Food and Agriculture
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615

ISSN 2070-7010

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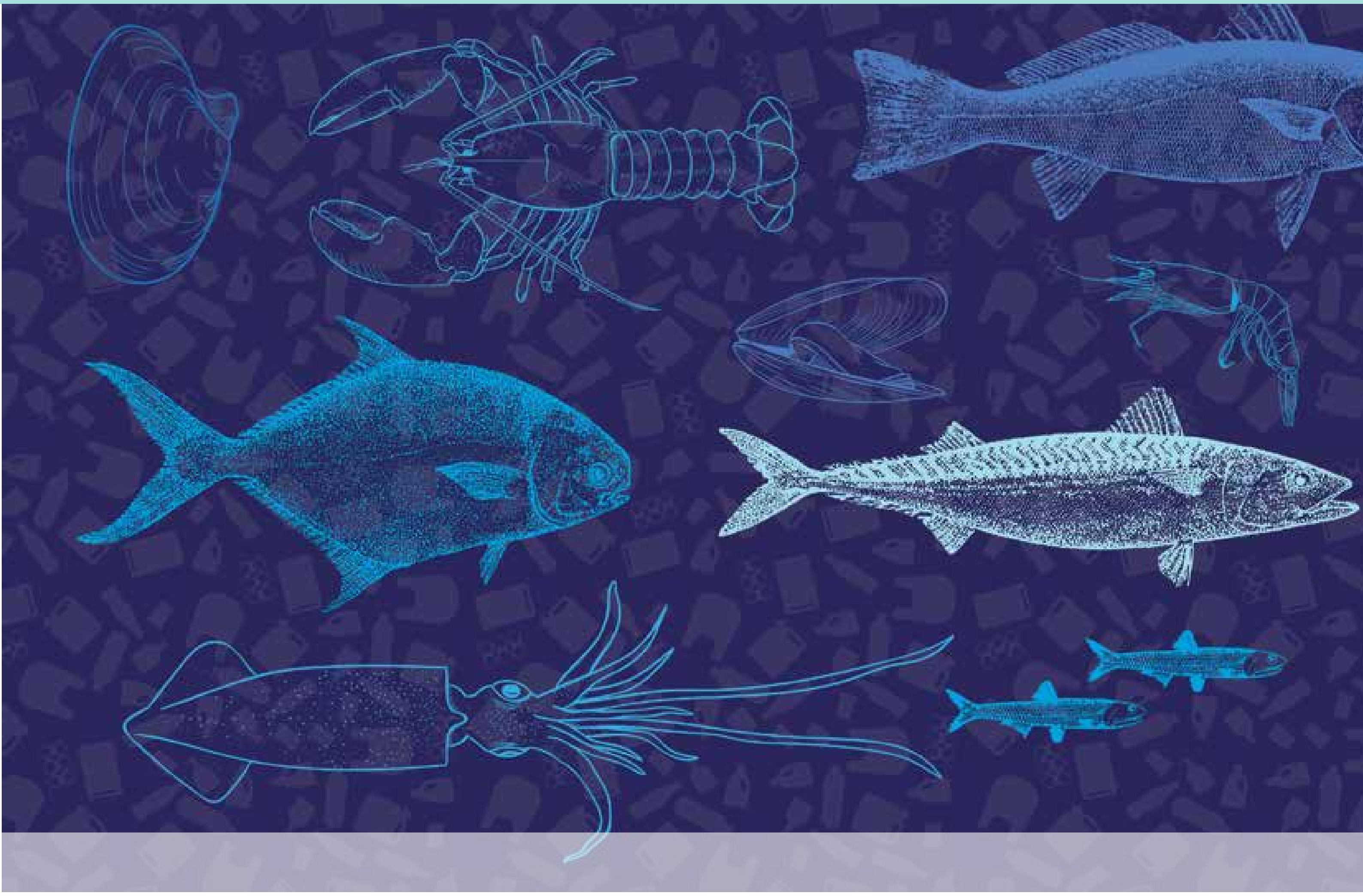
Status of knowledge on their occurrence and implications for aquatic organisms and food safety



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Rome, 2017

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ISBN 978-92-5-109882-0

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Preparation of this document

One of the recommendations of the GLOBAL OCEANS ACTION SUMMIT FOR FOOD SECURITY AND BLUE GROWTH, 22 to 25 APRIL 2014 (http://www.globaloceansactionsummit.com/) requested that the Food and Agriculture Organization of the United Nations (FAO), The International Maritime Organization (IMO) and the United Nations Environment Programme (UNEP) work together with the Group of Experts on the Scientific Aspects of Marine Environmental Protection (GESAMP[1]) to improve the knowledge base on microplastics in the marine environment and provide policy advice on this topic.

As a result, UNEP approached GESAMP, FAO and other partners with a proposal to contribute to the global assessment on sources, fate and impacts of microplastics on the marine environment and resources with funding provided by the Government of Norway.

FAO was requested to contribute specifically on fisheries and aquaculture; the two main concerns were to assess the potential impact of microplastics on consumers' health and perception, and understand the potential consequences on fish productivity as physiological processes are likely to be affected by microplastics (because of their occurrence and of the presence of additives and contaminants contained in the plastic). Samples collected so far and scientific literature increasingly and clearly show occurrence of microplastics in fish and shellfish as well as in fisheries and aquaculture products. This report is meant to contribute to take stock of the scientific knowledge available, provide information on the most likely pathways in terms of sources, transport and distribution in both marine food chains and seafood value chains, and provide a framework to assess the risks that may (or not) affect commercial fish stocks and consumers. In addition, the report looks into the current practices and limitations of microplastic sampling techniques.

The report is based on scientific literature published in international journals, as well as expert knowledge. Amy Lusher, Peter Hollman and Jeremy Mendoza-Hill produced the first draft of the report, with the support of Natalie Welden. The draft was then discussed by a group of invited experts during a workshop held in Rome from 5 to 8 December 2016. The experts reviewed the initial draft and provided additional sections and references to produce an advanced draft that was submitted to reviewers.

It is to be noted that given the high and rapid production of literature, taking stock of current knowledge on microplastics is challenging. Therefore, this report needs to be considered as a work in progress and might require periodic updates; however, in the time being, the publication will contribute to raise awareness and outreach to fisheries and aquaculture operators, scientists and policy makers to sensitize them to the issue of microplastics.

