common name	Number of individuals	% with microplastic	Mean particles per individual (SD)	Range	Polymer	Type of microplastic	Size ingested (mm)	Study location	Source
Sciades herzbergii	Pemecoe catfish	62	17.7	0.25	1 – 4				Goiana Estuary, Brazil	Possatto et al. 2011
Astronesthes indopacificus		7	*35	1			fragments	1 – 2.79	North Pacific	Boerger et al. 2010
Sternoptyx diaphana	Hatchetfish	4	25	1			fragments	1.58 mm	North Pacific	Davison and Asch 2011
Sternoptyx pseudobscura	Highlight hatchetfish	6	16.7	1			fragments	4.75 mm	North Pacific	Davison and Asch 2011
Idiacanthus antrostomus	Pacific black dragon	4	25	1			fragments	0.5 mm	North Pacific	Davison and Asch 2011
Zeus faber*	John Dory	46	47.6	2.65 (± 2.5)			fragments, fibres, beads	2.2 (± 2.2) mm	English Channel	Lusher et al. 2013
Zeus faber*	John Dory	1	100	1			fibre	<5 mm	North Eastern Atlantic	Neves et al. 2015
Scyliorhinus canicula*	Lesser-spotted catshark	20	20	0.27 ± 0.55			1 fragment; 5 fibres	<5 mm	North Eastern Atlantic	Neves et al. 2015
Raja asterias*	Starry ray	7	43	0.57 (± 0.79)			4 fibres	<5 mm	North Eastern Atlantic	Neves et al. 2015
Squalus acanthias*	Spiny dogfish	9	44	1.25 ± 0.5				<1 mm	Adriatic sea	Avio et al. 2015

Table AIII.3 Evidence of microplastic ingestion by seabirds

mean (±SD unless * = SE).

Species		n	Percentage with plastic (%)	Mean number of particles per individual	Mean size ingested ± SD (minimum – maximum) (mm)	Type of plastic	Location	Source
Family Procellariidae								
(Aphrodroma brevirostris) (As Pterodroma brevirostris)	Kerguelen petrel	26	3.8	1		pellets	North Island, New Zealand	Reed 1981
(Aphrodroma brevirostris) (As Pterodroma brevirostris)	Kerguelen petrel	13	8	0.2	mass <0.0083 g	pellets	Gough Island, South Atlantic	Furness 1985a
Aphrodroma brevirostris (As Pterodroma brevirostris)		63	22.2	/	-	pellets	Southern Ocean	Ryan 1987
Aphrodroma brevirostris		28	7	/	3 – 6 mm	fragments, pellets	Antarctica	Ainley et al. 1990
Cory's shearwater (Calonectris diomedea)		7	42.8	/		pellets	Southern Ocean	Ryan 1987
Calonectris diomedea		147	24.5	Stomach = 2 Gizzard = 3.1		beads	North Carolina, USA	Moser and Lee 1992
Calonectris diomedea		5	100	/	<10		Rio Grande do Sul, Brazil	Colabuono et al 2009
Calonectris diomedea		85	83	8 (± 7.9)	3.9 ± 3.5		Canary Islands, Spain	Rodríguez et al. 2012
Calonectris diomedea		49	96	14.6 (± 24.0)	2.5 ± 6.0 A		Catalan coast, Mediterranean	Codina-García et al. 2013
Cape petrel (Daption capense)		18	83.3	/		pellets	Southern Ocean	Ryan 1987
Daption capense		30	33	1	5		Ardery Island, Antarctica	van Franeker and Bell 1988
Daption capense		105	14	/	3 – 6 mm	fragments, pellets	Antarctica	Ainley et al. 1990
Northern fulmar (Fulmarus glacialis)		3	100	7.6	1 – 4 mm	pellets	California, USA	Baltz and Morejohn 1976
Fulmarus glacialis		79	92	11.9		pellets	Netherland and Arctic colonies	van Franeker 1985
Fulmarus glacialis		8	50	3.9		pellets	St. Kilda, UK	Furness 1985b
Fulmarus glacialis		13	92.3	10.6		pellets	Foula, UK	Furness 1985b

Species	n	Percentage with plastic (%)	Mean number of particles per individual	Mean size ingested ± SD (minimum – maximum) (mm)	Type of plastic	Location	Source
Fulmarus glacialis	1	100	1	4 mm	pellets	Oregon, USA	Bayer and Olson 1988
Fulmarus glacialis	44	86.4	Stomach = 3 Gizzard = 14	-	beads	North Carolina, USA	Moser and Lee 1992
Fulmarus glacialis	19	84.2	Max: 26	-	pellets	Alaska, USA	Robards et al. 1995
Fulmarus glacialis	3	100	7.7	-	pellets	Eastern North Pacific	Blight and Burger 1997
Fulmarus glacialis	15	36	3.6 (± 2.7)	7 (± 4.0)		Davis Strait, Canadian Arctic	Mallory et al. 2006
Fulmarus glacialis	1295	95	14.6 (± 2.0*) – 33.2(± 3.3*)	>1.0		North Sea	van Franeker et al. 2011
Fulmarus glacialis	67	92.5	36.8 (± 9.8*)	>0.5		Eastern North Pacific	Avery-Gomm et al. 2012
Fulmarus glacialis	58	79	6.0 (± 0.9*)	>1.0		Westfjords, Iceland	Kühn and van Franeker 2012
Fulmarus glacialis	176	93	26.6 (± 37.5)		fragments, pellets	Nova Scotia, Canada	Bond et al. 2014
Antarctic fulmar (Fulmarus glacialoides)	84	2	/	2 - 6 mm	fragments, pellets	Antarctica	Ainley et al. 1990
Fulmarus glacialoides	9	79	/	<10		Rio Grande do Sul, Brazil	Colabuono et al 2009
Blue petrel (Halobaena caerulea)	27	100	/		pellets	New Zealand	Reed 1981
Halobaena caerulea	74	85.1	/		pellets	Southern Ocean	Ryan 1987
Halobaena caerulea	62	56	/	3 – 6 mm	fragments, pellets	Antarctica	Ainley et al. 1990
Prions (Pachyptila spp.)	/	/	/		pellets	Gough Island, South Atlantic	Bourne and Imber 1982
Salvin's prion (Pachyptila salvini)	663	20	/	2.5 – 3.5 mm	pellets	Wellington, New Zealand	Harper and Fowler 1987
Pachyptila salvini	31	51.6	/		pellets	Southern Ocean	Ryan 1987

Species	n	Percentage with plastic (%)	Mean number of particles per individual	Mean size ingested ± SD (minimum – maximum) (mm)	Type of plastic	Location	Source
Thin-billed prion (Pachyptila belcheri)	152	6.6	/	2.5 – 3.5mm	pellets	Wellington, New Zealand	Harper and Fowler 1987
Pachyptila belcheri	32	68.7	/		pellets	Southern Ocean	Ryan 1987
Broad-billed prion (Pachyptila vittata)	31	39	0.6	max mass: 0.066g	pellets	Gough Island, South Atlantic	Furness 1985a
Pachyptila vittata	310	16.5	/	2.5 – 3.5 mm	pellets	Wellington, New Zealand	Harper and Fowler 1987
Pachyptila vittata	137	20.4	/		pellets	Southern Ocean	Ryan 1987
Pachyptila vittata	69	10	/	3 – 6 mm	fragments, pellets	Antarctica	Ainley et al. 1990
Pachyptila vittata	149	/	1987 – 1989 B1.73 ± 3.58		pellets	Southern Ocean	Ryan 2008
Pachyptila vittata	86	/	1999 B2.93 ± 3.80		pellets	Southern Ocean	Ryan 2008
Pachyptila vittata	95	/	2004 B2.66 ± 5.34		pellets	Southern Ocean	Ryan 2008
Antarctic prion (Pachyptila desolata)	35	14.3	/	2.5 – 3.5 mm	pellets	Wellington, New Zealand	Harper and Fowler 1987
Pachyptila desolata	88	47.7	/		pellets	Southern Ocean	Ryan 1987
Pachyptila desolata	2	100	1	6 – 8.1 mm		Heard Island, Australia	Auman et al. 2004
Fairy prion (Pachyptila turtur)	105	96.2	/	2.5 – 3.5 mm	pellets	Wellington, New Zealand	Harper and Fowler 1987
Snow petrel (Pagodroma nivea)	363	1	/	3 – 6 mm	fragments, pellets	Antarctica	Ainley et al. 1990
White-chinned petrel (Procellaria aequinoctialis)	193	/	1983-1985 B1.66 (± 3.04)		pellets	Southern Ocean	Ryan 1987, 2008
Procellaria aequinoctialis	526	/	2005 – 2006 B1.39 (± 3.25)		pellets	Southern Ocean	Ryan 2008
Procellaria aequinoctialis	41	/	/	<10		Rio Grande do Sul, Brazil	Colabuono et al. 2009

Species	n	Percentage with plastic (%)	Mean number of particles per individual	Mean size ingested ± SD (minimum – maximum) (mm)	Type of plastic	Location	Source
Procellaria aequinoctialis	34	44	/	<10		Rio Grande do Sul, Brazil	Colabuono et al. 2010
Spectacled petrel (Procellaria conspicillata)	3	33	/	<10		Rio Grande do Sul, Brazil	Colabuono et al. 2010
Procellaria conspicillata	9	/	/	<10		Rio Grande do Sul, Brazil	Colabuono et al. 2009
Tahiti petrel (Pseudobulweria rostrata)	121	<1	1		fragments	Tropical, North Pacific	Spear et al. 1995
Atlantic petrel (Pterodroma incerta)	13	8	0.1	max mass: 0.0053 g	pellets	Gough Island, South Atlantic	Furness 1985a
Pterodroma incerta	20	5	/		pellets	Southern Ocean	Ryan 1987
Great-winged petrel (Pterodroma macroptera)	13	7.6	/		pellets	Southern Ocean	Ryan 1987
Soft-plumaged petrel (Pterodroma mollis)	29	20.6	/		pellets	Southern Ocean	Ryan 1987
Pterodroma mollis	18	6	0.1	0.014 g	pellets	Gough Island, South Atlantic	Furness 1985a
Juan Fernández petrel (Pterodroma externa)	183	<1	1	3 – 5 mm	pellets	Offshore, North Pacific	Spear et al. 1995
White-necked petrel (Pterodroma cervicalis)	12	8.3	5	3 – 4 mm	fragments	Offshore, North Pacific	Spear et al. 1995
Pycroft's petrel (Pterodroma pycrofti)	5	40	2.5 (± 0.7)	3 – 5 mm	fragments and pellets	Offshore, North Pacific	Spear et al. 1995
White-winged petrel (Pterodroma leucoptera)	110	11.8	2.2 (± 3.0)	2 – 5 mm	fragments	Offshore, North Pacific	Spear et al. 1995
Collared petrel (Pterodroma brevipes)	3	66.7	1	2 – 5 mm		Offshore, North Pacific	Spear et al. 1995
Black-winged petrel (Pterodroma nigripenni)	66	4.5	3.0 (± 3.5)	3 – 5 mm	fragments	Offshore, North Pacific	Spear et al. 1995
Stejneger's petrel (Pterodroma longirostris)	46	73.9	6.8 (± 8.6)	2 – 5 mm	fragments and pellets	Offshore, North Pacific	Spear et al. 1995
Audubon's shearwater (Puffinus Ilherminieri)	119	5	Stomach = 1 Gizzard = 4.4		beads	North Carolina, USA	Moser and Lee 1992

Species	n	Percentage with plastic (%)	Mean number of particles per individual	Mean size ingested ± SD (minimum – maximum) (mm)	Type of plastic	Location	Source
Little shearwater (Puffinus assimilis)	13	8	0.8	max mass: 0.12 g	pellets	Gough Island, South Atlantic	Furness 1985a
Buller's shearwater (Puffinus bulleri)	3	100	8.5 (± 8.6)	2 – 8 mm	fragments and pellets	Tropical, North Pacific	Spear et al. 1995
Pink-footed shearwater (Puffinus creatopus)	5	20	2.2	1 – 4 mm	pellets	California, USA	Baltz and Morejohn 1976
Great shearwater (Puffinus gravis)	24	100	/		beads	Briar Island, Nova Scotia	Brown et al. 1981
Puffinus gravis	13	85	12.2	max mass: 1.13 g	pellets	Gough Island, South Atlantic	Furness 1985a
Puffinus gravis	55	63.6	Stomach = 1 Gizzard = 13		beads	North Carolina, USA	Moser and Lee 1992
Puffinus gravis	50	66	1983-1985 B16.5(± 19.0)		pellets	Southern Ocean	Ryan 1987, 2008
Puffinus gravis	53	/	2005-2006 B11.8 (± 18.9)		pellets	Southern Ocean	Ryan 2008
Puffinus gravis	19	89	/	<10 mm		Rio Grande do Sul, Brazil	Colabuono et al. 2009
Puffinus gravis	6	100	/	<3.2 – 5.3 mm	pellets	Rio Grande do Sul, Brazil	Colabuono et al. 2010
Puffinus gravis	84	88	11.8 (± 16.9)		fragments and pellets	Nova Scotia, Canada	Bond et al. 2014
Sooty shearwater (Puffinus griseus)	21	43	5.05	1 – 4 mm	pellets	California, USA	Baltz and Morejohn 1976
Puffinus griseus	5	100	/	Beads	beads	Briar Island, Nova Scotia, Canada	Brown et al. 1981
Puffinus griseus	36	58.3	11.4 (± 12.2)	3 – 20 mm	fragments and pellets	Tropical, North Pacific	Spear et al. 1995
Puffinus griseus	218	88.5	/		pellets	Offshore, North Pacific	Ogi et al. 1990

Species	n	Percentage with plastic (%)	Mean number of particles per individual	Mean size ingested ± SD (minimum – maximum) (mm)	Type of plastic	Location	Source
Puffinus griseus	20	75	3.4		pellets	Offshore eastern North Pacific	Blight and Burger 1997
Puffinus griseus	50	72	2.48 (± 2.7)		fragments and pellets	Nova Scotia, Canada	Bond et al. 2014
Balearic shearwater (Puffinus mauretanicus)	46	70	2.5 (± 2.9)	3.5 (± 10.5A)		Catalan coast, Mediterranean	Codina-García et al. 2013
Christmas shearwater (Puffinus nativitatis)	5	40	1	3 – 5 mm	fragments and pellets	Tropical, North Pacific	Spear et al. 1995
Wedge-tailed shearwater (Puffinus pacificus)	23	4	2.5 (± 2.1)		fragments and pellets	Tropical, North Pacific	Spear et al. 1995
dark phase	62	24.2	3.5 (± 2.7)				
Puffinus pacificus	20	60	max: 11	pellets 2 – 4 mm	pellets	Hawaii	Fry et al. 1987
Manx shearwater (Puffinus puffinus)	10	30	0.4		pellets	Rhum, UK	Furness 1985b
Puffinus puffinus	25	60	/	<10 mm		Rio Grande do Sul, Brazil	Colabuono et al. 2009
Puffinus puffinus	6	17	/		fragments	Rio Grande do Sul, Brazil	Colabuono et al. 2010
Short-tailed shearwater (Puffinus tenuirostris)	6	100	19.8	1 – 4 mm	pellets	California, USA	Baltz and Morejohn 1976
Puffinus tenuirostris	324	81.8	/		pellets	Offshore, North Pacific	Ogi et al. 1990
Puffinus tenuirostris	330	83.9	5.8 (± 0.4*)	2 – 5 mm	pellets	Bering Sea, North Pacific	Vlietstra and Parga 2002
Puffinus tenuirostris	5	80	/		fragments and pellets	Alaska, USA	Robards et al. 1995
Puffinus tenuirostris	99	100	15.1 (± 13.2)	>2 mm		Offshore, North Pacific	Yamashita et al. 2011
Puffinus tenuirostris	129	67	Adults: 4.5 Juvenile: 7.1	0.97 – 80.8 mm	fragments	North Stradbroke Island, Australia	Acampora et al. 2013

Species	n	Percentage with plastic (%)	Mean number of particles per individual	Mean size ingested ± SD (minimum – maximum) (mm)	Type of plastic	Location	Source
Puffinus tenuirostris	12	100	27	>2 mm		Offshore, North Pacific	Tanaka et al. 2013
Yelkouan shearwater (Puffinus yelkouan)	31	71	4.9 (± 7.3)	4.0 (± 13.0 A)		Catalan coast, Mediterranean	Codina-García et al. 2013
Antarctic petrel (Thalassoica antarctica)	184	<1	/	fragments, pellets 3 – 6 mm		Antarctica	Ainley et al. 1990
Family Hydrobatidae							
White-bellied storm petrel (Fregetta grallaria)	13	38	1.2	pellets max mass: 0.042 g		Gough Island, UK	Furness 1985a
						South Atlantic	
Fregetta grallaria	296	<1	1	fragment		Offshore, North Pacific	Spear et al. 1995
Fregetta grallaria	318	/	1987 – 1989	pellets 33.3%		Southern Ocean	Ryan 2008
			b0.63 ± 1.13				
Fregetta grallaria	137	/	1999	pellets 20.9%		Southern Ocean	Ryan 2008
			b0.63 ± 1.37				
Fregetta grallaria	95	/	2004	pellets 16.2%		Southern Ocean	Ryan 2008
			B0.72 ± 1.87				
Grey-backed storm petrel (Garrodia nereis)	11	27	0.3	pellets: max mass: 0.010 g		Gough Island, UK	Furness 1985a
						South Atlantic	
Garrodia nereis	12	8.3	/	pellets		Southern Ocean	Ryan 1987
Fork-tailed storm petrel (Oceanodroma furcata)	/	/	/	<5 mm		Aleutian Islands, USA	Ohlendorf et al. 1978
Oceanodroma furcata	21	85.7	Max: 12	pellets 22%		Alaska, USA	Robards et al. 1995
Oceanodroma furcata	7	100	20.1	pellets 16%		Eastern North Pacific	Blight and Burger 1997

Species	n	Percentage with plastic (%)	Mean number of particles per individual	Mean size ingested ± SD (minimum – maximum) (mm)	Type of plastic	Location	Source
Leach's storm petrel (Oceanodroma leucorhoa)	15	40	1.66 (± 1.2)	2 – 5 mm		Newfoundland, Canada	Rothstein 1973
Oceanodroma leucorhoa	17	58.8	2.9	pellets		St. Kilda, Scotland, UK	Furness 1985b
Oceanodroma leucorhoa	354	19.8	3.5 (± 2.6)	fragments, pellets 2 – 5 mm		Offshore, North Pacific	Spear et al. 1995
Oceanodroma leucorhoa	64	48.4	Max: 13	monofilament line, fragments, pellets		Alaska, USA	Robards et al. 1995
Wilson's storm petrel (Oceanites oceanicus)	20	75	4.4	2.9 mm		Ardery Island, Antarctica	van Franeker and Bell 1988
Oceanites oceanicus	91	19	/	fragments, pellets 3 – 6 mm		Antarctica	Ainley et al. 1990
Oceanites oceanicus	133	38.3	Stomach = 1.4 Gizzard = 5.4	26% beads		North Carolina, USA	Moser and Lee 1992
White-faced storm petrel (Pelagodroma marina)	19	84	11.7	pellets max mass: 0.34 g		Gough Island, UK South Atlantic	Furness 1985a
Pelagodroma marina	15	73.3	13.2 ± 9.5	pellets 2 – 5 mm		Offshore, North Pacific	Spear et al. 1985
Pelagodroma marina	24	20.8	/	pellets 41%		Southern Hemisphere	Ryan 1987
Pelagodroma marina	253		1987 – 1989 B3.98 ± 5.45	pellets 69.6%		Southern Ocean	Ryan 2008
Pelagodroma marina	86	/	1999 B4.06 ± 5.93	pellets 37.5%		Southern Ocean	Ryan 2008
Pelagodroma marina	5	/	2004 B2.52 ± 4.43	pellets 13.5%		Southern Ocean	Ryan 2008

Species	n	Percentage with plastic (%)	Mean number of particles per individual	Mean size ingested ± SD (minimum – maximum) (mm)	Type of plastic	Location	Source
Family Diomedeidae							
Sooty albatross (Phoebetria fusca)	73	42.7	/	pellets 34%		Southern Ocean	Ryan 1987
Laysan albatross (Phoebastria immutabilis)	/	52	/	pellets 2 – 5 mm		Hawaiian Islands, USA	Sileo et al. 1990
Black-footed albatross (Phoebastria nigripes)	/	12	/	pellets 2 – 5 mm		Hawaiian Islands, USA	Sileo et al. 1990
Phoebastria nigripes (As Diomedea nigripes)	3	100	5.3	pellets 50%		Offshore, eastern North Pacific	Blight and Burger 1997
Black-browed albatross (Thalassarche melanophris)	2	100	3	pellets 50%		Rio Grande do Sul, Brazil	Tourinho et al. 2010
Order Charadriiformes							
Family Laridae							
Audouin's gull (Larus audouinii)	15	13	49.3 (± 77.7)	2.5 (± 5.0*)		Catalan coast, Mediterranean	Codina-García et al. 2013
Glaucous-winged gull (Larus glaucescens)	589 boluses	12.2	/	<10 mm		Protection Island, USA	Lindborg et al. 2012
Heermann's Gull (Larus heermanni)	15	7	1	pellets 1 – 4 mm		California, USA	Baltz and Morejohn 1976
Mediterranean gull (Larus melanocephalus)	4	25	3.7 (± 7.5)	3.0 (± 5.0*)		Catalan coast, Mediterranean	Codina-García et al. 2013
Yellow-legged gull) (Larus michahellis)	12	33	0.9 (± 1.5)	2.0 (± 8.0*)		Catalan coast, Mediterranean	Codina-García et al. 2013
Red-legged kittiwake (Rissa brevirostris)	15	26.7	/	pellets: mean 5.87 mm		Alaska, USA	Robards et al. 1995
Black-legged kittiwake (Rissa tridactyla)	8	8	4	pellets 1 – 4 mm		California, USA	Baltz and Morejohn 1976
Rissa tridactyla	256	7.8	Max: 15	pellets		Alaska, USA	Robards et al. 1995
Rissa tridactyla	4	50	1.2 (± 1.9)	3.0 (± 5.0*)		Catalan coast, Mediterranean	Codina-García et al. 2013

Species	n	Percentage with plastic (%)	Mean number of particles per individual	Mean size ingested ± SD (minimum – maximum) (mm)	Type of plastic	Location	Source
Family Alcidae							
Parakeet auklet (Aethia psittacula)	/	/	/	<5 mm		Aleutians Islands, USA	Ohlendorf et al. 1978
Aethia psittacula	208	93.8	17.1	pellets 4.08 mm		Alaska, USA	Robards et al. 1995
Tufted puffin (Fratercula cirrhata)	489	24.5	Max: 51	pellets 4.10 mm		Alaska, USA	Robards et al. 1995
Fratercula cirrhata	9	89	3.3	pellets		Offshore, North Pacific	Blight and Burger 1997
Horned puffin (Fratercula corniculata)	/	/	/	<5 mm		Aleutian Islands, USA	Ohlendorf et al. 1978
Fratercula corniculata	120	36.7	Max: 14	pellets 5.03 mm		Alaska, USA	Robards et al. 1995
Fratercula corniculata	2	50	1.5	pellets		Offshore, North Pacific	Blight and Burger 1997
Common murre (Uria aalge)	1	100	2011 – 2012 1	6.6 (± 2.2)		Newfoundland, Canada	Bond et al. 2013
Thick-billed murre (Uria lomvia)	186	11	0.2 (± 0.8)	4.5 (± 3.8)		Canadian Arctic	Provencher et al. 2010
Uria lomvia	3	100	2011 – 2012 1	6.6 (± 2.2)		Newfoundland, Canada	Bond et al. 2013
Uria lomvia	1249	7.7	1985 – 1986 0.14 (± 0.7*)	10.1 (± 7.4)		Newfoundland, Canada	Bond et al. 2013
Family Stercorariidae							
Brown skua (Stercorarius antarcticus) (As Catharacta antarcticus)	494	22.7	/	pellets 67%		Southern Ocean	Ryan 1987
Tristan skua (Stercorarius hamiltoni) (As Catharacta hamiltoni)	11	9	0.3 Max: 3	pellets max mass: 0.064 g		Gough Island, UK South Atlantic	Furness 1985a

Species	n	Percentage with plastic (%)	Mean number of particles per individual	Mean size ingested ± SD (minimum – maximum) (mm)	Type of plastic	Location	Source
Long-tailed skua (Stercorarius longicaudus)	2	50	5	fragments, pellets		Eastern North Pacific	Spear et al. 1995
Arctic skua (Stercorarius parasiticus)	2	50	/	pellets 50%		Southern Ocean	Ryan 1987
Family Scolopacidae							
Grey phalarope (Phalaropus fulicarius)	20	100	Max: 36	beads 1.7 – 4.4 mm		California, USA	Bond 1971
Phalaropus fulicarius	7	85.7	5.7	pellets		California, USA	Connors and Smith 1982
Phalaropus fulicarius	2	50	/	pellets		Southern Ocean	Ryan 1987
Phalaropus fulicarius	55	69.1	Stomach = 1 Gizzard = 6.7	beads 16.7%		North Carolina, USA	Moser and Lee 1992
Red-necked phalarope (Phalaropus lobatus)	36	19.4	Stomach = 0 Gizzard = 3.7	beads 16.7%		North Carolina, USA	Moser and Lee 1992
Family Sternidae							
Sooty tern (Onychoprion fuscatus)	64	1.6	2	pellets 4 mm		Offshore, eastern North Pacific	Spear et al. 1995
White tern (Gygis alba)	8	12.5	5	fragments 3 – 4 mm		Offshore, eastern North Pacific	Spear et al. 1995
Order Suliformes							
Family Phalacrocoracidae							
Macquarie Shag (Phalacrocorax atriceps purpurascens)	C64	7.8	1 per bolus	polystyrene spheres		Macquarie Island, Australia	Slip et al. 1990

Table AIII.4 Diverse microbial assemblage found on microplastic debris

Proportion of different microbial groups in each sample (i.e. sum = 1)

Taxonomy	Polymer type					
	Polypropylene			Polyethylene		
	Sampling date					
	20/5/2012	22/5/2012	21/6/2010	20/5/2012	22/5/2012	7/7/2010
Bacteria; Acidobacteria; Acidobacteria; Acidobacteriales; Acidobacteriaceae; Acidobacterium; species	0	0	0.000156789	0	0	0
Bacteria; Acidobacteria; Acidobacteria; Acidobacteriales; Acidobacteriaceae; Chloroacidobacterium; species	0	0	0.000313578	0	0	0
Bacteria; Acidobacteria; Acidobacteria_Gp22; Unassigned; Unassigned; Gp22; species	0	0.000753509	0	0	0	0
Bacteria; Acidobacteria; Holophagae; Acanthopleuribacterales; Acanthopleuribacteraceae; Acanthopleuribacter	0	0.000565131	0	0	0	0
Bacteria; Actinobacteria; Actinobacteria; Acidimicrobiales; Iamiaceae; Iamia; species	0.001138822	0.000753509	0.000156789	0.001467505	0.001617353	0.000461201
Bacteria; Actinobacteria; Actinobacteria; Acidimicrobiales; family; genus; species	0	9.41886E-05	0	0.000209644	0	0.007494523
Bacteria; Actinobacteria; Actinobacteria; Actinomycetales; Corynebacteriaceae; Corynebacterium; species	0	0	0	0	0	0.000230601
Bacteria; Actinobacteria; Actinobacteria; Actinomycetales; Corynebacteriaceae; genus; species	0	0	0.000940734	0	0	0.000345901
Bacteria; Actinobacteria; Actinobacteria; Actinomycetales; Intrasporangiaceae; Serinicoccus; species	0	0	0	0	0	0.000230601
Bacteria; Actinobacteria; Actinobacteria; Actinomycetales; Microbacteriaceae; Agrococcus; jenensis	0	0	0	0.000524109	0	0
Bacteria; Actinobacteria; Actinobacteria; Actinomycetales; Nocardioidaceae; Marmoricola; species	0	0	0	0	0	0.0001153
Bacteria; Actinobacteria; Actinobacteria; Actinomycetales; Propionibacteriaceae; Propionibacterium; species	0.000227764	9.41886E-05	0.002038257	0	0	0.00311311
Bacteria; Actinobacteria; Actinobacteria; Nitriliruptorales; Nitriliruptoraceae; Nitriliruptor; species	0	0	0.000156789	0	0	0
Bacteria; Bacteroidetes; Flavobacteria; Flavobacteriales; Cryomorphaceae; Owenweeksia; species	0	0.000282566	0.000783945	0	0	0.000345901

Taxonomy	Polymer type					
	Polypropylene			Polyethylene		
	Sampling date					
	20/5/2012	22/5/2012	21/6/2010	20/5/2012	22/5/2012	7/7/2010
Bacteria; Bacteroidetes; Flavobacteria; Flavobacteriales; Cryomorphaceae; genus; species	0	0	0.000313578	0	0	0.000691802
Bacteria; Bacteroidetes; Flavobacteria; Flavobacteriales; Flavobacteriaceae; Aequorivita; species	0	0	0.000156789	0	0	0
Bacteria; Bacteroidetes; Flavobacteria; Flavobacteriales; Flavobacteriaceae; Algibacter; lectus	0	0	0.000313578	0	0	0
Bacteria; Bacteroidetes; Flavobacteria; Flavobacteriales; Flavobacteriaceae; Aquimarina; species	0.000341647	9.41886E-05	0.000470367	0	0	0.000691802
Bacteria; Bacteroidetes; Flavobacteria; Flavobacteriales; Flavobacteriaceae; Bizionia; species	0	0	0.000313578	0	0	0.000345901
Bacteria; Bacteroidetes; Flavobacteria; Flavobacteriales; Flavobacteriaceae; Cellulophaga; species	0	0	0.000313578	0	0	0
Bacteria; Bacteroidetes; Flavobacteria; Flavobacteriales; Flavobacteriaceae; Chryseobacterium; taichungense	0	0	0	0	0	0.0001153
Bacteria; Bacteroidetes; Flavobacteria; Flavobacteriales; Flavobacteriaceae; Croceibacter; atlanticus	0	0	0	0.000628931	0	0
Bacteria; Bacteroidetes; Flavobacteria; Flavobacteriales; Flavobacteriaceae; Croceitalea; eckloniae	0	0	0	0	0	0.000230601
Bacteria; Bacteroidetes; Flavobacteria; Flavobacteriales; Flavobacteriaceae; Croceitalea; species	0	0	0.000940734	0	0	0.002882509
Bacteria; Bacteroidetes; Flavobacteria; Flavobacteriales; Flavobacteriaceae; Flagellimonas; species	0	0	0	0.000104822	0	0
Bacteria; Bacteroidetes; Flavobacteria; Flavobacteriales; Flavobacteriaceae; Formosa; species	0	0	0.000156789	0	0	0
Bacteria; Bacteroidetes; Flavobacteria; Flavobacteriales; Flavobacteriaceae; Gramella; portivictoriae	0	9.41886E-05	0	0	0	0
Bacteria; Bacteroidetes; Flavobacteria; Flavobacteriales; Flavobacteriaceae; Kordia; algicida	0	0	0.000156789	0	0	0
Bacteria; Bacteroidetes; Flavobacteria; Flavobacteriales; Flavobacteriaceae; Kordia; species	0	0	0.000313578	0	0	0
Bacteria; Bacteroidetes; Flavobacteria; Flavobacteriales; Flavobacteriaceae; Kriegella; aquimaris	0	0	0	0	0	0.0001153

Taxonomy	Polymer type					
	Polypropylene			Polyethylene		
	Sampling date					
	20/5/2012	22/5/2012	21/6/2010	20/5/2012	22/5/2012	7/7/2010
Bacteria; Bacteroidetes; Flavobacteria; Flavobacteriales; Flavobacteriaceae; Lacinutrix; species	0	0	0.000156789	0	0	0
Bacteria; Bacteroidetes; Flavobacteria; Flavobacteriales; Flavobacteriaceae; Leeuwenhoekiella; species	0	0	0.000156789	0	0	0
Bacteria; Bacteroidetes; Flavobacteria; Flavobacteriales; Flavobacteriaceae; Lutibacter; species	0	0	0.000156789	0	0	0
Bacteria; Bacteroidetes; Flavobacteria; Flavobacteriales; Flavobacteriaceae; Marixanthomonas; ophiurae	0	0	0.001411101	0.000209644	0	0.000461201
Bacteria; Bacteroidetes; Flavobacteria; Flavobacteriales; Flavobacteriaceae; Marixanthomonas; species	0	0	0.002822201	0	0	0.001729505
Bacteria; Bacteroidetes; Flavobacteria; Flavobacteriales; Flavobacteriaceae; Mesoflavibacter; species	0	0	0.000313578	0	0	0
Bacteria; Bacteroidetes; Flavobacteria; Flavobacteriales; Flavobacteriaceae; Mesonia; species	0	0	0.002665412	0	0	0.001383604
Bacteria; Bacteroidetes; Flavobacteria; Flavobacteriales; Flavobacteriaceae; Muricauda; species	0	0.014316662	0	0.004402516	0.000665969	0.000461201
Bacteria; Bacteroidetes; Flavobacteria; Flavobacteriales; Flavobacteriaceae; Pseudozobellia; thermophila	0	0	0	0	0	0.000230601
Bacteria; Bacteroidetes; Flavobacteria; Flavobacteriales; Flavobacteriaceae; Psychroserpens; species	0	0	0	0	0	0.000345901
Bacteria; Bacteroidetes; Flavobacteria; Flavobacteriales; Flavobacteriaceae; Salegentibacter; holothuriorum	0	0	0.000156789	0	0	0
Bacteria; Bacteroidetes; Flavobacteria; Flavobacteriales; Flavobacteriaceae; Salinimicrobium; species	0	0	0.000156789	0	0	0
Bacteria; Bacteroidetes; Flavobacteria; Flavobacteriales; Flavobacteriaceae; Sediminibacter; species	0	0	0.000156789	0	0	0
Bacteria; Bacteroidetes; Flavobacteria; Flavobacteriales; Flavobacteriaceae; Stenothermobacter; species	0	0.000188377	0	0	0	0
Bacteria; Bacteroidetes; Flavobacteria; Flavobacteriales; Flavobacteriaceae; Stenothermobacter; spongiae	0	0	0	0	0	0.000345901
Bacteria; Bacteroidetes; Flavobacteria; Flavobacteriales; Flavobacteriaceae; Tenacibaculum; aestuarii	0	0	0	0	9.51384E-05	0