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[1] Joint IMO/FAO/UNESCO-IOC/UNIDO/WMO/IAEA/UN/UNEP/UNDP Group of Experts on the Scientific Aspects of Marine Environmental Protection. http://www.gesamp.org/

Abstract

Plastic production has increased exponentially since the early 1950s and reached 322 million tonnes in 2015, this figure does not include synthetic fibres which accounted for an additional 61 million tonnes in 2015. It is expected that production of plastics will continue to increase in the foreseeable future and production levels are likely to double by 2025. Inadequate management of plastic waste has led to increased contamination of freshwater, estuarine and marine environments. It has been estimated that in 2010 between 4.8 million to 12.7 million tonnes of plastic waste entered the oceans. Abandoned, lost or otherwise discarded fishing gears (ALDFG) are considered the main source of plastic waste by the fisheries and aquaculture sectors, but their relative contribution is not well known at regional and global levels.

Microplastics are usually defined as plastic items which measure less than 5 mm in their longest dimension, this definition includes also nanoplastics which are particles less than 100 nanometres (nm) in their longest dimension. Plastic items may be manufactured within this size range (primary micro- and nanoplastics) or result from the degradation and fragmentation of larger plastic items (secondary micro- and nanoplastics). Microplastics may enter aquatic environments through different pathways and they have been reported in all environmental matrices (beaches, sediments, surface waters and water column).

Ingestion of microplastics by aquatic organisms, including species of commercial importance for fisheries and aquaculture, has been documented in laboratory and field studies. In certain field studies it has been possible to source ingested microplastics to fisheries and aquaculture activities.

Microplastics contain a mixture of chemicals added during manufacture, the so-called additives, and efficiently sorb (adsorb or absorb) persistent, bioaccumulative and toxic contaminants (PBTs) from the environment. The ingestion of microplastics by aquatic organisms and the accumulation of PBTs have been central to the perceived hazard and risk of microplastics in the marine environment.

Adverse effects of microplastics ingestion have only been observed in aquatic organisms under laboratory conditions, usually at very high exposure concentrations that exceed present environmental concentrations by several orders of magnitude. In wild aquatic organisms microplastics have only been observed within the gastrointestinal tract, usually in small numbers, and at present there is no evidence that microplastics ingestion has negative effects on populations of wild and farmed aquatic organisms.

In humans the risk of microplastic ingestion is reduced by the removal of the gastrointestinal tract in most species of seafood consumed. However, most species of bivalves and several species of small fish are consumed whole, which may lead to microplastic exposure. A worst case estimate of exposure to microplastics after consumption of a portion of mussels (225 g) would lead to ingestion of 7 micrograms (µg) of plastic, which would have a negligible effect (less than 0.1 percent of total dietary intake) on chemical exposure to certain PBTs and plastic additives.

Microplastic contamination of aquatic environments will continue to increase in the foreseeable future and at present there are significant knowledge gaps on the occurrence in aquatic environments and organisms of the smaller sized microplastics (less than 150 µm), and their possible effects on seafood safety. Currently there are no methods available for the observation and quantification of nanoplastics in aquatic environments and organisms.

Lusher, A.L.; Hollman, P.C.H.; Mendoza-Hill, J.J. 2017.
Microplastics in fisheries and aquaculture: status of knowledge on their occurrence and implications for aquatic organisms and food safety.
FAO Fisheries and Aquaculture Technical Paper. No. 615. Rome, Italy.

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Acknowledgements

United Nations Environment Programme (UNEP) and the Government of Norway funded the work that led to the publication of this report through the project EP/GLO/662/UEP "Improve knowledge base on microplastics: occurrence in fisheries and aquaculture and potential impacts on food safety".

The authors wish to thank the reviewers who provided comments and valuable inputs: Bong Jin Cha (National Fisheries Research and Development Institute, Seoul, Republic of Korea), Karen Mackay (European Food Safety Authority. Biological Hazards and Contaminants Unit, Parma, Italy), Francesco Regoli (Dipartimento di Scienze della Vita e dell'Ambiente, Università Politecnica delle Marche, Ancona, Italy), Aileen Tan Shau-Hwai (School of Biological Sciences, Universiti Sains Malaysia, Penang, Malaysia).

Within the FAO Fisheries and Aquaculture Department, this work was coordinated by Tarub Bahri and supported by Uwe Barg (FAO Technical Secretary of GESAMP), Esther Garrido-Gamarro and Jogeir Toppe. Vittorio Fattori and Markus Lipp from the Food Safety Quality Unit (AGFF) at the Agriculture and Consumer Protection Department of FAO participated in the Rome workshop and collaborated in document review. Heidi Savelli, Global Programme of Action, UNEP, contributed to the launching of the project within the inter-agency cooperation between UNEP and FAO.

Magda Morales prepared the layout design for printing and contributed to the final quality control. Valerie Schneider and Paolo De Donno followed administrative aspects during the implementation of the project.

Abbreviations and acronyms

ALDFG	Abandoned, Lost or otherwise Discarded Fishing Gear
EFSA	European Food Safety Authority
FAO	Food and Agriculture Organization of the United Nations
FT-IR	Fourier Transform Infrared spectroscopy
GESAMP	Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection
JECFA	Joint FAO/WHO Expert Committee on Food Additives
PBTs	Persistent, Bioaccumulative and Toxic Compounds
POPs	Persistent Organic Pollutants
SD	Standard deviation
UNEA	United Nations Environment Assembly
UNEP	United Nations Environment Programme
WHO	World Health Organization

COMMON POLYMERS

ABS	Acrylonitrile butadiene styrene
AC	Acrylic
EP	Epoxy resin (thermoset)
EPS	Expanded polystyrene
HDPE	Polyethylene high density
LDPE	Polyethylene low density
LLDPE	Polyethylene linear low density
PA	Polyamide (Nylon) 4, 6, 11, 66
PC	Polycarbonate
PCL	Polycaprolactone
PE	Polyethylene
PET	Polyethylene terephthalate
PGA	Poly (glycolic acid)
PLA	Poly (lactide)
PMMA	Poly(methyl methacrylate)
PP	Polypropylene
PS	Polystyrene
PU	Polyurethane (also abbreviated as PUR)
PVA	Polyvinyl alcohol
PVC	Polyvinyl chloride
SBR	Styrene-butadiene rubber
TPU	Thermoplastic polyurethane

COMMON CHEMICAL ADDITIVES IN PLASTICS

BFRs	Brominated flame retardants
BPA	Bisphenol A
BPF	Bisphenol F
BPS	Bisphenol S
DBP	Dibutyl phthalate
DEP	Diethyl phthalate
DEHP	Di-(2-ethylhexyl)phthalate
FRs	Flame retardants
HBCD	Hexabromocyclododecane
NP	Nonylphenol
NPE	Nonyl phenol ethoxylate
PBDEs	Polybrominated diphenyl ethers (penta, octa and deca forms)
Phthalates	Phthalate esters
TBBPA	Tetrabromobisphenol