Taxonomy	Polymer type					
	Polypropylene			Polyethylene		
	Sampling date					
	20/5/2012	22/5/2012	21/6/2010	20/5/2012	22/5/2012	7/7/2010
Bacteria; Bacteroidetes; Flavobacteria; Flavobacteriales; Flavobacteriaceae; Tenacibaculum; amylolyticum	0	0	0.000470367	0.000314465	0	0
Bacteria; Bacteroidetes; Flavobacteria; Flavobacteriales; Flavobacteriaceae; Tenacibaculum; mesophilum	0	0	0.000156789	0	0	0
Bacteria; Bacteroidetes; Flavobacteria; Flavobacteriales; Flavobacteriaceae; Tenacibaculum; species	0.000227764	0.015352736	0.011445594	0.001991614	0.030634573	0.003574311
Bacteria; Bacteroidetes; Flavobacteria; Flavobacteriales; Flavobacteriaceae; Ulvibacter; species	0	0	0.001411101	0	0	0.000345901
Bacteria; Bacteroidetes; Flavobacteria; Flavobacteriales; Flavobacteriaceae; Vitellibacter; species	0	0	0.000470367	0	0	0.000230601
Bacteria; Bacteroidetes; Flavobacteria; Flavobacteriales; Flavobacteriaceae; Winogradskyella; species	0.000455529	0.000470943	0.015835685	0	0	0.021215266
Bacteria; Bacteroidetes; Flavobacteria; Flavobacteriales; Flavobacteriaceae; Yeosuana; species	0	0	0.000470367	0	0	0
Bacteria; Bacteroidetes; Flavobacteria; Flavobacteriales; Flavobacteriaceae; Croceitalea; species	0	0	0.000940734	0	0	0.002882509
Bacteria; Bacteroidetes; Flavobacteria; Flavobacteriales; Flavobacteriaceae; Flagellimonas; species	0	0	0	0.000104822	0	0
Bacteria; Bacteroidetes; Flavobacteria; Flavobacteriales; Flavobacteriaceae; Formosa; species	0	0	0.000156789	0	0	0
Bacteria; Bacteroidetes; Flavobacteria; Flavobacteriales; Flavobacteriaceae; Gramella; portivictoriae	0	9.41886E-05	0	0	0	0
Bacteria; Bacteroidetes; Flavobacteria; Flavobacteriales; Flavobacteriaceae; Kordia; algicida	0	0	0.000156789	0	0	0
Bacteria; Bacteroidetes; Flavobacteria; Flavobacteriales; Flavobacteriaceae; Kordia; species	0	0	0.000313578	0	0	0
Bacteria; Bacteroidetes; Flavobacteria; Flavobacteriales; Flavobacteriaceae; Kriegella; aquimaris	0	0	0	0	0	0.0001153
Bacteria; Bacteroidetes; Flavobacteria; Flavobacteriales; Flavobacteriaceae; genus; species	0.000113882	0.000188377	0.005330825	0	0.000570831	0.026980284
Bacteria; Bacteroidetes; Flavobacteria; Flavobacteriales; family; genus; species	0	0	0	0	0	0.000922403

Taxonomy	Polymer type					
	Polypropylene			Polyethylene		
	Sampling date					
	20/5/2012	22/5/2012	21/6/2010	20/5/2012	22/5/2012	7/7/2010
Bacteria; Bacteroidetes; Sphingobacteria; Sphingobacteriales; Chitinophagaceae; Balneola; species	0	0	0.000156789	0	0	0.002536608
Bacteria; Bacteroidetes; Sphingobacteria; Sphingobacteriales; Chitinophagaceae; Flavisolibacter; species	0	0	0	0.000104822	0	0.0001153
Bacteria; Bacteroidetes; Sphingobacteria; Sphingobacteriales; Chitinophagaceae; Lacibacter; species	0	0	0	0	0	0.000230601
Bacteria; Bacteroidetes; Sphingobacteria; Sphingobacteriales; Chitinophagaceae; Niastella; species	0	0	0	0	0	0.000230601
Bacteria; Bacteroidetes; Sphingobacteria; Sphingobacteriales; Chitinophagaceae; Sediminibacterium; species	0	0	0	0	0	0.029516892
Bacteria; Bacteroidetes; Sphingobacteria; Sphingobacteriales; Chitinophagaceae; genus; species	0.035645143	0.072148441	0.035591094	0.176939203	0.063552469	0.021561167
Bacteria; Bacteroidetes; Sphingobacteria; Sphingobacteriales; Cytophagaceae; Flexibacter; litoralis	0	0	0.000470367	0	0	0
Bacteria; Bacteroidetes; Sphingobacteria; Sphingobacteriales; Cytophagaceae; Flexibacter; species	0.000797176	9.41886E-05	0.000156789	0	9.51384E-05	0.003920212
Bacteria; Bacteroidetes; Sphingobacteria; Sphingobacteriales; Cytophagaceae; Microscilla; marina	0.005238583	9.41886E-05	0.000783945	0.000104822	9.51384E-05	0.0001153
Bacteria; Bacteroidetes; Sphingobacteria; Sphingobacteriales; Cytophagaceae; Microscilla; sericea	0.000227764	0.000188377	0.004390091	0.001362683	0.000285415	0.003459011
Bacteria; Bacteroidetes; Sphingobacteria; Sphingobacteriales; Cytophagaceae; Microscilla; species	0.000227764	0.0065932	0.000783945	0	0.000285415	0.000230601
Bacteria; Bacteroidetes; Sphingobacteria; Sphingobacteriales; Flammeovirgaceae; Amoebophilus; species	0.0121854	0.003861731	0.006898714	0.000419287	0	0.037126715
Bacteria; Bacteroidetes; Sphingobacteria; Sphingobacteriales; Flammeovirgaceae; Fabibacter; species	0	0	0.004860458	0	0	0.001383604
Bacteria; Bacteroidetes; Sphingobacteria; Sphingobacteriales; Flammeovirgaceae; Flexithrix; species	0.010249402	0	0.001411101	0	0	0.000230601
Bacteria; Bacteroidetes; Sphingobacteria; Sphingobacteriales; Flammeovirgaceae; Fulvivirga; species	0.002163763	0.002072148	0.027281279	0	0	0.001268304
Bacteria; Bacteroidetes; Sphingobacteria; Sphingobacteriales; Flammeovirgaceae; Marinoscillum; species	0.045780663	0.032024112	0.00297899	0.001886792	0.002473599	0.000576502

Taxonomy	Polymer type					
	Polypropylene			Polyethylene		
	Sampling date					
	20/5/2012	22/5/2012	21/6/2010	20/5/2012	22/5/2012	7/7/2010
Bacteria; Bacteroidetes; Sphingobacteria; Sphingobacteriales; Flammeovirgaceae; Reichenbachiella; agariperforan	0	0	0.000313578	0	0	0.0001153
Bacteria; Bacteroidetes; Sphingobacteria; Sphingobacteriales; Flammeovirgaceae; Reichenbachiella; species	0.000455529	0	0.000470367	0	0	0.0001153
Bacteria; Bacteroidetes; Sphingobacteria; Sphingobacteriales; Flammeovirgaceae; Roseivirga; ehrenbergii	0	9.41886E-05	0	0	0	0
Bacteria; Bacteroidetes; Sphingobacteria; Sphingobacteriales; Flammeovirgaceae; Roseivirga; species	0	0.000470943	0.000940734	0	0	0
Bacteria; Bacteroidetes; Sphingobacteria; Sphingobacteriales; Flammeovirgaceae; genus; species	0.016399043	0.001883771	0.005957981	0.000209644	0.000951384	0.000230601
Bacteria; Bacteroidetes; Sphingobacteria; Sphingobacteriales; Rhodothermaceae; Salisaeta; species	0	0	0.000313578	0	0	0
Bacteria; Bacteroidetes; Sphingobacteria; Sphingobacteriales; Rhodothermaceae; genus; species	0	0	0.002351834	0	0	0.000691802
Bacteria; Bacteroidetes; Sphingobacteria; Sphingobacteriales; Saprospiraceae; Aureispira; marina	0	0	0.000783945	0	0	0
Bacteria; Bacteroidetes; Sphingobacteria; Sphingobacteriales; Saprospiraceae; Aureispira; species	0	0	0.007212292	0.003039832	9.51384E-05	0.001614205
Bacteria; Bacteroidetes; Sphingobacteria; Sphingobacteriales; Saprospiraceae; Haliscomenobacter; species	0.00911058	0.003108223	0.006271558	0.011844864	0.01693464	0.008416926
Bacteria; Bacteroidetes; Sphingobacteria; Sphingobacteriales; Saprospiraceae; Lewinella; agarilytica	0	0.00065932	0.005017247	0.000314465	0	0.000807103
Bacteria; Bacteroidetes; Sphingobacteria; Sphingobacteriales; Saprospiraceae; Lewinella; antarctica	0	0	0	0.000104822	0.000380554	0
Bacteria; Bacteroidetes; Sphingobacteria; Sphingobacteriales; Saprospiraceae; Lewinella; cohaerens	0	0	0.002822201	0	9.51384E-05	0.000345901
Bacteria; Bacteroidetes; Sphingobacteria; Sphingobacteriales; Saprospiraceae; Lewinella; marina	0	0	0	0	0	0.000230601
Bacteria; Bacteroidetes; Sphingobacteria; Sphingobacteriales; Saprospiraceae; Lewinella; nigricans	0.000113882	0	0	0	0	0.0001153
Bacteria; Bacteroidetes; Sphingobacteria; Sphingobacteriales; Saprospiraceae; Lewinella; persicus	0	0.000470943	0.001097523	0	0.000285415	0.000345901

Taxonomy	Polymer type					
	Polypropylene			Polyethylene		
	Sampling date					
	20/5/2012	22/5/2012	21/6/2010	20/5/2012	22/5/2012	7/7/2010
Bacteria; Bacteroidetes; Sphingobacteria; Sphingobacteriales; Saprospiraceae; Lewinella; species	0.022207038	0.022887821	0.030730637	0.018763103	0.030729712	0.01602675
Bacteria; Bacteroidetes; Sphingobacteria; Sphingobacteriales; Saprospiraceae; Saprospira; species	0	0	0.00156789	0.000314465	0	0
Bacteria; Bacteroidetes; Sphingobacteria; Sphingobacteriales; Saprospiraceae; genus; species	0.029039973	0.008476971	0.011759172	0.069916143	0.071734374	0.008647527
Bacteria; Bacteroidetes; Sphingobacteria; Sphingobacteriales; family; genus; species	0.000455529	0.000188377	0	0	0	0.000230601
Bacteria; Bacteroidetes; class; order; family; genus; species	0.000455529	0	0	0	0	0
Bacteria; Chlamydiae; Chlamydiae; Chlamydiales; Parachlamydiaceae; Neochlamydia; species	0	0	0	0	0	0.000576502
Bacteria; Chlamydiae; Chlamydiae; Chlamydiales; Parachlamydiaceae; Parachlamydia; species	0.001594351	0	0	0	0	0.000230601
Bacteria; Chlamydiae; Chlamydiae; Chlamydiales; Parachlamydiaceae; Protochlamydia; species	0	0	0	0	0	0.0001153
Bacteria; Chlamydiae; Chlamydiae; Chlamydiales; Simkaniaceae; Fritschea; species	0.000911058	0.000188377	0	0	0	0
Bacteria; Chlorobi; Chlorobia; Chlorobiales; family; genus; species	0	9.41886E-05	0	0	0.000190277	0
Bacteria; Chloroflexi; Anaerolineae; Anaerolineales; Anaerolinaceae; Longilinea; species	0	0	0	0	0	0.000461201
Bacteria; Chloroflexi; Anaerolineae; Anaerolineales; Anaerolinaceae; genus; species	0.06491288	0.170198738	0.000940734	0.013207547	0.006183998	0.000922403
Bacteria; Chloroflexi; Caldilineae; Caldilineales; Caldilineaceae; Caldilinea; species	0	0	0	0	0	0.000345901
Bacteria; Chloroflexi; Caldilineae; Caldilineales; family; genus; species	0.002847056	0.002354714	0.000940734	0.005555556	0.00542289	0.000461201
Bacteria; Chloroflexi; Dehalococcoidetes; Unassigned; Unassigned; Dehalogenimonas; species	0	0	0	0.000209644	0	0
Bacteria; Chloroflexi; Thermomicrobia; Sphaerobacterales; Sphaerobacteraceae; Sphaerobacter; species	0	0	0.000156789	0	0	0
Bacteria; Cyanobacteria; Cyanobacteria; SubsectionI; Unassigned; Prochlorococcus; species	0.000227764	0	0.000313578	0	0	0.002767209

Taxonomy	Polymer type					
	Polypropylene			Polyethylene		
	Sampling date					
	20/5/2012	22/5/2012	21/6/2010	20/5/2012	22/5/2012	7/7/2010
Bacteria; Cyanobacteria; Cyanobacteria; SubsectionI; Unassigned; Synechococcus; species	0.000341647	0.00065932	0.000783945	0	0.003900675	0.006802721
Bacteria; Cyanobacteria; Cyanobacteria; SubsectionI; family; genus; species	0.000227764	0.000376754	0.005330825	0.00293501	0.000761107	0
Bacteria; Cyanobacteria; Cyanobacteria; SubsectionII; SubgroupII; Pleurocapsa; species	0.000113882	0.010549119	0.002038257	0.00639413	0.011416611	0.004035513
Bacteria; Cyanobacteria; Cyanobacteria; SubsectionIII; Unassigned; Halomicronema; species	0.000113882	0	0	0	0	0
Bacteria; Cyanobacteria; Cyanobacteria; SubsectionIII; Unassigned; Leptolyngbya; species	0	0	0	0	0	0.0001153
Bacteria; Cyanobacteria; Cyanobacteria; SubsectionIII; Unassigned; Microcoleus; species	0	0	0.010348071	0	0	0
Bacteria; Cyanobacteria; Cyanobacteria; SubsectionIII; Unassigned; Oscillatoria; species	0	0	0.005644403	0	0	0.0001153
Bacteria; Cyanobacteria; Cyanobacteria; SubsectionIII; Unassigned; Phormidium; species	0.057510534	0.047188471	0.025086234	0.034696017	0.2490724	0.033667704
Bacteria; Cyanobacteria; Cyanobacteria; SubsectionIII; Unassigned; Prochlorothrix; species	0	0	0.000156789	0	0	0
Bacteria; Cyanobacteria; Cyanobacteria; SubsectionIII; Unassigned; Trichocoleus; species	0	0	0.000470367	0	0	0
Bacteria; Cyanobacteria; Cyanobacteria; SubsectionIII; Unassigned; Trichodesmium; species	0	0	0	0	0	0.000807103
Bacteria; Cyanobacteria; Cyanobacteria; SubsectionIII; family; genus; species	0.069809817	0.064707545	0.014111007	0.020125786	0.0217867	0.03562781
Bacteria; Cyanobacteria; Cyanobacteria; SubsectionIV; SubgroupII; Calothrix; species	0.000113882	0	0	0	0	0
Bacteria; Cyanobacteria; Cyanobacteria; SubsectionIV; SubgroupII; Rivularia; species	0.008085639	0.033248564	0.008623393	0.125786164	0.049091428	0.007263923
Bacteria; Cyanobacteria; Cyanobacteria; SubsectionIV; Unassigned; Cylindrospermum; species	0	0	0	0.000733753	0	0
Bacteria; Cyanobacteria; Cyanobacteria; order; family; genus; species	0	0	0	0	0	0.0001153

Taxonomy	Polymer type					
	Polypropylene			Polyethylene		
	Sampling date					
	20/5/2012	22/5/2012	21/6/2010	20/5/2012	22/5/2012	7/7/2010
Bacteria; Cyanobacteria; class; order; family; genus; species	0.000797176	0.00065932	0.000313578	0.005031447	0.001997907	0.001844806
Bacteria; Deferribacteres; Deferribacteres; Deferribacterales; SAR406; genus; species	0	0	0	0	0	0.002882509
Bacteria; Deinococcus-Thermus ; Deinococci; Deinococcales; Deinococcaceae; Deinococcus; species	0	0	0	0	0.000570831	0.000230601
Bacteria; Firmicutes; Bacilli; Bacillales; Bacillaceae; Bacillus; species	0	0	0	0	0	0.0001153
Bacteria; Firmicutes; Bacilli; Bacillales; Paenibacillaceae; Paenibacillus; species	0	0	0	0	0	0.002190707
Bacteria; Firmicutes; Bacilli; Bacillales; Planococcaceae; Planococcus; species	0	0	0	0	0	0.0001153
Bacteria; Firmicutes; Bacilli; Lactobacillales; Streptococcaceae; Streptococcus; species	0	0	0	0	0	0.000230601
Bacteria; Firmicutes; Clostridia; Clostridiales; Peptococcaceae; Peptococcus; species	0	0	0.000156789	0	0	0
Bacteria; Fusobacteria; Fusobacteria; Fusobacteriales; Leptotrichiaceae; genus; species	0.000455529	0	0	0	0.000285415	0
Bacteria; OD1; class; order; family; genus; species	0	0.001036074	0	0	0	0
Bacteria; OP9; class; order; family; genus; species	0	0	0	0.000104822	0	0
Bacteria; Planctomycetes; Phycisphaerae; Phycisphaerales; Phycisphaeraceae; Phycisphaera; species	0.00102494	0.000188377	0.000470367	0	0	0.000230601
Bacteria; Planctomycetes; Phycisphaerae; Phycisphaerales; Phycisphaeraceae; genus; species	0.002733174	0.002260526	0.004860458	0.002201258	0.001331938	0.004842615
Bacteria; Planctomycetes; Phycisphaerae; order; family; genus; species	0	9.41886E-05	0	0	0	0
Bacteria; Planctomycetes; Planctomycetacia; Planctomycetales; Planctomycetaceae; Blastopirellula; species	0.002163763	0.001507017	0.000156789	0.002830189	0.010370088	0
Bacteria; Planctomycetes; Planctomycetacia; Planctomycetales; Planctomycetaceae; Gemmata; species	0	9.41886E-05	0	0	0	0
Bacteria; Planctomycetes; Planctomycetacia; Planctomycetales; Planctomycetaceae; Pirellula; species	0	0	0.000156789	0	0	0
Bacteria; Planctomycetes; Planctomycetacia; Planctomycetales; Planctomycetaceae; Planctomyces; brasiliensis	0	0	0	0.000314465	0	0

Taxonomy	Polymer type					
	Polypropylene			Polyethylene		
	Sampling date					
	20/5/2012	22/5/2012	21/6/2010	20/5/2012	22/5/2012	7/7/2010
Bacteria; Planctomycetes; Planctomycetacia; Planctomycetales; Planctomycetaceae; Planctomyces; species	0.000683293	0.000282566	0	0.001048218	0	0.000922403
Bacteria; Planctomycetes; Planctomycetacia; Planctomycetales; Planctomycetaceae; Rhodopirellula; species	0.011046578	0.000282566	0	0.000524109	0.000285415	0.000230601
Bacteria; Planctomycetes; Planctomycetacia; Planctomycetales; Planctomycetaceae; genus; species	0.002847056	0.000282566	0.001724679	0.000733753	0.000285415	0.000576502
Bacteria; Proteobacteria; Alphaproteobacteria; Caulobacterales; Caulobacteraceae; Brevundimonas; species	0	0.000753509	0	0	0.000190277	0.000461201
Bacteria; Proteobacteria; Alphaproteobacteria; Caulobacterales; Caulobacteraceae; Caulobacter; species	0.000113882	9.41886E-05	0	0	0.000285415	0
Bacteria; Proteobacteria; Alphaproteobacteria; Caulobacterales; Caulobacteraceae; Phenylobacterium; species	0	0	0	0	0.000475692	0
Bacteria; Proteobacteria; Alphaproteobacteria; Caulobacterales; Caulobacteraceae; genus; species	0	0	0.000313578	0	0	0.0001153
Bacteria; Proteobacteria; Alphaproteobacteria; Caulobacterales; Hyphomonadaceae; Hellea; species	0.00102494	0.000376754	0.002351834	0.000838574	0.000285415	0.000345901
Bacteria; Proteobacteria; Alphaproteobacteria; Caulobacterales; Hyphomonadaceae; Hirschia; species	0	0	0.000156789	0	0	0
Bacteria; Proteobacteria; Alphaproteobacteria; Caulobacterales; Hyphomonadaceae; Hyphomonas; Henriciella	0.12424553	0.074314778	0.021950455	0.050524109	0.019883931	0.021791768
Bacteria; Proteobacteria; Alphaproteobacteria; Caulobacterales; Hyphomonadaceae; Hyphomonas; Hyphomonas	0.049652659	0.029763587	0.003292568	0.012893082	0.01512701	0.012452439
Bacteria; Proteobacteria; Alphaproteobacteria; Caulobacterales; Hyphomonadaceae; Hyphomonas; species	0	0	0.000156789	0	0	0
Bacteria; Proteobacteria; Alphaproteobacteria; Caulobacterales; Hyphomonadaceae; Maricaulis; species	0	0	0.000313578	0	0	0
Bacteria; Proteobacteria; Alphaproteobacteria; Caulobacterales; Hyphomonadaceae; Oceanicaulis; species	0	0	0.000627156	0	0	0
Bacteria; Proteobacteria; Alphaproteobacteria; Caulobacterales; Hyphomonadaceae; genus; species	0.000455529	9.41886E-05	0.000156789	0.001257862	0.001141661	0
Bacteria; Proteobacteria; Alphaproteobacteria; Parvularculales; Parvularculaceae; Parvularcula; lutaonensis	0.019587746	0.00065932	0.012543117	0.014779874	0.00485206	0.002767209

Taxonomy	Polymer type					
	Polypropylene			Polyethylene		
	Sampling date					
	20/5/2012	22/5/2012	21/6/2010	20/5/2012	22/5/2012	7/7/2010
Bacteria; Proteobacteria; Alphaproteobacteria; Parvularculales; Parvularculaceae; Parvularcula; species	0.011388225	0.017236507	0.003762935	0.017295597	0.023023499	0.001614205
Bacteria; Proteobacteria; Alphaproteobacteria; Rhizobiales; Aurantimonadaceae; genus; species	0.001708234	0	0	0	0	0
Bacteria; Proteobacteria; Alphaproteobacteria; Rhizobiales; Bradyrhizobiaceae; Afipia; species	0	0	0	0	0	0.000345901
Bacteria; Proteobacteria; Alphaproteobacteria; Rhizobiales; Bradyrhizobiaceae; Bradyrhizobium; species	0	0.000188377	0	0.000209644	0.000285415	0.000922403
Bacteria; Proteobacteria; Alphaproteobacteria; Rhizobiales; Bradyrhizobiaceae; genus; species	0	0	0	0	0	0.000230601
Bacteria; Proteobacteria; Alphaproteobacteria; Rhizobiales; Hyphomicrobiaceae; Devosia; neptuniae	0.000227764	0.000376754	0.000313578	0.001991614	0.001141661	0.000461201
Bacteria; Proteobacteria; Alphaproteobacteria; Rhizobiales; Hyphomicrobiaceae; Filomicrobium; species	0.022093156	0.037110295	0.00454688	0.025052411	0.014175626	0.002421308
Bacteria; Proteobacteria; Alphaproteobacteria; Rhizobiales; Hyphomicrobiaceae; Hyphomicrobium; species	0	0	0.000156789	0	0	0
Bacteria; Proteobacteria; Alphaproteobacteria; Rhizobiales; Hyphomicrobiaceae; Prosthecomicrobium; species	0	0	0	0.000104822	0	0
Bacteria; Proteobacteria; Alphaproteobacteria; Rhizobiales; Methylobacteriaceae; Methylobacterium; species	0.00102494	0	0	0.000419287	0	0.001960106
Bacteria; Proteobacteria; Alphaproteobacteria; Rhizobiales; Methylocystaceae; genus; species	0	0	0	0	9.51384E-05	0
Bacteria; Proteobacteria; Alphaproteobacteria; Rhizobiales; Phyllobacteriaceae; Ahrensia; species	0	0	0.001097523	0.007861635	9.51384E-05	0.002651908
Bacteria; Proteobacteria; Alphaproteobacteria; Rhizobiales; Phyllobacteriaceae; Aquamicrobium; species	0	0	0	0	0	0.0001153
Bacteria; Proteobacteria; Alphaproteobacteria; Rhizobiales; Phyllobacteriaceae; Hoeflea; species	0.000569411	0	0.000156789	0	0	0
Bacteria; Proteobacteria; Alphaproteobacteria; Rhizobiales; Phyllobacteriaceae; Mesorhizobium; species	0	0	0.000313578	0	0	0
Bacteria; Proteobacteria; Alphaproteobacteria; Rhizobiales; Phyllobacteriaceae; Nitratireductor; species	0.023687507	0	0.009720916	0.03951782	0.009133289	0.002075406

Taxonomy	Polymer type					
	Polypropylene			Polyethylene		
	Sampling date					
	20/5/2012	22/5/2012	21/6/2010	20/5/2012	22/5/2012	7/7/2010
Bacteria; Proteobacteria; Alphaproteobacteria; Rhizobiales; Phyllobacteriaceae; Phyllobacterium; species	0	0	0.000313578	0	0	0
Bacteria; Proteobacteria; Alphaproteobacteria; Rhizobiales; Phyllobacteriaceae; genus; species	0.003985879	0.000847697	0.001097523	0.010062893	0.008657597	0.000345901
Bacteria; Proteobacteria; Alphaproteobacteria; Rhizobiales; Rhizobiaceae; Rhizobium; species	0	9.41886E-05	0.000156789	0	0.000190277	0
Bacteria; Proteobacteria; Alphaproteobacteria; Rhizobiales; Rhodobiaceae; Rhodobium; species	0.000341647	0	0.000627156	0	0	0.002075406
Bacteria; Proteobacteria; Alphaproteobacteria; Rhizobiales; Unassigned; Amorphus; species	0.000113882	0	0	0	0	0
Bacteria; Proteobacteria; Alphaproteobacteria; Rhizobiales; Unassigned; Nordella; species	0	0	0.000156789	0	0	0
Bacteria; Proteobacteria; Alphaproteobacteria; Rhizobiales; Xanthobacteraceae; Pseudolabrys; species	0	0	0.000313578	0	0	0.000230601
Bacteria; Proteobacteria; Alphaproteobacteria; Rhizobiales; family; genus; species	0	0.000282566	0	0	0	0
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Albidovulum; species	0.014121398	0.000847697	0.000783945	0.004612159	0.000190277	0.0001153
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Albimonas; donghaensis	0	9.41886E-05	0.000470367	0	0.000285415	0
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Dinoroseobacter; species	0	9.41886E-05	0	0	0.000475692	0.000576502
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Donghicola; species	0.000227764	0	0.000940734	0.000628931	0.003995814	0.001153004
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Jannaschia; helgolandensis	0	0.001695394	0	0	0	0
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Jannaschia; rubra	0	9.41886E-05	0.000156789	0	0	0.0001153
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Jannaschia; seosinensis	0	0.000188377	0	0	0	0
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Jannaschia; species	0	0	0.000156789	0	0	0.0001153

Taxonomy	Polymer type					
	Polypropylene			Polyethylene		
	Sampling date					
	20/5/2012	22/5/2012	21/6/2010	20/5/2012	22/5/2012	7/7/2010
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Labrenzia; alexandrii	0	0	0.000156789	0	0	0
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Labrenzia; species	0.018562806	0	0.000470367	0	0	0
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Loktanella; agnita	0	0.000565131	0	0	0	0
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Loktanella; hongkongensis	0	0	0.000156789	0	0	0
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Loktanella; koreensis	0	0	0.000313578	0.000524109	0	0.000345901
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Loktanella; species	0.002619292	0.016482999	0.006428347	0.011215933	0.001522215	0.003804912
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Lutimaribacter; species	0.000113882	0	0	0	0	0
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Maribius; salinus	0	0	0.000470367	0	0	0.000230601
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Maribius; species	0	0	0.000783945	0	0	0.000461201
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Marinosulfomonas; sp.	0	0	0.000313578	0	0	0
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Marinovum; species	0.000455529	0.000188377	0.001724679	0.000419287	0	0.000345901
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Maritimibacter; species	0	0.000188377	0.000156789	0.001572327	0.006754828	0.000230601
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Marivita; species	0.000113882	0.002354714	0.00297899	0.001048218	0.000570831	0.001498905
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Nautella; species	0.000113882	0	0	0	0	0
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Oceanicola; batsensis	0	0.000282566	0	0	0	0
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Oceanicola; granulosus	0	0	0.000156789	0.000104822	0	0

Taxonomy	Polymer type					
	Polypropylene			Polyethylene		
	Sampling date					
	20/5/2012	22/5/2012	21/6/2010	20/5/2012	22/5/2012	7/7/2010
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Oceanicola; nanhaiensis	0	0	0.000156789	0	0	0
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Oceanicola; pacificus	0	0	0.000313578	0	0	0
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Oceanicola; species	0.000911058	0.013468965	0.003762935	0.012054507	0.002949291	0.002536608
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Octadecabacter; species	0	0	0	0.000104822	0	0
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Palleronia; species	0	0	0	0	0	0.0001153
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Paracoccus; species	0	0	0.000156789	0.000209644	0	0.000691802
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Phaeobacter; species	0	0	0.000156789	0	0	0
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Pseudoruegeria; aquimaris	0.000341647	0.000376754	0.000627156	0.000419287	0.000380554	0.0001153
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Pseudoruegeria; species	0	0	0.000156789	0	0	0
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Rhodobaca; bogoriensis	0	0	0	0.000104822	0	0
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Rhodobacter; maris	0.000113882	9.41886E-05	0	0	0	0
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Rhodobacter; species	0.001594351	9.41886E-05	0.000156789	0.000838574	0.000665969	0.001498905
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Rhodovulum; marinum	0.001708234	0	0.000470367	0.000733753	0	0
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Rhodovulum; species	0.001366587	0.03937082	0.000313578	0.000419287	0.000190277	0.000345901
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Rhodovulum; thermophilus	0.000455529	0	0	0	0	0
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Roseibaca; ekhonensis	0	0	0.000156789	0	0	0

Taxonomy	Polymer type					
	Polypropylene			Polyethylene		
	Sampling date					
	20/5/2012	22/5/2012	21/6/2010	20/5/2012	22/5/2012	7/7/2010
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Roseivivax; species	0	0	0.000156789	0	0	0.0001153
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Roseobacter; brevis	0	0	0.000156789	0	0	0
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Roseobacter; species	0.000569411	0.000188377	0.002822201	0.001257862	9.51384E-05	0.002190707
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Roseovarius; aestuarii	0.001138822	9.41886E-05	0.000156789	0	0	0
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Roseovarius; nubinhibens	0.000341647	0	0	0	0.000570831	0
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Roseovarius; species	0.024712447	0.003579165	0.015992474	0.019392034	0.009608981	0.002997809
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Rubrimonas; species	0.001822116	0.001601206	0.000627156	0.000419287	9.51384E-05	0.000230601
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Ruegeria; species	0	0	0.000313578	0	0	0.000230601
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Shimia; species	0	0	0.000313578	0	0	0
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Silicibacter; species	0.000113882	0	0.000156789	0	0	0
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Stappia; kahanamokuae	0.000455529	0	0	0	0	0
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Sulfitobacter; species	0	0	0.008309815	0.000314465	9.51384E-05	0.02248357
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Tateyamaria; species	0.016399043	0.002731468	0.014267796	0.005660377	0.025687375	0.005303816
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Thalassobacter; arenae	0	0	0	0	0	0.000345901
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Thalassobacter; species	0	0.000188377	0	0.000209644	0	0.000576502
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Thalassobius; aestuarii	0.000113882	9.41886E-05	0	0.00230608	0.001712492	0.000230601

Taxonomy	Polymer type					
	Polypropylene			Polyethylene		
	Sampling date					
	20/5/2012	22/5/2012	21/6/2010	20/5/2012	22/5/2012	7/7/2010
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Thalassobius; species	0.005238583	0.02298201	0.024929445	0.03427673	0.010179812	0.007955725
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Thioclava; species	0.001594351	0.000282566	0.000156789	0	0	0
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Tropicibacter; naphthalenivoran	0.00204988	0	0	0	0	0
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Tropicibacter; species	0	0.000188377	0.000470367	0.000104822	9.51384E-05	0
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Tropicimonas; species	0	9.41886E-05	0.000940734	0.000314465	0	0
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Wenxinia; marina	0	0	0.000470367	0	0	0
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; Wenxinia; species	0.016399043	0.005086183	0.004076513	0.012264151	0.005993721	0.001614205
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodobacterales; Rhodobacteraceae; genus; species	0.054207949	0.054064237	0.016306052	0.078930818	0.039577585	0.016949153
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodospirillales; Acetobacteraceae; Acidocella; species	0	0	0.000156789	0	0	0
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodospirillales; Rhodospirillaceae; Azospirillum; species	0	0	0	0	9.51384E-05	0
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodospirillales; Rhodospirillaceae; Defluviicoccus; species	0.000113882	9.41886E-05	0	0.000314465	0	0.001037703
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodospirillales; Rhodospirillaceae; Oceanibaculum; species	0	0	0	0	9.51384E-05	0
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodospirillales; Rhodospirillaceae; Pelagibius; species	0.002960938	0	0	0	0	0
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodospirillales; Rhodospirillaceae; Rhodocista; species	0	0	0.000156789	0	0	0
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodospirillales; Rhodospirillaceae; Rhodovibrio; species	0.000227764	0	0	0	0	0
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodospirillales; Rhodospirillaceae; Roseospirillum; species	0	0	0	0	0.000190277	0