EXAMPLES OF ORGANIC CONTAMINANTS OFTEN DESCRIBED SORBED BY PLASTICS

DDT	Dichlorodiphenyltrichloroethane
HCHs	Hexachlorocyclohexane
PAHs	Polycyclic aromatic hydrocarbons
PCBs	Polychlorinated biphenyls

Executive summary

This report focuses on the present state of knowledge on the occurrence and effects of microplastics (i.e. plastic particles less than 5 mm in their longest dimension) on aquatic organisms, especially commercially important species, as well as the possible implications for seafood safety and security. The widespread use of plastic materials in the fisheries and aquaculture sectors, and the sources of microplastic contamination, particularly those derived from fisheries and aquaculture activities, are also covered. The report is intended for a wide audience including fisheries scientists and managers, health authorities, fisheries and aquaculture associations, environmental and fisheries ministries, regional fisheries bodies and regional seas organizations.

Production of fishery and aquaculture has increased approximately eightfold since FAO started to compile data from member countries in 1950. Production of fishery and aquaculture products has outpaced human population growth during the last five decades. In 2015, production reached a record high of around 170 million tonnes of animal products, without considering an addition of approximately 29 million tonnes from farmed aquatic plants. Since the 1980s, most of this growth in production has been achieved from aquaculture activities. The increased production of fisheries and aquaculture products has resulted in greater global per capita consumption. In 2013, these products represented around 17 percent of animal protein intake by the world´s population. Consumption of fish and fisheries products have well established health benefits due to their unique nutritional composition, but in some cases fish accumulate significant levels of contaminants from the environment, resulting in some fish products being potentially harmful depending on the amount consumed. Emerging concerns such as the potential human health impacts of microplastics in seafood should not be seen in isolation, but in the context of the health benefits derived from seafood consumption.

Plastic production has been increasing exponentially since the early 1950s and it reached 322 million tonnes in 2015. Market demand for plastic products is expected to continue to increase and projections indicate production levels may reach around 600 million tonnes by 2025 and to exceed one billion tonnes by 2050. Plastic is a catch-all term used to describe a range of polymer materials that are moulded under specific temperature and pressure, and have different properties depending on the requirements of the end product. Plastic polymers are highly diversified with around twenty distinct groups, as product requirements and applications evolve so will the types of plastic materials. Depending on the desired properties of the final product, the polymers can be mixed with different additives to enhance their performance, such as plasticizers, antioxidants, flame retardants, ultraviolet stabilizers, lubricants and colourants. The most common additives used in the fabrication processes are phthalates, bisphenol A (BPA), nonylphenol (NP) and flame retardants (FRs).

The development of fisheries and aquaculture has relied heavily on plastic use and is likely to continue doing so in the foreseeable future. Ropes and netting made from synthetic fibres offer greater strength and durability at a lesser weight when compared to natural fibres. Plastic materials are used in boat construction (including painting and anti-fouling coats), boat maintenance, fishing gears (gill nets, trawl nets, dredge nets, traps, floats, lures, hook and lines), fish hold insulation and fish crates. In aquaculture plastic materials are used for seafood packaging and transportation, ropes, floats, fish crates and boxes, fish cages, pond lining, fish feeders and fish tanks.

Mariculture structures (primarily made of plastics) are kept afloat by buoyant plastics (often Expanded polystyrene (EPS) or plastic buoys) and held in place with lines and ropes (mostly non-buoyant plastic lines). In the fisheries and aquaculture sector it is considered that abandoned, lost or otherwise discarded fishing gears (ALDFG) are the main source of plastic waste into the marine environment, but there is substantial spatial variability in their distribution and abundance. At present there are no current global estimates of the contribution of fisheries and aquaculture to total plastic waste in aquatic environments.

As stated above, the most widely used definition is that microplastics are particles less than 5 mm in their longest dimension. Plastic particles are manufactured within this size range for industrial purposes (primary microplastics) or are generated by the degradation and fragmentation of plastic products and items (secondary microplastics). The size range defined has been adopted in practical terms as it is considered the size under which ingestion by many species of biota occurs. On the other hand, nanoplastics, which are captured under the traditional definition of microplastics, are defined as plastic particles ranging from 0.001 micrometres (µm) to 0.1 µm.

Microplastics have been documented in many aquatic habitats of inland waters, the open-ocean and enclosed seas, including beaches, surface waters, the water column and the deep seafloor. In oceans, the small size and low density of microplastics contributes to their widespread transport across large distances particularly by ocean currents. Oceanic transport can move buoyant microplastics to distant shorelines or entrained particles can accumulate in central ocean regions. In areas of coastal mariculture and fishing, these activities may be responsible for the presence of microplastics, whereas the sources of microplastics in offshore fishing grounds may be harder to interpret because of the influence of oceanic distribution.

Microplastics and larger macroplastic items are associated with a mixture of chemicals added during manufacturing (such as, plasticizers, antioxidants, flame retardants, ultraviolet stabilizers, lubricants and colourants) or accumulated from the surrounding environment (such as, persistent, bioaccumulative and toxic substances (PBTs), including Persistent Organic Pollutants (POPs) as defined under the Stockholm Convention).

The ubiquitous presence of microplastics raises concerns regarding interaction with biota and potential contamination of the human food supply. This concern has led to a number of exposure and toxicological studies under laboratory conditions. These studies have confirmed that a diverse array of aquatic organisms, across trophic levels, can ingest microplastics. This includes protists, annelids, echinoderms, cnidaria, amphipods, decapods, isopods, molluscs and fish. Also, trophic transfer of microplastics has been observed in several studies under laboratory conditions. However, it is not likely that trophic transfer of microplastics will lead to accumulation, because most microplastics will not translocate into the tissues of their hosts. Metabolic and negative physiological responses from microplastic ingestion in aquatic organisms have only been observed under laboratory conditions after exposure to very high levels.

Observations of microplastic uptake by wildlife have been reported in a range of habitats, including the sea surface, water column, benthos, estuaries, beaches and aquaculture. Over 220 different species have been found to ingest microplastic debris *in natura*. Excluding birds, turtles and mammals, 55 percent are species (invertebrates to fish) of commercial importance, such as: mussels, oysters, clams, common shrimp (*Crangon crangon*), Norway lobster (*Nephrops norvegicus*), anchovies, sardines, Atlantic herring (*Clupea harengus*), Atlantic and chub mackerels, scads, Blue whiting (*Micromesistius poutassou*), Atlantic cod (*Gadus morhua*), common carp (*Cyprinus carpio*) and Acoupa weakfish (*Cynoscion acoupa*), among others. At present there is no direct evidence of trophic transfer of microplastics in wild populations. Additionally, field observations show no evidence of negative effects from microplastic ingestion at the population or community levels in aquatic organisms.