Taxonomy	Polymer type					
	Polypropylene			Polyethylene		
	Sampling date					
	20/5/2012	22/5/2012	21/6/2010	20/5/2012	22/5/2012	7/7/2010
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodospirillales; Rhodospirillaceae; Thalassobaculum; species	0	0	0.000156789	0	0	0
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodospirillales; Rhodospirillaceae; Thalassospira; species	0	0.000188377	0.000156789	0.000524109	0	0
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodospirillales; Rhodospirillaceae; Tistrella; species	0	0	0	0	0.008372182	0
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodospirillales; Rhodospirillaceae; genus; species	0.000911058	0.000282566	0.001411101	0	0	0.054191168
Bacteria; Proteobacteria; Alphaproteobacteria; Rhodospirillales; family; genus; species	0	0	0.000156789	0	0	0.000230601
Bacteria; Proteobacteria; Alphaproteobacteria; Rickettsiales; Holosporaceae; Holospora; obtusa	0	0	0.000156789	0	0	0
Bacteria; Proteobacteria; Alphaproteobacteria; Rickettsiales; Holosporaceae; Holospora; species	0	0	0	0	0	0.000230601
Bacteria; Proteobacteria; Alphaproteobacteria; Rickettsiales; Rickettsiaceae; Rickettsia; endosymbiont	0.000455529	0	0	0	0	0
Bacteria; Proteobacteria; Alphaproteobacteria; Rickettsiales; Rickettsiaceae; Rickettsia; species	0.007060699	0	0.002665412	0.002725367	0.000761107	0.000345901
Bacteria; Proteobacteria; Alphaproteobacteria; Rickettsiales; SAR116; genus; species	0.000569411	0.001695394	0.000156789	0.002515723	0.0012368	0.033552404
Bacteria; Proteobacteria; Alphaproteobacteria; Rickettsiales; SAR11; Pelagibacter; species	0	0	0.000156789	0	0	0.118182866
Bacteria; Proteobacteria; Alphaproteobacteria; Rickettsiales; SAR11; genus; species	0	0	0.000156789	0	0	0.036434913
Bacteria; Proteobacteria; Alphaproteobacteria; Rickettsiales; Unassigned; Caedibacter; species	0	0	0.000156789	0	0	0.003804912
Bacteria; Proteobacteria; Alphaproteobacteria; Rickettsiales; Unassigned; Midichloria; species	0.000797176	0	0.002665412	0	0.000190277	0
Bacteria; Proteobacteria; Alphaproteobacteria; Rickettsiales; family; genus; species	0.001138822	0.000753509	0.001254312	0.001781971	0.000475692	0.008301626
Bacteria; Proteobacteria; Alphaproteobacteria; Sphingomonadales; Erythrobacteraceae; Altererythrobacter; sp.	0	0	0.002195045	0	0	0.000691802

Taxonomy	Polymer type					
	Polypropylene			Polyethylene		
	Sampling date					
	20/5/2012	22/5/2012	21/6/2010	20/5/2012	22/5/2012	7/7/2010
Bacteria; Proteobacteria; Alphaproteobacteria; Sphingomonadales; Erythrobacteraceae; Erythrobacter; litoralis	0	0	0.000156789	0.000838574	0.000475692	0.0001153
Bacteria; Proteobacteria; Alphaproteobacteria; Sphingomonadales; Erythrobacteraceae; Erythrobacter; longus	0.000455529	0	0.002038257	0	0	0.003804912
Bacteria; Proteobacteria; Alphaproteobacteria; Sphingomonadales; Erythrobacteraceae; Erythrobacter; seohaensis	0	0	0.000313578	0	0	0
Bacteria; Proteobacteria; Alphaproteobacteria; Sphingomonadales; Erythrobacteraceae; Erythrobacter; species	0.035189614	0.023358764	0.01505174	0.008595388	0.001712492	0.009915831
Bacteria; Proteobacteria; Alphaproteobacteria; Sphingomonadales; Erythrobacteraceae; Porphyrobacter; meromictius	0	0	0	0	0	0.0001153
Bacteria; Proteobacteria; Alphaproteobacteria; Sphingomonadales; Erythrobacteraceae; Porphyrobacter; species	0	0	0.000156789	0	0	0.0001153
Bacteria; Proteobacteria; Alphaproteobacteria; Sphingomonadales; Erythrobacteraceae; genus; species	0	0	0.000156789	0	0	0
Bacteria; Proteobacteria; Alphaproteobacteria; Sphingomonadales; Sphingomonadaceae; Novosphingobium; species	0	0	0.000156789	0	9.51384E-05	0.0001153
Bacteria; Proteobacteria; Alphaproteobacteria; Sphingomonadales; Sphingomonadaceae; Novosphingobium; stygium	0	0	0	0	0	0.001037703
Bacteria; Proteobacteria; Alphaproteobacteria; Sphingomonadales; Sphingomonadaceae; Sandarakinorhabdus; limnophila	0	0	0	0	0	0.000230601
Bacteria; Proteobacteria; Alphaproteobacteria; Sphingomonadales; Sphingomonadaceae; Sphingobium; species	0	0	0	0	0	0.002306007
Bacteria; Proteobacteria; Alphaproteobacteria; Sphingomonadales; Sphingomonadaceae; Sphingomonas; sp.	0	0	0.000313578	0	9.51384E-05	0.001729505
Bacteria; Proteobacteria; Alphaproteobacteria; order; family; genus; species	0.017537866	0.003767543	0.004860458	0.007861635	0.005232613	0.007033322
Bacteria; Proteobacteria; Betaproteobacteria; Burkholderiales; Alcaligenaceae; genus; species	0	0.000470943	0	0.000419287	0.000285415	0.000345901
Bacteria; Proteobacteria; Betaproteobacteria; Burkholderiales; Burkholderiaceae; Burkholderia; species	0	0	0.000156789	0	0	0.000345901
Bacteria; Proteobacteria; Betaproteobacteria; Burkholderiales; Burkholderiaceae; Cupriavidus; species	0	0	0.000156789	0.000104822	0	0

Taxonomy	Polymer type					
	Polypropylene			Polyethylene		
	Sampling date					
	20/5/2012	22/5/2012	21/6/2010	20/5/2012	22/5/2012	7/7/2010
Bacteria; Proteobacteria; Betaproteobacteria; Burkholderiales; Burkholderiaceae; Ralstonia; pickettii	0.000113882	0	0	0	0	0
Bacteria; Proteobacteria; Betaproteobacteria; Burkholderiales; Burkholderiaceae; Ralstonia; species	0.003871996	0.000941886	0.003292568	0	0.003710399	0.001844806
Bacteria; Proteobacteria; Betaproteobacteria; Burkholderiales; Comamonadaceae; Acidovorax; species	0	0	0	0.000838574	9.51384E-05	0.000576502
Bacteria; Proteobacteria; Betaproteobacteria; Burkholderiales; Comamonadaceae; Aquabacterium; species	0	0	0	0	0.000285415	0
Bacteria; Proteobacteria; Betaproteobacteria; Burkholderiales; Comamonadaceae; Comamonas; species	0	0	0.000156789	0	0	0.000807103
Bacteria; Proteobacteria; Betaproteobacteria; Burkholderiales; Comamonadaceae; Curvibacter; species	0	0	0.000156789	0	0	0
Bacteria; Proteobacteria; Betaproteobacteria; Burkholderiales; Comamonadaceae; Delftia; species	0	0	0.000156789	0	0	0
Bacteria; Proteobacteria; Betaproteobacteria; Burkholderiales; Comamonadaceae; Pelomonas; species	0	0	0	0	0	0.00322841
Bacteria; Proteobacteria; Betaproteobacteria; Burkholderiales; Comamonadaceae; Variovorax; species	0	0.000376754	0	0	0	0
Bacteria; Proteobacteria; Betaproteobacteria; Burkholderiales; Comamonadaceae; genus; species	0	0	0	0.000104822	0	0
Bacteria; Proteobacteria; Betaproteobacteria; Burkholderiales; Oxalobacteraceae; Janthinobacterium; species	0.000683293	0	0	0	0.000285415	0
Bacteria; Proteobacteria; Betaproteobacteria; Burkholderiales; Oxalobacteraceae; Massilia; species	0	0	0.000156789	0	0	0
Bacteria; Proteobacteria; Betaproteobacteria; Burkholderiales; Oxalobacteraceae; Undibacterium; species	0	0	0.000783945	0	0	0.000807103
Bacteria; Proteobacteria; Betaproteobacteria; Nitrosomonadales; Nitrosomonadaceae; genus; species	0	0	0	0	0	0.000922403
Bacteria; Proteobacteria; Betaproteobacteria; Rhodocyclales; Rhodocyclaceae; Dechloromonas; species	0	0	0	0	0.000190277	0
Bacteria; Proteobacteria; Betaproteobacteria; order; family; genus; species	0	0	0	0	0	0.003689611

Taxonomy	Polymer type					
	Polypropylene			Polyethylene		
	Sampling date					
	20/5/2012	22/5/2012	21/6/2010	20/5/2012	22/5/2012	7/7/2010
Bacteria; Proteobacteria; Deltaproteobacteria; Bdellovibrionales; Bacteriovoraceae; Bacteriovorax; species	0.001822116	0.000282566	0	0	0.000190277	0.0001153
Bacteria; Proteobacteria; Deltaproteobacteria; Bdellovibrionales; Bacteriovoraceae; Peredibacter; species	0.000683293	0	0.000313578	0.000209644	0.000285415	0.000807103
Bacteria; Proteobacteria; Deltaproteobacteria; Bdellovibrionales; Bacteriovoraceae; genus; species	0.000113882	0	0	0	9.51384E-05	0.0001153
Bacteria; Proteobacteria; Deltaproteobacteria; Bdellovibrionales; Bdellovibrionaceae; Bdellovibrio; species	0.003416467	0.001412828	0.000313578	0.013836478	0.001997907	0.001268304
Bacteria; Proteobacteria; Deltaproteobacteria; Bdellovibrionales; Bdellovibrionaceae; genus; species	0.000455529	0	0.001254312	0.001572327	0.000380554	0.000807103
Bacteria; Proteobacteria; Deltaproteobacteria; Desulfarculales; Desulfarculaceae; genus; species	0	0	0	0.000104822	0	0
Bacteria; Proteobacteria; Deltaproteobacteria; Desulfobacterales; Desulfobulbaceae; Desulfobulbus; species	0.000113882	0	0	0.000104822	0.000380554	0.0001153
Bacteria; Proteobacteria; Deltaproteobacteria; Desulfuromonadales; family; genus; species	0.000455529	0.003673354	0.001724679	0.000524109	0	0.001498905
Bacteria; Proteobacteria; Deltaproteobacteria; Myxococcales; Haliangiaceae; Haliangium; species	0	0	0.000156789	0.000314465	0	0
Bacteria; Proteobacteria; Deltaproteobacteria; Myxococcales; Nannocystaceae; Enhygromyxa; species	0	0	0	0	0	0.0001153
Bacteria; Proteobacteria; Deltaproteobacteria; Myxococcales; Nannocystaceae; Plesiocystis; species	0	0.000188377	0.002195045	0	0	0.000461201
Bacteria; Proteobacteria; Deltaproteobacteria; Myxococcales; Nannocystaceae; genus; species	0	0.00263728	0.000940734	0	0	0.000345901
Bacteria; Proteobacteria; Deltaproteobacteria; Myxococcales; Polyangiaceae; Byssovorax; species	0	0	0.000313578	0	0	0.000230601
Bacteria; Proteobacteria; Deltaproteobacteria; Myxococcales; Polyangiaceae; genus; species	0.000569411	0.000470943	0	0	0	0.0001153
Bacteria; Proteobacteria; Deltaproteobacteria; Myxococcales; family; genus; species	0.008541168	0.037581238	0.000940734	0.013312369	0.006754828	0.001844806
Bacteria; Proteobacteria; Deltaproteobacteria; SAR324; family; genus; species	0	0	0	0	0	0.002536608

Taxonomy	Polymer type					
	Polypropylene			Polyethylene		
	Sampling date					
	20/5/2012	22/5/2012	21/6/2010	20/5/2012	22/5/2012	7/7/2010
Bacteria; Proteobacteria; Deltaproteobacteria; order; family; genus; species	0.002733174	0.000376754	0.000156789	0.000733753	0.001522215	0.000691802
Bacteria; Proteobacteria; Gammaproteobacteria; Acidithiobacillales; Acidithiobacillaceae; Acidithiobacillus; species	0	0	0.000156789	0	0	0
Bacteria; Proteobacteria; Gammaproteobacteria; Alteromonadales; Alteromonadaceae; Aestuariibacter; species	0.000227764	0	0	0.000104822	0	0
Bacteria; Proteobacteria; Gammaproteobacteria; Alteromonadales; Alteromonadaceae; Alteromonas; hispanica	0	0	0	0	0	0.0001153
Bacteria; Proteobacteria; Gammaproteobacteria; Alteromonadales; Alteromonadaceae; Alteromonas; macleodii	0	0	0	0	9.51384E-05	0
Bacteria; Proteobacteria; Gammaproteobacteria; Alteromonadales; Alteromonadaceae; Alteromonas; species	0.003074821	0.022887821	0.00611477	0.018448637	0.057368471	0.020062262
Bacteria; Proteobacteria; Gammaproteobacteria; Alteromonadales; Alteromonadaceae; Endobugula; species	0.015829632	0	0	0	0.001046523	0
Bacteria; Proteobacteria; Gammaproteobacteria; Alteromonadales; Alteromonadaceae; Glaciecola; species	0	0	0.000156789	0	0	0.000230601
Bacteria; Proteobacteria; Gammaproteobacteria; Alteromonadales; Alteromonadaceae; Haliea; species	0.002163763	0	0.000156789	0	0	0.005765018
Bacteria; Proteobacteria; Gammaproteobacteria; Alteromonadales; Alteromonadaceae; Marinobacter; litoralis	0	0	0.000313578	0	0	0
Bacteria; Proteobacteria; Gammaproteobacteria; Alteromonadales; Alteromonadaceae; Marinobacter; species	0	0	0.003762935	0	0	0.000345901
Bacteria; Proteobacteria; Gammaproteobacteria; Alteromonadales; Alteromonadaceae; SAR92; species	0	0	0	0	0	0.001844806
Bacteria; Proteobacteria; Gammaproteobacteria; Alteromonadales; Alteromonadaceae; Salinimonas; species	0.000797176	0	0	0.000209644	0	0
Bacteria; Proteobacteria; Gammaproteobacteria; Alteromonadales; Alteromonadaceae; Teredinibacter; turnerae	0	0	0.000627156	0	0	0
Bacteria; Proteobacteria; Gammaproteobacteria; Alteromonadales; Alteromonadaceae; Unassigned; Haliea	0.000569411	0	0.000313578	0	0	0
Bacteria; Proteobacteria; Gammaproteobacteria; Alteromonadales; Alteromonadaceae; Unassigned; Melitea	0	0	0.000313578	0	0	0

Taxonomy	Polymer type					
	Polypropylene			Polyethylene		
	Sampling date					
	20/5/2012	22/5/2012	21/6/2010	20/5/2012	22/5/2012	7/7/2010
Bacteria; Proteobacteria; Gammaproteobacteria; Alteromonadales; Alteromonadaceae; genus; species	0	0	0	0	0	0.0001153
Bacteria; Proteobacteria; Gammaproteobacteria; Alteromonadales; Colwelliaceae; Thalassomonas; species	0	0	0	0	0.000380554	0
Bacteria; Proteobacteria; Gammaproteobacteria; Alteromonadales; Idiomarinaceae; Idiomarina; species	0	0.001695394	0.000156789	0	0.003139568	0
Bacteria; Proteobacteria; Gammaproteobacteria; Alteromonadales; Pseudoalteromonadaceae; Pseudoalteromonas; mariniglutinosa	0	0	0	0	0	0.000461201
Bacteria; Proteobacteria; Gammaproteobacteria; Alteromonadales; Pseudoalteromonadaceae; Pseudoalteromonas; ruthenica	0	0	0.000156789	0	0.000380554	0.001614205
Bacteria; Proteobacteria; Gammaproteobacteria; Alteromonadales; Pseudoalteromonadaceae; Pseudoalteromonas; species	0	0.000470943	0.002038257	0	0.000475692	0.023982474
Bacteria; Proteobacteria; Gammaproteobacteria; Alteromonadales; Pseudoalteromonadaceae; Pseudoalteromonas; spongiae	0	0	0	0	0	0.000691802
Bacteria; Proteobacteria; Gammaproteobacteria; Alteromonadales; Pseudoalteromonadaceae; genus; species	0	0.000282566	0	0	0	0
Bacteria; Proteobacteria; Gammaproteobacteria; Chromatiales; Chromatiaceae; Rheinheimera; perlucida	0	0	0.000156789	0	0	0.0001153
Bacteria; Proteobacteria; Gammaproteobacteria; Chromatiales; Chromatiaceae; Rheinheimera; species	0.000113882	0	0	0	0	0
Bacteria; Proteobacteria; Gammaproteobacteria; Chromatiales; Ectothiorhodospiraceae; Ectothiorhodospira; variabilis	0	0	0	0	0	0.0001153
Bacteria; Proteobacteria; Gammaproteobacteria; Chromatiales; Ectothiorhodospiraceae; Thioalkalispira; sp.	0	0	0.002508623	0	0	0
Bacteria; Proteobacteria; Gammaproteobacteria; Chromatiales; Granulosicoccaceae; Granulosicoccus; species	0	0.000470943	0.000313578	0.000419287	0	0
Bacteria; Proteobacteria; Gammaproteobacteria; Chromatiales; family; genus; species	0.000341647	0	0.000156789	0.000104822	0	0
Bacteria; Proteobacteria; Gammaproteobacteria; Enterobacteriales; Enterobacteriaceae; Citrobacter; species	0	9.41886E-05	0	0	0	0
Bacteria; Proteobacteria; Gammaproteobacteria; Enterobacteriales; Enterobacteriaceae; Enterobacter; species	0	0	0	0	0	0.0001153

Taxonomy	Polymer type					
	Polypropylene			Polyethylene		
	Sampling date					
	20/5/2012	22/5/2012	21/6/2010	20/5/2012	22/5/2012	7/7/2010
Bacteria; Proteobacteria; Gammaproteobacteria; Enterobacteriales; Enterobacteriaceae; Erwinia; toletana	0	0	0	0	0	0.000230601
Bacteria; Proteobacteria; Gammaproteobacteria; Enterobacteriales; Enterobacteriaceae; Escherichia; coli	0	0	0	0.000104822	0	0
Bacteria; Proteobacteria; Gammaproteobacteria; Enterobacteriales; Enterobacteriaceae; Escherichia; species	0	9.41886E-05	0	0	0	0
Bacteria; Proteobacteria; Gammaproteobacteria; Enterobacteriales; Enterobacteriaceae; Klebsiella; species	0	0	0	0	9.51384E-05	0
Bacteria; Proteobacteria; Gammaproteobacteria; Enterobacteriales; Enterobacteriaceae; Serratia; species	0.000113882	9.41886E-05	0	0.000104822	0.000570831	0
Bacteria; Proteobacteria; Gammaproteobacteria; Enterobacteriales; Enterobacteriaceae; genus; species	0.000341647	0.002731468	0	0.00754717	0.000475692	0.000922403
Bacteria; Proteobacteria; Gammaproteobacteria; Legionellales; Coxiellaceae; Aquicella; species	0.000113882	0	0	0	0	0.000922403
Bacteria; Proteobacteria; Gammaproteobacteria; Legionellales; Coxiellaceae; Coxiella; species	0.000683293	0	0.003606146	0.000419287	0	0.001614205
Bacteria; Proteobacteria; Gammaproteobacteria; Oceanospirillales; Alcanivoracaceae; Alcanivorax; species	0	0	0	0	0	0.000345901
Bacteria; Proteobacteria; Gammaproteobacteria; Oceanospirillales; Alcanivoracaceae; Kangiella; koreensis	0	0	0.000156789	0	0	0
Bacteria; Proteobacteria; Gammaproteobacteria; Oceanospirillales; Alcanivoracaceae; Kangiella; species	0	0	0.003135779	0	0	0.001844806
Bacteria; Proteobacteria; Gammaproteobacteria; Oceanospirillales; Hahellaceae; Endozoicomonas; elysicola	0	0	0.000470367	0	0	0
Bacteria; Proteobacteria; Gammaproteobacteria; Oceanospirillales; Hahellaceae; Endozoicomonas; species	0	0	0.000156789	0	0	0
Bacteria; Proteobacteria; Gammaproteobacteria; Oceanospirillales; Halomonadaceae; Halomonas; species	0	0	0.000470367	0.000104822	0	0
Bacteria; Proteobacteria; Gammaproteobacteria; Oceanospirillales; Litoricolaceae; Litoricola; species	0.000227764	0.000282566	0	0.000104822	0	0.004496714
Bacteria; Proteobacteria; Gammaproteobacteria; Oceanospirillales; Oceanospirillaceae; Bermanella; species	0.000797176	0.002166337	0	0.007127883	0.002188184	0

Taxonomy	Polymer type					
	Polypropylene			Polyethylene		
	Sampling date					
	20/5/2012	22/5/2012	21/6/2010	20/5/2012	22/5/2012	7/7/2010
Bacteria; Proteobacteria; Gammaproteobacteria; Oceanospirillales; Oceanospirillaceae; Marinomonas; species	0	0.000188377	0	0	0	0
Bacteria; Proteobacteria; Gammaproteobacteria; Oceanospirillales; Oceanospirillaceae; Oceaniserpentilla; sp.	0	0.012527079	0	0.000209644	0.081533631	0
Bacteria; Proteobacteria; Gammaproteobacteria; Oceanospirillales; Oceanospirillaceae; Oleispira; species	0	0	0	0	0.001427076	0
Bacteria; Proteobacteria; Gammaproteobacteria; Oceanospirillales; Oceanospirillaceae; Pseudospirillum; sp.	0	0	0	0	0	0.000461201
Bacteria; Proteobacteria; Gammaproteobacteria; Oceanospirillales; Oceanospirillaceae; genus; species	0	9.41886E-05	0.000313578	0	0	0.000807103
Bacteria; Proteobacteria; Gammaproteobacteria; Oceanospirillales; Oleiphilaceae; Oleiphilus; species	0	0	0	0	0	0.000230601
Bacteria; Proteobacteria; Gammaproteobacteria; Oceanospirillales; SAR86; genus; species	0.000227764	0	0.000156789	0	0	0.105499827
Bacteria; Proteobacteria; Gammaproteobacteria; Oceanospirillales; family; genus; species	0.000341647	0	0	0.000209644	0.0012368	0.000461201
Bacteria; Proteobacteria; Gammaproteobacteria; Pseudomonadales; Moraxellaceae; Acinetobacter; species	0.000227764	9.41886E-05	0	0.000209644	0.002663876	0.000576502
Bacteria; Proteobacteria; Gammaproteobacteria; Pseudomonadales; Moraxellaceae; Psychrobacter; marincola	0	0	0.000156789	0	0	0
Bacteria; Proteobacteria; Gammaproteobacteria; Pseudomonadales; Moraxellaceae; Psychrobacter; piscidermidis	0	0	0.000156789	0	0	0
Bacteria; Proteobacteria; Gammaproteobacteria; Pseudomonadales; Moraxellaceae; Psychrobacter; species	0	0	0.02869238	0	0	0.00980053
Bacteria; Proteobacteria; Gammaproteobacteria; Pseudomonadales; Pseudomonadaceae; Pseudomonas; fluorescens	0	0	0	0	9.51384E-05	0
Bacteria; Proteobacteria; Gammaproteobacteria; Pseudomonadales; Pseudomonadaceae; Pseudomonas; sp.	0.001480469	0.001601206	0.000470367	0.00524109	0.004281229	0.000230601
Bacteria; Proteobacteria; Gammaproteobacteria; Pseudomonadales; Pseudomonadaceae; Pseudomonas; stutzeri	0	9.41886E-05	0	0	0	0
Bacteria; Proteobacteria; Gammaproteobacteria; Thiotrichales; Piscirickettsiaceae; Piscirickettsia; species	0	0	0.000313578	0	0	0

Taxonomy	Polymer type					
	Polypropylene			Polyethylene		
	Sampling date					
	20/5/2012	22/5/2012	21/6/2010	20/5/2012	22/5/2012	7/7/2010
Bacteria; Proteobacteria; Gammaproteobacteria; Thiotrichales; Piscirickettsiaceae; genus; species	0	0.000188377	0	0	0	0.0001153
Bacteria; Proteobacteria; Gammaproteobacteria; Thiotrichales; Unassigned; Caedibacter; species	0	0	0.000940734	0	0	0
Bacteria; Proteobacteria; Gammaproteobacteria; Thiotrichales; Unassigned; Fangia; species	0	0	0.010348071	0	0	0
Bacteria; Proteobacteria; Gammaproteobacteria; Unassigned; Unassigned; Thiobios; species	0	0	0.001254312	0	0	0
Bacteria; Proteobacteria; Gammaproteobacteria; Vibrionales; Vibrionaceae; Aliivibrio; logei	0	0	0.000156789	0	0	0
Bacteria; Proteobacteria; Gammaproteobacteria; Vibrionales; Vibrionaceae; Aliivibrio; species	0	0	0.000156789	0	0	0.0001153
Bacteria; Proteobacteria; Gammaproteobacteria; Vibrionales; Vibrionaceae; Photobacterium; leiognathi	0	0	0.000940734	0	0	0.0001153
Bacteria; Proteobacteria; Gammaproteobacteria; Vibrionales; Vibrionaceae; Photobacterium; species	0	0	0.001097523	0	0	0.000461201
Bacteria; Proteobacteria; Gammaproteobacteria; Vibrionales; Vibrionaceae; Salinivibrio; proteolyticus	0	0	0.000156789	0	0	0
Bacteria; Proteobacteria; Gammaproteobacteria; Vibrionales; Vibrionaceae; Vibrio; alginolyticus	0	0	0.000627156	0	0	0
Bacteria; Proteobacteria; Gammaproteobacteria; Vibrionales; Vibrionaceae; Vibrio; azureus	0	0	0.000470367	0	0	0
Bacteria; Proteobacteria; Gammaproteobacteria; Vibrionales; Vibrionaceae; Vibrio; fortis	0	0	0.000156789	0	0	0
Bacteria; Proteobacteria; Gammaproteobacteria; Vibrionales; Vibrionaceae; Vibrio; harveyi	0	0	0.000940734	0	0	0.0001153
Bacteria; Proteobacteria; Gammaproteobacteria; Vibrionales; Vibrionaceae; Vibrio; inusitatus	0	0	0.000313578	0	0	0
Bacteria; Proteobacteria; Gammaproteobacteria; Vibrionales; Vibrionaceae; Vibrio; lentus	0	0	0.000156789	0	0	0
Bacteria; Proteobacteria; Gammaproteobacteria; Vibrionales; Vibrionaceae; Vibrio; parahaemolyticus	0	0	0.003762935	0	0	0.000230601

Taxonomy	Polymer type					
	Polypropylene			Polyethylene		
	Sampling date					
	20/5/2012	22/5/2012	21/6/2010	20/5/2012	22/5/2012	7/7/2010
Bacteria; Proteobacteria; Gammaproteobacteria; Vibrionales; Vibrionaceae; Vibrio; proteolyticus	0	0	0.000470367	0	0	0
Bacteria; Proteobacteria; Gammaproteobacteria; Vibrionales; Vibrionaceae; Vibrio; species	0	0.000376754	0.247883349	0.00115304	0.000761107	0.023867174
Bacteria; Proteobacteria; Gammaproteobacteria; Vibrionales; Vibrionaceae; Vibrio; vulnificus	0	0	0.000783945	0	0	0
Bacteria; Proteobacteria; Gammaproteobacteria; Vibrionales; Vibrionaceae; genus; species	0	0	0.000627156	0	0	0
Bacteria; Proteobacteria; Gammaproteobacteria; Xanthomonadales; Sinobacteraceae; genus; species	0.016171279	0.010078177	0.000783945	0.008909853	0.004661783	0.000807103
Bacteria; Proteobacteria; Gammaproteobacteria; Xanthomonadales; Xanthomonadaceae; Stenotrophomonas; sp	0	0.00065932	0.000156789	0	0	0.000230601
Bacteria; Proteobacteria; Gammaproteobacteria; order; family; genus; species	0.001138822	0.00593388	0.104578238	0.000314465	9.51384E-05	0.007033322
Bacteria; Proteobacteria; class; order; family; genus; species	0	9.41886E-05	0	0	0	0
Bacteria; SR1; class; order; family; genus; species	0.000113882	0	0	0	0	0
Bacteria; Spirochaetes; Spirochaetes; Spirochaetales; Leptospiraceae; Turneriella; parva	0	0	0	0	0	0.0001153
Bacteria; Spirochaetes; Spirochaetes; Spirochaetales; Spirochaetaceae; Spirochaeta; species	0	0	0	0.000104822	0	0
Bacteria; TM6; class; order; family; genus; species	0	0	0	0	0	0.000576502
Bacteria; TM7; class; order; family; genus; species	0	9.41886E-05	0	0	0	0
Bacteria; Verrucomicrobia; Opitutae; Puniceicoccales; Puniceicoccaceae; Cerasicoccus; species	0.001480469	0	0	0	0	0
Bacteria; Verrucomicrobia; Opitutae; Puniceicoccales; Puniceicoccaceae; Coraliomargarita; akajimensis	0	0	0	0	0	0.0001153
Bacteria; Verrucomicrobia; Opitutae; Puniceicoccales; Puniceicoccaceae; Coraliomargarita; species	0.000455529	0.002166337	0.000313578	0.000419287	0.002854153	0.010838234
Bacteria; Verrucomicrobia; Opitutae; Puniceicoccales; Puniceicoccaceae; Fucophilus; species	0.000341647	0	0	0	0	0
Bacteria; Verrucomicrobia; Opitutae; Puniceicoccales; Puniceicoccaceae; Puniceicoccus; species	0	0	0	0	0	0.000345901

Taxonomy	Polymer type					
	Polypropylene			Polyethylene		
	Sampling date					
	20/5/2012	22/5/2012	21/6/2010	20/5/2012	22/5/2012	7/7/2010
Bacteria; Verrucomicrobia; Opitutae; Puniceicoccales; Puniceicoccaceae; genus; species	0	0	0	0	0	0.005303816
Bacteria; Verrucomicrobia; Spartobacteria; Chthoniobacter; family; genus; species	0	0	0	0	0	0.0001153
Bacteria; Verrucomicrobia; Spartobacteria; order; family; genus; species	0.000113882	0	0	0	0	0.000230601
Bacteria; Verrucomicrobia; Verrucomicrobiae; Verrucomicrobiales; Rubritaleaceae; Rubritalea; species	0	0.000188377	0.033866416	0	0	0.000691802
Bacteria; Verrucomicrobia; Verrucomicrobiae; Verrucomicrobiales; Verrucomicrobiaceae; Acidimethylosilex; species	0.010591049	0	0.000156789	0	0	0
Bacteria; Verrucomicrobia; Verrucomicrobiae; Verrucomicrobiales; Verrucomicrobiaceae; Haloferula; species	0	0	0.000156789	0	0	0
Bacteria; Verrucomicrobia; Verrucomicrobiae; Verrucomicrobiales; Verrucomicrobiaceae; Luteolibacter; species	0	0	0	0	0	0.000345901
Bacteria; Verrucomicrobia; Verrucomicrobiae; Verrucomicrobiales; Verrucomicrobiaceae; Prosthecobacter; vanneervenii	0	0	0	0	0	0.000807103
Bacteria; Verrucomicrobia; Verrucomicrobiae; Verrucomicrobiales; Verrucomicrobiaceae; Roseibacillus; persicicus	0	0	0	0.000104822	0.000856246	0.000345901
Bacteria; Verrucomicrobia; Verrucomicrobiae; Verrucomicrobiales; Verrucomicrobiaceae; Roseibacillus; species	0	0	0.000313578	0	0	0
Bacteria; Verrucomicrobia; Verrucomicrobiae; Verrucomicrobiales; Verrucomicrobiaceae; genus; species	0.000911058	0.00065932	0.001724679	0.000104822	0.002188184	0.000461201
Bacteria; phylum; class; order; family; genus; species	0	0.000470943	0.000313578	0	0.000380554	0.000807103

ANNEX IV – REGULATION OF MARINE LITTER IN THE SHIPPING SECTOR

Much of the existing legislation as well as providing guidelines on what waste can or cannot be discarded at sea, also provides guidelines on waste management practices. For example, the MARPOL Convention (IMO, 2015 #338) provides guidance and regulations on the implementation of port reception facilities as well as training and education on the issue. It also stipulates how garbage should be managed at sea, including the use of placards, garbage management plans, record books, incinerators and control of cargo residues (Øhlenschlæger, 2013 #437).