Microplastics have been found in various types of human food (e.g. in beer, honey, sugar and table salt) and the majority of the reports have studied their occurrence in seafood. Thus, seafood appears to be the most understood source of microplastics to humans. At present, there are no data on the occurrence of nanoplastics in foods because analytical methods to identify nanoplastics remain to be developed.

Adverse human health effects from micro- and nanoplastics in seafood may be caused by the plastic particles themselves, or by additives and sorbed contaminants, such as persistent, bioaccumulative, and toxic substances (PBTs). Human intake of microplastics from seafood (i.e. mussels) has been estimated to equal anywhere from 1 particle per day to 30 particles per day depending on seafood consumption habits and exposure of organisms to microplastics. In microplastic exposed aquatic organisms the digestive tract contains the largest quantities of microplastics. However, seafood innards are normally discarded before human consumption, except for bivalves, echinoderms and some species of small fish. As an example, a worst case estimate of exposure to microplastics after consumption of a portion of mussels (225 g) would be 7 µg of plastics. Based on this estimate and considering the highest concentrations of additives or contaminants reported in microplastics, and complete release from the microplastics upon ingestion, the microplastics will have a negligible effect on the exposure to PBTs and additives, as this contaminant burden is equivalent to less than 0.1 percent of the total dietary exposure of humans to these compounds.

Plastic contamination of aquatic environments will continue to increase, resulting in growing amounts of micro- and nanoplastics in these environments. There is some basic knowledge on the occurrence of microplastics in aquatic environments, organisms and seafood, but details are still lacking. Gaps in the occurrence of microplastics include details on entry rates and global distribution in aquatic environments and organisms, their distribution in the water column, and the specific contribution of the fisheries and aquaculture sectors to microplastic contamination. Trophic transfer of microplastics will not lead to accumulation in seafood, and associated PBTs and additives have a negligible effect on the total human dietary intake of these compounds. In contrast, basic knowledge on nanoplastics is still lacking. Data on nanoplastics are essential, because there is concern that nanoplastics may have a high biological impact.

1. General introduction

1.1 CONTEXT

1.1.1 Fisheries and aquaculture production

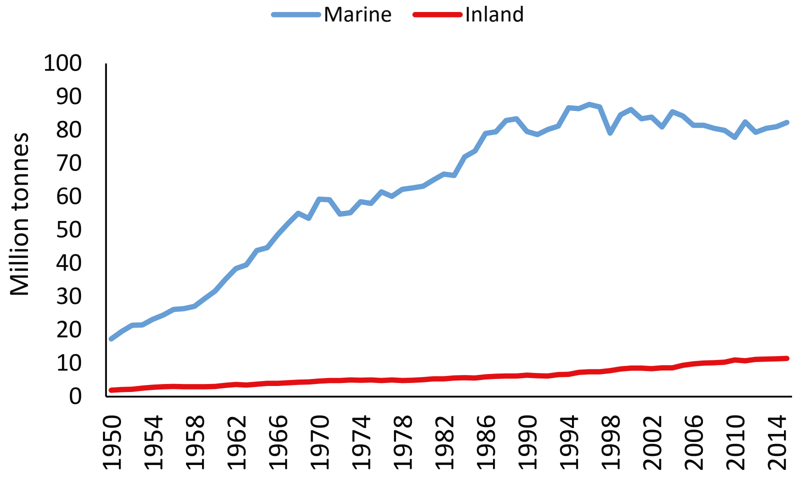
Production of fishery and aquaculture has increased approximately eightfold since FAO started to compile data from member countries in 1950. This increased production has outpaced human population growth during the last five decades (FAO, 2016b). In 2015, production reached a record high of around 170 million tonnes of animal products, without considering an additional of approximately 29 million tonnes from farmed aquatic plants. Since the 1980s, most of this growth has been achieved from aquaculture activities. Additionally, approximately 57 million persons were engaged in the primary sector of fisheries and aquaculture, and the proportion working in the primary aquaculture sector has increased from 17 percent in 1990 to 33 percent in 2014 (FAO, 2016b). Moreover, the international trade of fishery and aquaculture plays a significant role in employment, food supply, income, and contributes to economic growth and development. International trade of seafood represents around 9 percent of total agricultural exports and about 1 percent of total world trade in value terms (FAO, 2016b).

Marine capture fisheries

Marine capture fisheries reached a maximum of approximately 88 million tonnes in 1996 and have since fluctuated around a mean value of 82 million tonnes, reaching 82.3 million tonnes in 2015. In comparison, inland capture fisheries have been increasing steadily for most of the time series available and showed a maximum value of approximately 11.5 million tonnes in 2015 (Figure 1.1). It has been estimated that around 260 million people are involved in global marine capture fisheries, including full-time and part-time jobs in the direct (primary) and indirect (secondary and ancillary) sectors (Teh & Sumaila, 2013).

In marine fisheries, the most productive areas are the Northwest Pacific, the Central Western Pacific and the Northeast Atlantic accounting for 27.2 percent, 15.3 percent and 11.4 percent of total catches in 2015, respectively. Major fishing nations in 2015 are represented by China (15.6 million tonnes), Indonesia (6.1 million tonnes), United States of America (5.0 million tonnes), Peru (4.8 million metric tonnes), Russian Federation (4.2 million tonnes), India (3.5 million tonnes) and Japan (3.5 million tonnes). These seven nations accounted for approximately 52 percent of global marine fisheries landings in 2015.

FIGURE 1.1
Marine and inland capture fisheries landings from 1950 to 2015



Source: FAO.

In 2015, finfishes dominated marine capture fisheries with 67.4 million tonnes, followed by crustaceans (6 million tonnes), cephalopods (4.7 million tonnes) and other molluscs (2.4 million tonnes). There are presently 2 033 species records in the FAO database for capture fisheries, however, 25 major species and genera account for about 41 percent of global marine capture fisheries production (FAO, 2016b). Among pelagic fishes the most important species or genera in 2015 FAO statistics were anchoveta (*Engraulis ringens*), skipjack tuna (*Katsuwonus pelamis*), *Sardinella* spp., chub mackerel (*Scomber japonicus*) and Atlantic herring (*Clupea harengus*). Main demersal fish species were represented by Alaska pollock (*Theragra chalcogramma*) and Atlantic cod (*Gadus morhua*). The two most important fisheries for crustacean species in 2015 were for Gazami crab (*Portunus trituberculatus*) and Akiami shrimp (*Acetes japonicus*). Cephalopod catches consisted mainly of jumbo flying squid (*Dosidicus gigas*) and Argentine shortfin squid (*Illex argentinus*).