In addition to legislation, a number of voluntary schemes exist which provide further guidelines on waste management at sea. The International Organization for Standardisation (ISO) has two standards relevant to MARPOL V, specifically for ships[19] and ports.[20] Similarly, the Blue Angel offers a label for "environmentally sound" ship operations[21] (RAL gGmbH, 2010). Further certification or guidance may be available to specific industries such as for the *Clean Shipping Index* (CSI)[22] for container ships or the role of the trade association Cruise Lines International Association (CLIA)[23] for the cruise industry. There is no obligation for ships or ports to follow these standards and there may be costs for implementation and certification but they may also provide competitive advantage. In addition, specific vessel operators may make strategies to further differentiate themselves in the market or express their commitment to stewardships in the marine environment.[24,25] However, some of these certifications and strategies have come under criticism for not being more ambitious than the minimum requirements of maritime law (Sherrington, 2014 #401@30), in addition the parity between certification and practice is not guaranteed (Klein, 2011 #355). With regards to marine plastics, not all of the certifications programmes are explicit about the role of plastics, and rather refer to waste in general terms.

Port reception facilities are one of the most important tools for addressing waste generated at sea from all sectors, and if appropriately designed can incentivise best practices (Newman, 2015 #376). Well-designed port reception facilities will encourage shippers to dispose of their waste correctly, relying on clear waste definitions, communication between actors, timely administration and appropriate inspections (Øhlenschlæger, 2013 #437). MARPOL V requires the provision of facilities for the reception of ship generated residues and litter (IMO, 2012 #336@25). The IMO have also published a Comprehensive Manual on Port Reception Facilities (IMO, 1999 #334), giving guidance on waste management strategies, types of waste, collecting and treating waste, financing and cost recovery. Since 2006 the IMO have also integrated a port reception facility module, or the Port Reception Facility Database (PRFD) into their Global Integrated Shipping Information System (GISIS) (IMO, 2015 #337).

Awareness-raising can also help to reduce shipping related marine litter impacts and costs by highlighting the costs to stakeholders in both socio-economic and environmental terms. For instance, the shipping industry now has compulsory training on marine litter, following leverage from the Dutch Government and the ProSea Foundation on the IMO to amend the STWC (International Convention on Standards of Training) (ProSea, 2011 #388). Such training, as well as the enforcement of good practices will also be associated with a number of costs, which would also need to be included in a socio-economic assessment.

[19] ISO 21070:2011 Management and handling of shipboard garbage (ISO, 2011).
[20] ISO 16304:2013 Arrangement and management of port waste reception facilities (ISO, 2013)
[21] Requirements 3.3.5 Waste Disposal; 3.3.6 Waste Incineration; and 3.3.16 Environmentally Sound Recycling all refer to waste management on ships. In addition to the guidelines included in MARPOL, they recommend actions such as purchasing strategies with aim towards avoiding waste.
[22] CSI provides a tool for cargo operators to calculate and minimize the carbon footprint of their vessels. One of the environmental parameters of the CSI is for waste control, although it is not apparent that its requirements go beyond those laid out in MARPOL V.
[23] Membership to the CLIA
[24] E.g. MATSON Navigation a shipping operator in the Pacific Ocean have a Zero Waste Policy, including a number of further waste related projects. This involved an initial investment of $224,000 to include a container designed for storing waste on board each of their vessels (MATSON, 2014).
[25] E.g. Royal Caribbean Cruises have published a number of reports reporting on their commitment to environmental stewardship, including indicators on waste to land fill, recycling etc. (Royal Caribbean Cruises Ltd., 2014)

ANNEX V – ECONOMIC COSTS OF ACTIONS TO REDUCE MARINE LITTER

Shoreline-based clean-ups

Table A.1 Estimated paid clean-up and management costs of marine litter

Country / Region	Estimated cost at national and municipality level	Source
Belgium	EUR 10.4 million (ave. EUR 200,000/municipality/yr)	Mouat 2010
Netherlands	EUR 10.4 million (ave. EUR 200,000/municipality/yr) Costs are higher for areas with high visitor numbers, for example, the Hague Municipality spends EUR 626,709/year with costs for processing litter (including transport) about EUR165/ton	Mouat 2010; OSPAR 2009
Peru	$2.5 million in labour costs (ave. $400,000/yr in municipality of Ventanillas)	Alfaro 2006 cited in UNEP 2009
UK	EUR 18 million (ave. EUR 146,000/municipality/yr) (per km cleaning costs range from EUR 171 to EUR 82,000/km/yr). Specific municipality costs: • Suffolk: approx. GBP 60,000/yr on 40 km of beaches • Carrick District Council (Devon): approx. GBP 32,000/yr on 5 km of beaches • Studland (Dorset): GBP 36,000/yr to collect 12 to 13 tonnes of litter each week in the summer along 6 km of beaches • Kent coastline: direct and indirect cost of litter estimated at over GBP 11 million/yr • Annual expenditure on beach cleaning in 56 local authorities ranged from GBP 15/km in West Dunbartonshire to GBP 50,000/km in Wyre	Fanshawe 2002; Mouat 2010; OSPAR 2009
Bay of Biscay and Iberian coast	A Spanish council with 30 beaches (5 Blue Flags) spends around EUR 80,000/year on beach cleaning A French council with 30 beaches (5 Blue Flags) spends around EUR 400,000/year on 'beach caring' (includes beach clearing, monitoring of buoys, coastguards, etc.), of which around 20% relates to beach clearing In Landes, the cost of cleaning-up 108 km of sandy beaches was EUR 8 million between 1998 and 2005 Cost of beach cleaning between EUR 4,500 to 50,000/year/council corresponding to average cost of EUR 6,500/km of cleaned beach/year.	OSPAR 2009
Poland	Beach cleaning and removing litter from harbour waters cost EUR 570,000 in 2006 (same amount also spent in five communes and two ports)	UNEP 2009
Oregon, California, Washington (USA)	Annual combined expenditure of $520 million ($13 dollars/resident/year) to combat litter and curtail potential marine debris	Stickel 2012
APEC region	$1,500/tonne in 2007 terms	McIlgorm et al. 2009

Taxes and levies on single-use plastics

In a number of countries, levies on e.g. single-use plastic bags have helped reduce the number of these items. The Irish plastic bag levy is a widely discussed and cited example of the successful application of an economic instrument. After introducing a €0.15 levy on retail plastic bags, sales in retail outlets dropped by 90%. The levy was also very cost-effective, as stores could use the existing Value Added Tax scheme for collecting and reporting the levy (Convery et al. 2007; Pape et al. 2011).

A recent study commissioned by the Welsh Government has shown that, since the introduction of the levy in 2011, the SUCB (single use carrier bag) use has declined by 71%. Wales was the first nation of the UK to introduce a levy on the use of SUCB (Welsh Government 2015). In addition, the report shows that the impact of the levy on retailers has been either neutral, or positive. Consumers' support of the levy, already strong in 2011 (61%) has been growing and has reached now 74% of the whole population.

Deposit Schemes are Applicable at Different Scales

Australia

When discussing options for reducing litter reaching the marine environment, large-scale solutions such as state- or country-wide deposit schemes come into mind. For example, Hardesty et al. (2014) report that South Australia's container deposit scheme, which applies a AUS $0.10 refundable deposit to beverage containers, resulted in a 3-fold reduction in the number of beverage containers lost to beaches.

However, this instrument can in principle be applied at all scales and most locations. Hardesty (2015, oral communication) is reporting an initiative at the Boronia West Primary school in Victoria, Australia, where the school introduced a 10 cent deposit on candy wrappers sold at the school refectory.

The idea originated from the children themselves just after learning about the impacts of litter on the marine environment, notably by following a class with a post-mortem examination of seabirds with plastic material in their stomachs. The children could then connect the impact of litter on wildlife with their school environment, where littering does occur and where candy wrappers are often found on the schoolyard. The deposit scheme is now in place and has been extended to a second school.

http://studentplanetsavers.global2.vic.edu.au/2013/03/05/emerald-primary-container-deposit-scheme/

Ecuador charge on plastic bottles

In 2011, purchasers of plastic beverage bottles were charged with a refundable tax of $0.02 per PET bottle. This has led to a significant increase in PET bottle recycling from 30% in 2011 to 80% in 2012, when 1.13 million of PET bottles were recycled out of 1.40 million produced.

Source: Ministry of Environment of Ecuador http://www.ambiente.gob.ec/ecuador-incremento-la-recoleccion-de-botellas-pet-en-2012/

ANNEX VI – EXAMPLES OF LITTER REDUCTION MEASURES

Table A VI.1 Measures addressing marine litter issues from tourism – from prevention to clean-up

Type of measure	Example from practice
Pier-side reception facilities	In several US states, pier-side reception facilities are provided for safe disposal of monofilament line by recreational fishers. Collected fishing gear is subsequently recycled (Macfadyen 2009). For example, the Reel in and Recycle scheme, launched by Boat U.S. Foundation and sponsored by NOAA and the National Fish and Wildlife Foundation Boat U.S. Foundation, has the intention to reduce marine litter by recycling the materials, raising awareness as well as monitoring the waste collected. Within its first four years, the scheme installed over 1,200 bins across 32 states in the US, with a continued growing interest and demand (Shingledecker, 2010 #402).
Awareness raising and targeted education campaigns	The Green Blue initiative in the UK led by The Royal Yachting Association & The British Marine Federation raises awareness of marine litter among the recreational boating community, providing education, solutions and toolkits. The Special Monitoring and Coastal Environmental Assessment Regional Activity Centre of the Northwest Pacific Action Plan (NOWPAP CEARAC) developed marine litter guidelines for tourists and tour operations in marine and coastal areas which set out best practices for tourists participating in marine recreational activities (e.g. cruising, fishing and diving) and coastal recreational activities (e.g. camping, barbequing and bathing) as well as suggested actions for tour operators to reduce tourist-generated marine litter (NOWPAP CEARAC, 2011 #378). The Travel Corporation (an international travel group with a number of established brands such as Contiki Tours) established The TreadRight Foundation to encourage sustainable tourism within its family of brands. This Foundation supports a number of projects across the world including a partnership between Contiki's conservation programme – Contiki Cares – and Surfrider Foundation Australia which sponsors a number of beach clean-ups along the coast, and awareness raising activities.[26] TreadRight's has supported the production and subsequent distribution of a documentary – 'Scars of Freedom' – which chronicles a whale's fight for life off the coast of Chile's Juan Fernandez Archipelago after getting caught in drift net.[27] UNEP launched the Marine Litter MOOC (Massive Open Online Course), a large-scale training and educational effort in October 2015. With the focus to stimulate leadership and offer opportunities for actionable and change oriented learning related to marine litter, this course had more than 5000 signups within two weeks of the launch. The main objectives of the MARLISCO project (involving 15 European countries, http://www.marlisco.eu/) were to increase the awareness of the consequences of societal behaviour in relation to waste production and management on marine socio-ecological systems, to promote co-responsibility among the different actors, to define a more sustainable collective vision, and to facilitate grounds for concerted actions through the successful implementation of the MMLAP.

[26] http://www.treadright.org/project/rising-against-plastic-surfrider-foundation
[27] http://www.treadright.org/Scars%20of%20Freedom

Type of measure	Example from practice
Sustainable tourism initiatives	Members of The Caribbean Hotel Association (CHA) established The Caribbean Alliance for Sustainable Tourism (CAST) which aims to promote responsible environmental and social management within the hotel and tourism sector. CAST focuses on the development of sustainable tourism certification and standards, provides guidance and expertise in awareness raising programmes, environmental management systems (EMS) and best practices to support sustainable tourism.[28] In Barbados, Green Globe Certified Hotels including Almond Hotel Group, The Bougainvillea, The PomMarine Hotel, The Sand Acres Hotel, The Southern Palms Hotel, The Palm Beach Group, members of the Green Hotels Association of the US and CAST support local programmes for improved solid waste management in beach areas (UNEP-CAR/RCU, 2008 #421). In 1999, the Roteiros de Charme Hotel Association in Brazil developed a voluntary Ethics and Environmental Code of Conduct in co-operation with UNEP's Tourism Programme which provides a benchmark for biodiversity conservation and the quality of holiday destinations. Implementation of the code has helped to reduce pressures on the environment for example preventing pollution from untreated sewage and contamination of waterways and marine environments, reducing solid waste generation and inappropriate waste disposal practices, strengthening public awareness and protecting biodiversity.[29]
Clean-up activities	In the UK, there are a number of voluntary clean-up initiatives such as Adopt-a-Beach which involves local communities, businesses, schools and individuals in regular beach cleans and surveys, Beach Watch which coordinates regular and a large annual national beach clean activity and marine litter survey organized by the Marine Conservation Society, community beach clean-up projects organized by Surfers Against Sewage and Keep Britain Tidy. Whilst these activities can only access part of the existing problem, they can lead to greater pro-environmental intentions, which in turn can reduce the litter entering the environment (Wyles, revised and resubmitted #435). Some clean-up activities engage recreational users in both collecting litter and recording what is found in a specific area. For example a number of initiatives engage scuba divers such as Neptune's Army of Rubbish Cleaners, the Green Fins project, Dive Against Debris and Project AWARE (a global movement of scuba divers). Travel Trawl loans equipment to recreational sailors to collect samples of plastic debris during their own sailing trips and report back to the Algalita Foundation. A number of hotels, groups and travel operators are involved in beach clean-up activities. For example, in 2014, a multinational travel operator – the TUI Group – organized a series of Big Holiday Beach Clean events worldwide to raise awareness about marine litter among tourists and local authorities (TUI Group, 2014). The Berjaya Hotels & Resorts group in Malaysia supports annual clean-up events on various beaches such as the Redang Island Clean-up Day and Tioman Island.[30] The Conrad Hotel Maldives supports regular beach clean-up activities with SubAqua Dive Center and supports improved waste management practices including reduced use of plastic water bottles.[31]

[28] http://www.caribbeanhotelandtourism.com/CAST.php
[29] http://www.roteirosdecharme.com.br/aboutus.php
[30] http://www.berjayahotel.com/en/corporate_social_responsibility
[31] http://news.conradhotels.com/assets/CNRD/properties/International/ConradMaldivesRangaliIsland/2013/10ConradMaldivesRangaliIsland_the_environment_and_CSR_Jan2014.pdf

ANNEX VII – SUMMARY OF POTENTIAL ECOLOGICAL, SOCIAL AND ECONOMIC IMPACTS OF MARINE PLASTICS AND MICROPLASTICS, SUB-DIVIDED BY SIZE RANGE

Key: relative weight of evidence			
Major knowledge/ evidence gaps	Weak knowledge and evidence	Fair knowledge and evidence	Good knowledge and evidence

		Nano <1 um	Micro <5 mm	Meso <2.5 cm	Macro <1 m	Mega >1 m
	Examples of marine litter	e.g. nanofibres from textiles; rubber dust from tyre wear; nanoparticles in products and pharmaceuticals. Have not yet been detected as litter due to technical limitations, but undoubtedly present in environment	e.g. microbeads from personal care products; fragmentation of existing (plastic) products; rubber dust from tyre wear; fibres from textiles; plastic resin beads from industry; polystyrene; plastic from blasting in shipyards; potentially particulates from poor waste incineration	e.g. bottle caps; cigarette filters and butts; plastic pellets; windblown/ storm-washed waste	e.g. beverage bottles and cans; plastic bags; food packaging; other packaging; disposable tableware/cutlery; beer-ties; fishing lines and floats, buoys; tyres; pipes; balloons; toys; whole textiles	e.g. abandoned fishing nets and traps; rope; boats; plastic films from agriculture; construction PVC (Polyvinyl chloride)
Ecological impact	**Mechanism of impact** Ingestion of particles/ debris or other uptake mechanism	Uptake via absorption, ventilation, and/or ingestion; transfer of chemicals: e.g. mussels, oysters, sponges, fish, corals, phytoplankton	Uptake via absorption, ventilation and/or ingestion: e.g. fish, birds, oysters, corals	Ingestion: e.g. birds, fish and marine mammals	Ingestion/entanglement: e.g. birds, crustaceans, turtles, whales, dolphins, sea lions	Entanglement: e.g. whales, dolphins, sea lions, turtles, birds, fish
			Uptake via absorption, ventilation and/or ingestion with transfer of chemicals: e.g. fish, birds, oysters, corals	Ingestion with transfer of chemicals: e.g. birds, fish and marine mammals		
	Entanglement	n/a	invertebrates	birds, fish and marine mammals		

		Nano <1 um	Micro <5 mm	Meso <2.5 cm	Macro <1 m	Mega >1 m
	Rafting: movement of organisms using plastic as a raft Individual; & ecological impact					Rafting, movement of plants, seeds, animals.
	Impacts: Individual organism	Sub-lethal impacts at lower levels of organization, e.g. cellular intrusion, changes in gene expression.	Potential effects from physical presence of ingested plastic, concerns about possible effects from transfer of chemicals: reduced feeding, sub-lethal impacts at lower levels of organization, e.g. cellular intrusion, changes in gene expression.	Sub-organismal impacts: e.g. organ damage. Organismal impacts: death, reduced feeding & impairment of digestive process: impacts on fitness & reproduction	Sub-organismal impacts: e.g. organ damage. Organismal impacts: death, reduced feeding & impairment of digestive process: impacts on fitness & reproduction	Death and injury due to entanglement in ALDFG
	Ecological impacts (e.g. population, assemblages, ecosystems)	Potential for population decline, changes in assemblages and ecosystem functioning, e.g. shift in microbial community	Potential effects from physical presence of ingested plastic, concerns about possible effects from transfer of chemicals: reduced feeding, sub-lethal impacts at lower levels of organization, e.g. cellular intrusion, changes in gene expression. Potential for population decline, changes in assemblages and ecosystem functioning, e.g. endocrine disruption in fish	Sub-organismal impacts: e.g. organ damage. Organismal impacts: death, reduced feeding & impairment of digestive process: impacts on fitness & reproduction Potential for population decline, changes in assemblages and ecosystem functioning, e.g. mass decline in population due to ingestion causing mortality in sea turtles and sea birds	Evidence of effects on assemblages and ecosystem functioning, e.g. plastic bags. Population decline, changes in assemblages and ecosystem functioning, e.g. from mass strandings of sea turtles from entanglement; changes in assemblages due to changes in habitat structure	Population decline, changes in assemblages and ecosystem functioning, e.g. changes in populations and assemblages due to ghost fishing. Newly introduced species (NIS) predation / displacement of indigenous species.