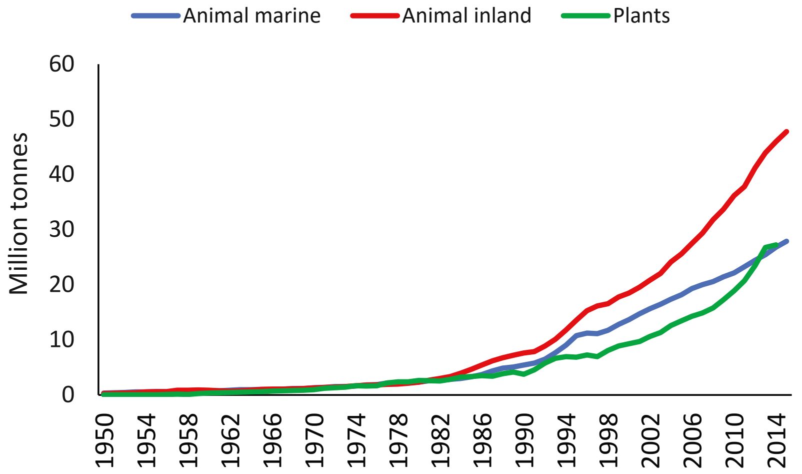
Aquaculture

Aquaculture animal production increased from around 0.6 million tonnes in 1950 to 4.7 million tonnes in 1980, since then increases in demand and investments have resulted in significant growth of this sector which reached around 77 million tonnes of aquatic animals in 2015 (Figure 1.2). It is estimated that around 18.7 million people are employed in the primary sector of aquaculture, of which approximately 84 percent are in Asian countries. The aquaculture sector is highly diverse and fragmented, ranging from small ponds in the developing world to international companies with an annual turnover in excess of US$ 1 billion (Bostock *et al.*, 2010). A total of 543 species or species groups of animals are present in the FAO aquaculture database, including 362 finfish, 104 molluscs, 62 crustaceans, 6 species of amphibians and reptiles, and 9 aquatic invertebrates (FAO, 2016b). The top 25 aquaculture producing countries accounted for about 96 percent of total world production in 2014, of which the first major 5 producers were Asian countries (China, Indonesia, India, Viet Nam and Bangladesh) and accounted for 81 percent of total production (FAO, 2016b).

Inland aquaculture production amounted to about 48 million tonnes in 2015, which represented around 63 percent of total aquaculture production. Inland aquaculture production is dominated by several species of finfish, especially carps and tilapias. On the other hand, marine and coastal aquaculture produced about 28 million tonnes, of which molluscs represented around 65 percent, crustaceans 28 percent and finfish 7 percent. Main molluscan species are oysters, clams and mussels, while penaeid shrimps are the most important crustacean taxonomic group.

FIGURE 1.2

Aquaculture production of animals (marine and inland waters) and aquatic plants from 1950 to 2015



Source: FAO.

1.1.2 Plastics and microplastics

Plastic production has increased substantially since large scale industrial manufacture started in the early 1950s. Almost all aspects of daily life involve plastics. In the European Union (PlasticsEurope, 2016), for example, the main applications of plastics include: packaging (39.9 percent, much of which is single-use), building and construction (19.7 percent), automotive industry (8.9 percent), electrical and electronic (5.8 percent), agriculture (3.3 percent) and other (22.4 percent) applications (including consumer and home appliances, furniture, sport, health and safety).

One of the most appreciated qualities of plastic products is their durability. However, this quality when combined with improper waste management leads to environmental contamination on land, in freshwater and in marine environments. Plastic products will degrade slowly over time, particularly when exposed to sunlight (ultraviolet radiation) and high temperatures. This degradation will lead to the breakdown of the material into smaller sizes ranging from the macroscopic to the microscopic and eventually to presently undetectable dimensions, the nanoplastics.

Small plastic particles known as microplastics, commonly defined as being 5 mm or less in their longest dimension (Box 1.1), have been found in different environmental matrices (atmosphere, soils, freshwater and marine). In freshwater and marine environments microplastics have been found in beaches, shelf and deepwater sediments, and in surface and subsurface waters. Additionally, ingestion of microplastics has been observed in many species of aquatic organisms, including commercially important fish and invertebrates (GESAMP, 2016; Lusher, 2015).

Microplastic contamination is certainly not a new phenomenon and the first observations of their occurrence date back to the 1960s (Carpenter and Smith, 1972; GESAMP, 2015). However, it is only in the last decade that increased attention has been given to the problem by the scientific community, international organizations, governments and public media. This rise in interest has primarily been driven by environmental and human health concerns (UNEP, 2016). There are concerns that fish and fishery products may be contaminated with microplastics and their associated contaminants. These concerns are based on potential implications for food safety and human health, marketability as well as potential impacts on food security in local contexts.

It is reasonable to suspect that contamination by plastics and microplastics began immediately in the first half of the 20th century with the onset of high-volume production of diverse plastic polymers and products. Initially, microplastics were likely derived solely from abrasion, degradation, and physical breakdown. More recently, manufacturing of plastics at the micro- and nano-scale have further exacerbated environmental occurrence and potential risks. First reports of plastic contamination from debris of various sizes occurred as early as the 1960s (Rothstein, 1973; Ryan, 2015). Publication trends (GESAMP, 2015) suggest that the topic will gain further attention and is projected to peak in the 2020s (Halden, 2015).

BOX 1.1

Definitions

In this report we refer to different sizes of marine plastic debris. All possible shapes of plastic particles are considered, namely fragments, fibres/filaments, beads/spheres, films/sheets and pellets. The size definitions used in this report are:

Macroplastics: large items of marine plastic litter which are greater than 5mm in size.

Microplastics: plastic particles in the size range between 0.1 micrometres to 5 000 micrometres (µm) in their longest dimension.

Nanoplastics: plastic particles of size ranging from 1 to 100 nanometres (nm) (0.001 µm-0.1 µm).

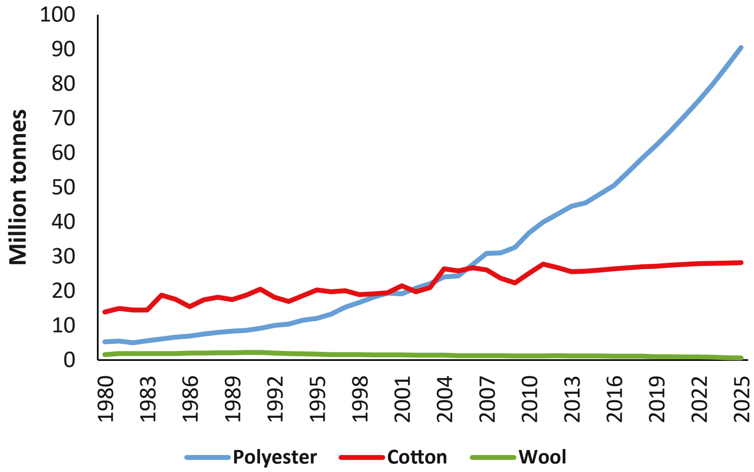
1.2 GLOBAL TRENDS

Large scale plastic production started in the early 1950s, when production levels were about two million tonnes per year, and by 2015 production of plastics reached 322 million tonnes (PlasticsEurope, 2016). Projections based on present growth rates indicate that plastic production should double by 2025 and more than triple by 2050. Synthetic fibres constitute another large part of synthetic polymer production, their worldwide production was about 61 million tonnes in 2015. Synthetic fibres have replaced natural fibres such as cotton, wool and silk because of their low price, mass production and customization (Figure 1.3).

Mass production and mass consumption of plastics have led to the accumulation of plastics in natural habitats, and adverse impacts on biota and the economy. Environmental impacts include habitat damage, entanglement and ingestion of marine litter by biota, and the introduction of non-native species, mainly microorganisms, seaweeds and invertebrates, through rafting on floating litter (Barnes and Milner, 2005; Calder *et al.*, 2014; Kiessling, Gutow and Thiel, 2015; Kühn, Rebolledo and van Franeker, 2015). The Secretariat of the Convention on Biological Diversity and the Scientific and Technical Advisory Panel-GEF (2012) reported that 373 species of wildlife had been affected through entanglement and ingestion of marine litter, among which 15 percent of species are on the International Union for Conservation of Nature (IUCN) red list. In the most recent review an additional 156 species were affected by entanglement and ingestion raising the total number to 529 species (Secretariat of the Convention on Biological Diversity, 2016). Economic impacts of marine litter include loss of income, cost of cleanup, reduced fisheries stock or reduced tourism (UNEP, 2016). Marine litter can also cause navigational hazards for fisheries and shipping.

FIGURE 1.3

Global fibre production and projections until the year 2025

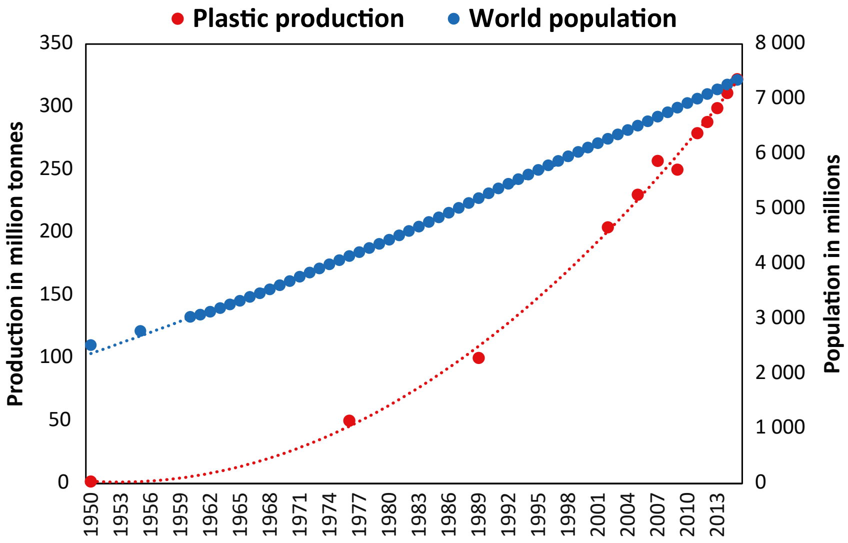


Source: Tecnon OrbiChem.

Plastic production and its waste generation are related to human population growth. The human population has increased from approximately 3.1 billion in 1961 to around 7.3 billion in 2015 (Figure 1.4) and is expected to exceed 9 billion by 2050. The needs of this growing population will drive the plastics commodities market, as well as the demand for safe fishery and aquaculture products. Among the approximately 2.5 billion tonnes of solid waste produced globally in 2010, about 275 million tonnes was mismanaged plastic waste generated from coastal countries, and it is estimated that between 4.8 million tonnes to 12.7 million tonnes of this plastic waste entered the oceans (Jambeck *et al.*, 2015). Appropriate waste management strategies are necessary in order to mitigate the effects and impacts of plastic and microplastic pollution in coastal and aquatic habitats. Microplastic contamination is very likely to increase in the foreseeable future as a result of environmental breakdown and fragmentation of present stocks and future production of plastic items.

FIGURE 1.4

Growth in human population and the growth in plastic production. Trend lines are second degree polynomials



Source: World Bank and PlasticsEurope.

Plastic production contributes to the generation of greenhouse gases that drive climate change. For example, in Europe between four and six percent of gas and oil consumption is used to manufacture plastic materials and, in 2014, 39.5 percent of post-consumer plastic waste was used for energy recovery (PlasticsEurope, 2016). Also, it is expected that increased temperatures will modify the partitioning of environmental contaminants into different matrices (i.e. air, soil, water, bottom sediments and plastic debris) and may enhance the toxicity of certain compounds such as Persistent Organic Pollutants (POPs), but will also likely increase their rates of chemical degradation (Noyes *et al.*, 2009). As the degradation and fragmentation of plastic debris is temperature dependent increased temperatures will likely affect the generation rate of microplastic and nanoplastic particles. Additionally, global warming is expected to liberate microplastic particles presently frozen in Arctic Sea ice (Obbard *et al.*, 2014). Changes in stratification patterns of the water column due to ocean warming (Seggel, De Young and Soto, 2016) will likely affect vertical distribution of microplastics in marine environments. Furthermore, changes in precipitation patterns and increases in the intensity and frequency of storm events linked to climate change will likely influence spatial patterns and rates at which plastic debris and associated chemicals enter aquatic environments (Welden and Lusher, 2017).