		Nano <1 um	Micro <5 mm	Meso <2.5 cm	Macro <1 m	Mega >1 m
Social impacts	**Human health**	Risk from nano particles passing cell walls; possible gender differences in chemical uptake. Potential perceived (subjective) risk from chemical contamination in fish and shellfish eaten in the future, and possible transfer of pathogens.	Perceived (subjective) risk from chemical contamination in fish and shellfish eaten, and possible transfer of pathogens.	Injury on beaches, danger to fishers, gender-differentiated well-being loss / mental health impacts from degraded environment.	Loss of protein (where fish availability reduced). Physical health risks of boats / the individual becoming entangled, and mental health risks from degraded environment. More indirect: loss of health benefits by avoiding coastlines now with litter	Navigation hazard
	Communities (coastal tourism)	Potential loss of well-being depending on awareness and perception of risk		Loss of well-being, fish stocks, tourist revenue in plastic-polluted living environments	Loss of well-being, fish stocks, tourist revenue in plastic-polluted living environments, risk to community cohesion / local identity / cultural values	Loss of well-being, fish stocks, tourist revenue in plastic-polluted living environments, risk to community cohesion / local identity / cultural values
	Communities (coastal fishing communities)	Potential perceived (subjective) risk from chemical contamination in fish, shellfish and aquatic plants eaten. Pending perception issues this can lead to lower demand for and/or value of fish/seafood.		Ingestion could lead to (perceptions of) lower quality fish and hence lesser market value	Entanglement in: propellers and damage to fishing vessel; related loss of fishing time, loss of fish and associated revenues. Potential risks to fisheries community cohesion / local identity / cultural values	Ghost fishing: loss of output and hence livelihoods; damage to boats and equipment
Economic impacts	**Fisheries and aquaculture**	Unlikely to have any discernible impact on sales unless new information comes forward on health impacts that leads to perception changes. No current knowledge of actual reproductive rate or health/size of fish from nano and micro.		Only if integrated into beach labelling	Evidence of marine litter can discourage tourism and recreation on beaches, reducing income and/or well-being	Reduction in tourist and recreation numbers and hence income / well-being. Increased costs of clean-up to maintain activities. Damage to vessels (propellers, cooling systems).
	Tourism and Recreation	n/a	unlikely	Loss of income and livelihoods	Loss of income and livelihoods	Loss of income and livelihoods

		Nano <1 um	Micro <5 mm	Meso <2.5 cm	Macro <1 m	Mega >1 m
	Shipping	n/a	n/a	n/a	Damage to vessels (propellers, cooling systems); potential loss of productivity and revenues from delays or accidents affecting supply chains.	Damage to vessels (propellers, cooling systems); potential loss of productivity and revenues from delays or accidents affecting supply chains.
	Local authorities and municipalities	Degradation of the natural environment within their jurisdiction. Potential increased cost of wastewater treatment	Degradation of the natural environment within their jurisdiction. Potential increased cost of wastewater treatment	Degradation of the natural environment/heritage; Cost of clean-up and infrastructures. Loss of income and livelihoods	Degradation of the natural environment / heritage; Cost of clean-up and infrastructures. Loss of income and livelihoods	Clean-up costs including debris from shipping accidents

ANNEX VIII – LIST OF GESAMP REPORTS AND STUDIES

The following reports and studies have been published so far. They are available from the GESAMP website: http://gesamp.org

1. Report of the seventh session, London, 24-30 April 1975. (1975). Rep. Stud. GESAMP, (1):pag.var. Available also in French, Spanish and Russian

2. Review of harmful substances. (1976). Rep. Stud. GESAMP, (2):80 p.

3. Scientific criteria for the selection of sites for dumping of wastes into the sea. (1975). Rep. Stud. GESAMP, (3):21 p. Available also in French, Spanish and Russian

4. Report of the eighth session, Rome, 21-27 April 1976. (1976). Rep. Stud. GESAMP, (4):pag.var. Available also in French and Russian

5. Principles for developing coastal water quality criteria. (1976). Rep. Stud. GESAMP, (5):23 p.

6. Impact of oil on the marine environment. (1977). Rep. Stud. GESAMP, (6):250 p.

7. Scientific aspects of pollution arising from the exploration and exploitation of the sea-bed. (1977). Rep. Stud. GESAMP, (7):37 p.

8. Report of the ninth session, New York, 7-11 March 1977. (1977). Rep. Stud. GESAMP, (8):33 p. Available also in French and Russian

9. Report of the tenth session, Paris, 29 May - 2 June 1978. (1978). Rep. Stud. GESAMP, (9):pag.var. Available also in French, Spanish and Russian

10. Report of the eleventh session, Dubrovnik, 25-29 February 1980. (1980). Rep. Stud. GESAMP, (10):pag.var. Available also in French and Spanish

11. Marine Pollution implications of coastal area development. (1980). Rep. Stud. GESAMP, (11):114 p.

12. Monitoring biological variables related to marine pollution. (1980). Rep. Stud. GESAMP, (12):22 p. Available also in Russian

13. Interchange of pollutants between the atmosphere and the oceans. (1980). Rep. Stud. GESAMP, (13):55 p.

14. Report of the twelfth session, Geneva, 22-29 October 1981. (1981). Rep. Stud. GESAMP, (14):pag.var. Available also in French, Spanish and Russian

15. The review of the health of the oceans. (1982). Rep. Stud. GESAMP, (15):108 p.

16. Scientific criteria for the selection of waste disposal sites at sea. (1982). Rep. Stud. GESAMP, (16):60 p.

17. The evaluation of the hazards of harmful substances carried by ships. (1982). Rep. Stud. GESAMP, (17):pag.var.

18. Report of the thirteenth session, Geneva, 28 February - 4 March 1983. (1983). Rep. Stud. GESAMP, (18):50 p. Available also in French, Spanish and Russian

19. An oceanographic model for the dispersion of wastes disposed of in the deep sea. (1983). Rep. Stud. GESAMP, (19):182 p.

20. Marine pollution implications of ocean energy development. (1984). Rep. Stud. GESAMP, (20):44 p.

21. Report of the fourteenth session, Vienna, 26-30 March 1984. (1984). Rep. Stud. GESAMP, (21):42 p. Available also in French, Spanish and Russian

22. Review of potentially harmful substances. Cadmium, lead and tin. (1985). Rep. Stud. GESAMP, (22):114 p.

23. Interchange of pollutants between the atmosphere and the oceans (part II). (1985). Rep. Stud. GESAMP, (23):55 p.

24. Thermal discharges in the marine Environment. (1984). Rep. Stud. GESAMP, (24):44 p.

25. Report of the fifteenth session, New York, 25-29 March 1985. (1985). Rep. Stud. GESAMP, (25):49 p. Available also in French, Spanish and Russian

26. Atmospheric transport of contaminants into the Mediterranean region. (1985). Rep. Stud. GESAMP, (26):53 p.

27. Report of the sixteenth session, London, 17-21 March 1986. (1986). Rep. Stud. GESAMP, (27):74 p. Available also in French, Spanish and Russian

28. Review of potentially harmful substances. Arsenic, mercury and selenium. (1986). Rep. Stud. GESAMP, (28):172 p.

29. Review of potentially harmful substances. Organosilicon compounds (silanes and siloxanes). (1986). Published as UNEP Reg. Seas Rep. Stud., (78):24 p.

30. Environmental capacity. An approach to marine pollution prevention. (1986). Rep. Stud. GESAMP, (30):49 p.

31. Report of the seventeenth session, Rome, 30 March - 3 April 1987. (1987). Rep. Stud. GESAMP, (31):36 p. Available also in French, Spanish and Russian

32. Land-sea boundary flux of contaminants: contributions from rivers. (1987). Rep. Stud. GESAMP, (32):172 p.

33. Report on the eighteenth session, Paris, 11-15 April 1988. (1988). Rep. Stud. GESAMP, (33):56 p. Available also in French, Spanish and Russian

34. Review of potentially harmful substances. Nutrients. (1990). Rep. Stud. GESAMP, (34):40 p.

35. The evaluation of the hazards of harmful substances carried by ships: Revision of GESAMP Reports and Studies No. 17. (1989). Rep. Stud. GESAMP, (35):pag.var.

36. Pollutant modification of atmospheric and oceanic processes and climate: some aspects of the problem. (1989). Rep. Stud. GESAMP, (36):35 p.

37. Report of the nineteenth session, Athens, 8-12 May 1989. (1989). Rep. Stud. GESAMP, (37):47 p. Available also in French, Spanish and Russian

38. Atmospheric input of trace species to the world ocean. (1989). Rep. Stud. GESAMP, (38):111 p.

39. The state of the marine environment. (1990). Rep. Stud. GESAMP, (39):111 p. Available also in Spanish as Inf. Estud.Progr.Mar.Reg.PNUMA, (115):87 p.

40. Long-term consequences of low-level marine contamination: An analytical approach. (1989). Rep. Stud. GESAMP, (40):14 p.

41. Report of the twentieth session, Geneva, 7-11 May 1990. (1990). Rep. Stud. GESAMP, (41):32 p. Available also in French, Spanish and Russian

42. Review of potentially harmful substances. Choosing priority organochlorines for marine hazard assessment. (1990). Rep. Stud. GESAMP, (42):10 p.

43. Coastal modelling. (1991). Rep. Stud.GESAMP, (43):187 p.

44. Report of the twenty-first session, London, 18-22 February 1991. (1991). Rep. Stud. GESAMP, (44):53 p. Available also in French, Spanish and Russian

45. Global strategies for marine environmental protection. (1991). Rep. Stud. GESAMP, (45):34 p.

46. Review of potentially harmful substances. Carcinogens: their significance as marine pollutants. (1991). Rep. Stud. GESAMP, (46):56 p.

47. Reducing environmental impacts of coastal aquaculture. (1991). Rep. Stud. GESAMP, (47):35 p.

48. Global changes and the air-sea exchange of chemicals. (1991). Rep. Stud. GESAMP, (48):69 p.

49. Report of the twenty-second session, Vienna, 9-13 February 1992. (1992). Rep. Stud. GESAMP, (49):56 p. Available also in French, Spanish and Russian

50. Impact of oil, individual hydrocarbons and related chemicals on the marine environment, including used lubricant oils, oil spill control agents and chemicals used offshore. (1993). Rep. Stud. GESAMP, (50):178 p.

51. Report of the twenty-third session, London, 19-23 April 1993. (1993). Rep. Stud. GESAMP, (51):41 p. Available also in French, Spanish and Russian

52. Anthropogenic influences on sediment discharge to the coastal zone and environmental consequences. (1994). Rep. Stud. GESAMP, (52):67 p.

53. Report of the twenty-fourth session, New York, 21-25 March 1994. (1994). Rep. Stud. GESAMP, (53):56 p. Available also in French, Spanish and Russian

54. Guidelines for marine environmental assessment. (1994). Rep. Stud. GESAMP, (54):28 p.

55. Biological indicators and their use in the measurement of the condition of the marine environment. (1995). Rep. Stud. GESAMP, (55):56 p. Available also in Russian

56. Report of the twenty-fifth session, Rome, 24-28 April 1995. (1995). Rep. Stud. GESAMP, (56):54 p. Available also in French, Spanish and Russian

57. Monitoring of ecological effects of coastal aquaculture wastes. (1996). Rep. Stud. GESAMP, (57):45 p.

58. The invasion of the ctenophore Mnemiopsis leidyi in the Black Sea. (1997). Rep. Stud. GESAMP, (58):84 p.

59. The sea-surface microlayer and its role in global change. (1995). Rep. Stud. GESAMP, (59):76 p.

60. Report of the twenty-sixth session, Paris, 25-29 March 1996. (1996). Rep. Stud. GESAMP, (60):29 p. Available also in French, Spanish and Russian

61. The contributions of science to integrated coastal management. (1996). Rep. Stud. GESAMP, (61):66 p.

62. Marine biodiversity: patterns, threats and development of a strategy for conservation. (1997). Rep. Stud. GESAMP, (62):24 p.

63. Report of the twenty-seventh session, Nairobi, 14-18 April 1997. (1997). Rep. Stud. GESAMP, (63):45 p. Available also in French, Spanish and Russian

64. The revised GESAMP hazard evaluation procedure for chemical substances carried by ships. (2002). Rep. Stud. GESAMP, (64):121 p.

65. Towards safe and effective use of chemicals in coastal aquaculture. (1997). Rep. Stud. GESAMP, (65):40 p.

66. Report of the twenty-eighth session, Geneva, 20-24 April 1998. (1998). Rep. Stud. GESAMP, (66):44 p.

67. Report of the twenty-ninth session, London, 23-26 August 1999. (1999). Rep. Stud. GESAMP, (67):44 p.

68. Planning and management for sustainable coastal aquaculture development. (2001). Rep. Stud. GESAMP, (68):90 p.

69. Report of the thirtieth session, Monaco, 22-26 May 2000. (2000). Rep. Stud. GESAMP, (69):52 p.

70. A sea of troubles. (2001). Rep. Stud. GESAMP, (70):35 p.

71. Protecting the oceans from land-based activities - Land-based sources and activities affecting the quality and uses of the marine, coastal and associated freshwater environment.(2001). Rep. Stud. GESAMP, (71):162p.

72. Report of the thirty-first session, New York, 13-17 August 2001. (2002). Rep. Stud. GESAMP, (72):41 p.

73. Report of the thirty-second session, London, 6-10 May 2002. (in preparation). Rep. Stud. GESAMP, (73)

74. Report of the thirty-third session, Rome, 5-9 May 2003 (2003) Rep. Stud. GESAMP, (74):36 p.

75. Estimations of oil entering the marine environment from sea-based activities (2007), Rep. Stud. GESAMP, (75):96 p.

76. Assessment and communication of risks in coastal aquaculture (2008). Rep. Stud. GESAMP, (76):198 p.

77. Report of the thirty-fourth session, Paris, 8-11 May 2007 (2008), Rep. Stud. GESAMP, (77):83 p.

78. Report of the thirty-fifth session, Accra, 13-16 May 2008 (2009), Rep. Stud. GESAMP, (78):73 p.

79. Pollution in the open oceans: a review of assessments and related studies (2009). Rep. Stud. GESAMP, (79):64 p.

80. Report of the thirty-sixth session, Geneva, 28 April - 1 May 2009 (2011), Rep. Stud. GESAMP, (80):83 p.

81. Report of the thirty-seventh session, Bangkok, 15 - 19 February 2010 (2010), Rep. Stud. GESAMP, (81):74 p.

82. Proceedings of the GESAMP International Workshop on Micro-plastic Particles as a Vector in Transporting Persistent, Bio-accumulating and Toxic Substances in the Oceans (2010). Rep. Stud. GESAMP, (82):36 p.

83. Establishing Equivalency in the Performance Testing and Compliance Monitoring of Emerging Alternative Ballast Water Management Systems (EABWMS). A Technical Review. Rep. Stud. GESAMP, (83):63 p, GloBallast Monographs No. 20.

84. The Atmospheric Input of Chemicals to the Ocean (2012). Rep. Stud. GESAMP, (84) GAW Report No. 203.

85. Report of the 38th Session, Monaco, 9 to 13 May 2011 (pre-publication copy), Rep. Stud. GESAMP, (85): 118 p.

86. Report of the Working Group 37: Mercury in the Marine Environment (in prep.). Rep. Stud. GESAMP, (86).

87. Report of the 39th Session, New York, 15 to 20 April 2012 (pre-publication copy), Rep. Stud. GESAMP, (87):92 p.

88. Report of the 40th Session, Vienna, 9 to 13 September 2013, Rep. Stud. GESAMP, (88):86p.

89. Report of the 41st Session, Malmö, Sweden 1 to 4 September 2014, Rep. Stud. GESAMP, (89):90p.

90. Report of Working Group 40: Sources, fate and effects of microplastics in the marine environment: a global assessment. Rep. Stud. GESAMP (90):96 p.

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