1.3 PURPOSE AND TARGET AUDIENCE OF THE REPORT

Microplastics are widespread in the marine environment and have been observed in seawater, lakes, rivers, estuaries, sediments and many species of biota. Small numbers of microplastics have been reported in commercial species of finfish and shellfish from field observations, and in fishery and aquaculture products. Concerns have been raised that the presence of microplastics, and their associated chemicals, represents a risk for fish productivity, fisheries resources (and potentially adverse effects on food security) and may result in contamination of foodstuffs (i.e. implications for seafood safety). The presence of microplastics in the environment and in foodstuffs, including fishery and aquaculture products, creates a need for assessing environmental, ecological and human health risks.

This report summarizes the current state of knowledge on the occurrence and impacts of microplastics in commercial species of finfish and shellfish, in prey species, and in products of fisheries and aquaculture. It provides an assessment of potential risks to fisheries, aquaculture and human health due to the presence of microplastics and nanoplastics in the marine environment. The aim is to provide a more reliable evidence base for a wide range of organizations and stakeholders to be used in developing appropriate advice and possible measures to reduce exposure to microplastics of fish resources and consumers, if this is justified. The target audience of this report includes fisheries scientists and managers, health authorities, fisheries and aquaculture associations, environmental and fisheries ministries, regional fisheries bodies and regional seas organizations.

1.4 STRUCTURE AND SCOPE OF THE REPORT

This report presents an overview of plastic production, the types of plastics and their chemical constituents, as well as the contribution of plastics to marine litter and, particularly, the contribution of fisheries and aquaculture activities to the generation of marine litter. Additionally, the origin, characteristics and known occurrence of microplastics in aquatic environments is presented and their affinity for contaminants in aquatic environments is discussed, including how fisheries and aquaculture activities may contribute to the generation of microplastic particles. The report also covers what is known on intake and ingestion of microplastics by aquatic organisms, including commercially important species. The report briefly presents the human health benefits associated with the consumption of fishery and aquaculture products. Furthermore, the human health issues related to the presence of microplastics and nanoplastics in fishery and aquaculture products is discussed, and the risk associated with the ingestion of microplastics and nanoplastics is assessed. Key reference documents on the subject of microplastic contamination are presented in Box 1.2.

BOX 1.2

Key reference documents

AMAP, 1998. AMAP Assessment Report: Arctic Pollution Issues. Arctic Monitoring and Assessment Programme (AMAP), Oslo, Norway. xii+859 pp.

EFSA, 2016. Presence of microplastics and nanoplastics in food, with particular focus on Seafood. EFSA Journal, 14(6), 4501. 30 pp.

GESAMP, 2015. Sources, fate and effects of microplastics in the marine environment: a global assessment (Kershaw, P. J., ed.). (Joint IMO/FAO/UNESCO-IOC/UNIDO/WMO/IAEA/UN/UNEP/UNDP Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection). Rep. Stud. GESAMP No. 90, 96 p.

GESAMP, 2016. Sources, fate and effects of microplastics in the marine environment: part two of a global assessment. (Kershaw, P.J. & Rochman, C.M., eds). (IMO/FAO/UNESCO-IOC/UNIDO/WMO/IAEA/UN/UNEP/UNDP Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection). Rep. Stud. GESAMP No. 93, 220 pp.

UNEP, 2016. Marine plastic debris and microplastics – Global lessons and research to inspire action and guide policy change. United Nations Environment Programme, Nairobi.

2. Plastics: production, utilization and waste

2.1 WHAT ARE PLASTICS?

Since their first development in the 1800s, plastics have advanced to benefit every manufacturing sector from health and goods preservation, to transportation and enhancing the digital age, and are now an indispensable component of day-to-day life. At less than 100 years old, plastics are a relatively modern material compared to the traditional materials of wood, metal, stone and glass. Plastics are utilized in almost every aspect of society including: packaging, building and construction, transportation, medicine and health, sport and leisure, electronics, agriculture, design and manufacturing, and currency. Plastics have enabled technological advances, design solutions, eco-performance enhancements and monetary savings. Due to their functional properties, (such as low face-value cost disregarding external costs, strength, durability, resistance to corrosion, thermal and electric insulation), plastics have helped to make our lives easier, safer and more enjoyable.

Derived from the Greek words, πλαστικός, plastikos (meaning fit for moulding) and πλαστός, plastos (moulded), the term plastic refers to a material's ductility during manufacture. This property allows the material to be cast or shaped into numerous forms for a variety of uses. Plastics evolved through the chemical modification of natural materials to the manufacture of completely synthetic molecules. The earliest plastics were derived from organic polymers and biological materials. During the Industrial Revolution in the 1800s, the necessity for progression and advancements in production encouraged research into the manufacture of mouldable materials. Possibly the first man-made plastic was Parkesine, patented by Alexander Parkes in Birmingham, United Kingdom of Great Britain and Northern Ireland, in 1856, however, the first fully synthetic plastic, Bakelite, was created in the early 1900s; these first thermoplastics were unstable and unsuitable for many applications. After the First World War, material shortages, a need for enhanced technologies and improvements in chemical engineering, led to an expansion in the variety of plastics being produced. Mass production began fully after the Second World War, and during the 1960s and 1970s the onset of consumerism encouraged a breakaway from traditional materials to plastics (Barnes *et al.*, 2009). Plastics have displaced traditional materials because they are cheap, versatile and easy to manufacture into a variety of forms.

2.2 TYPES

Plastic is a catch-all term used to describe a range of polymer materials that are moulded under specific temperature and pressure, and have different properties depending on the requirements of the end-product. Plastics are a broad family of synthetic and semisynthetic polymers derived from fossil resources (coal, natural gas, crude oil) and organic products including cellulose, salt and renewable compounds (grains, corn, potatoes, palm, sugar beet and cane, starch, seaweed and vegetable oils). In a move to break away from conventional fossil fuels, advances in technology have led to the development of hydrocarbon based plastics using renewable resources. There have also been advances in the development of bioplastics. Both sources (fossil fuel and biomass) can be used to produce either non-biodegradable or biodegradable plastics (Box 2.1).

There are three broad categories of plastics: thermoplastics, thermosets and elastomers. Thermoplastics soften on heating and harden on cooling (e.g. polyethylene (PE), polypropylene (PP), polytetrafluoro-ethylene (Teflon), polyethylene terephthalate (PET), polyamide (PA), polyvinyl chloride (PVC) and polystyrene (PS)); whereas, thermosets never soften once they have been moulded (e.g. Epoxy resins, polyurethane (PU), polyester resins and Bakelite). Elastomers are polymers which are elastic in character: the material can return to its original shape after stretching (e.g. rubber and neoprene). Plastic polymers are highly diversified with around twenty distinct groups. Besides the well-established polymers (PE, PP, PS, PET, PVC, PU, PA), a wide variety of polymers and co-polymers (corresponding to a mixture of two or more polymers) with specific physical and chemical properties are produced (Annex 2, Table A2.1). Indeed, as product requirements and applications evolve so will the types of plastic materials. This adds a layer of complexity to their application and can complicate recycling as well as their impact on the environment.

BOX 2.1
Bioplastics and some misconceptions

Bioplastics which include biodegradable, oxo-biodegradable and bio-based plastics are a recent development in plastic production:

- Biodegradable plastics can be broken down by microorganisms into water and carbon dioxide (or methane) under specific conditions (i.e. temperature)
- Oxo-biodegradable plastics are conventional polyolefin plastics that contain small amounts of metal salts which catalyse the degradation process under favourable conditions.
- Bio-based plastics are made from biological and renewable sources, and may be more susceptible to biodegradation due to their weaker polymer construction.

These special plastics are more easily degraded than ordinary plastics and results from oxidative and cell-mediated phenomena, either simultaneously or successively; however, caution must be taken when considering bio-based plastics, as those containing variable proportions of both renewable and petrochemical carbons can also be labelled as “bio-based plastics”, but are not easily biodegradable (Tokiwa *et al.*, 2009; Sekiguchi *et al.*, 2011).

Full degradation of a plastic item implies complete breakdown and decomposition into water, carbon dioxide, methane and other non-synthetic molecules. For the large majority of plastic items, even if they disintegrate by breaking down into smaller and smaller plastic debris under the influence of weathering, the polymer itself may not necessarily fully degrade into natural chemical compounds or chemical elements under marine conditions.

In addition, it is important to note that once the polymer is synthesized, the material properties will be the same, regardless of the type of raw material used. That is why some biodegradable plastics are made from fossil fuels, and some non-biodegradable plastics are made from biomass, and vice-versa (see Annex 2, Table A2.1) (UNEP, 2015).

2.3 PLASTIC PRODUCTION

Plastics can be produced in different sizes depending on their applications. Throughout this document we will refer to macro-, micro- and nanoplastics. These size classifications (Box 1.1) have emerged in parallel to their identification as a form of marine litter (see Section 2.6). Large plastic items are formed from the melting and moulding of pre-production resin pellets, or from fibre manipulation, and have a host of applications from clothing to industrial structures. Plastics produced in the microscale include those used to form larger plastic products (pre-production resin pellets), microbeads used in cosmetics and abrasive scrubbers, and microplastic powders. Plastics which are produced in the nano scale include, biomedical products, pharmaceutical drug delivery systems and medical diagnostics (Koelmans, Besseling and Shim, 2015).

2.3.1 Fabrication processes

Monomer polymerization

Monomers, such as styrene, ethylene and propylene, are the building blocks of polymers. Plastics are manufactured using a variety of different monomers and polymers depending on the requirements of the final product (McKeen, 2008). Plastic production can be complex and requires several steps before achieving the final product (BPF, 2016). The final steps are polymerization and polycondensation. In the polymerization reaction monomers, such as ethylene and propylene, are linked to form polymer chains using a catalyst. Whereas, polycondensation is the elimination of a small molecule, such as water, to form a polymer by chemical condensation (McKeen, 2008). The polymerization resulting polymers are differentiated by their own individual properties, size and structure. The most common functional groups found in the bonds of polymers include esters, amides, ethers, imides, sulphones and urethanes.

Additive incorporation

Depending on the requirements of the final product, the polymers can be mixed with different additives to enhance their performance, such as plasticizers, antioxidants, flame retardants, ultraviolet stabilizers, lubricants and colourants in order to customize the characteristics of plastics (e.g. flexibility, strength, resistance to heat, electrical isolation, etc.) (Lithner, Larsson and Dave, 2011). The most common additives used in the fabrication processes and reported in macro- and microplastic debris collected in environmental surveys are phthalates, bisphenol A (BPA), flame retardants (FRs), among which polybrominated diphenyl ethers (PBDEs) represent the main group, and to a lesser extent nonylphenols (NP).

Phthalates

Phthalates are used as additives to make plastics more flexible and resistant to unwanted breakage (Oehlmann *et al.*, 2009). The most commonly used phthalates are Di-(2-ethylhexyl)phthalate (DEHP), Dibutyl phthalate (DBP) and Diethyl phthalate (DEP), and they are mainly used in PVC production leading to a high proportion (from 10 percent to 60 percent) of phthalates by weight (Net *et al.*, 2015). These molecules cause concern as some have the capability to act as endocrine disruptors (EDs). Indeed, phthalates are not chemically bound to plastics, so they leach from consumer products into the environment during manufacturing, use and disposal (Talsness *et al.*, 2009; Net *et al.*, 2015). As a consequence, phthalates are bioavailable to marine organisms (Cheng *et al.*, 2013; Fossi *et al.*, 2012, 2014; Gobas *et al.*, 2003; Hermabessiere *et al.*, 2017; Huang *et al.*, 2008; Oehlmann *et al.*, 2009) entering the aquatic food web (Mackintosh *et al.*, 2004) with a potential for impact.

Bisphenol A

Bisphenol A (BPA) is one of the most produced chemicals, with a worldwide production capacity of approximately 5.2 million tonnes in 2008 (Arnold *et al.*, 2013). It is primarily used as the monomeric building block of polycarbonate plastics and epoxy resin (Erickson, 2008) but it can also be used as additive in other polymers (PE, PP, PVC) (PlasticsEurope, 2016). Leaching of BPA in the aquatic environment can occur from food and drink packaging debris (Sajiki and Yonekubo, 2003) or via untreated wastewater entering the environment (Guerra *et al.*, 2015). There is some concern that BPA may act as an endocrine disrupting compound (Michałowicz, 2014; Perez-Lobato *et al.*, 2016), because it binds to the estrogen receptors α (ERα) and β (ERβ), although with an affinity four orders of magnitude lower than that of estradiol (Kuiper *et al.*, 1998) Its use is still permitted for food contact material in the European Union as the European Food Safety Authority (EFSA) considers that *"BPA poses no health risk to consumers of any age group (including unborn children, infants and adolescents) at current exposure levels"* (EFSA, 2015a). Other bisphenol compounds are used in plastics manufacturing (bisphenol F (BPF) and bisphenol S (BPS)) but their toxicity is unknown (Chen *et al.*, 2016).

Flame retardants

Flame retardants (FRs) are used as a safety feature in electronic devices, isolation foams furniture, fabrics and many other plastic items to reduce their flammability. FRs include a wide range of chemicals among which polybrominated diphenyl ethers (PBDEs) and hexabromocyclododecane (HBCD) are two of the most commonly used compounds in plastic manufacture. PBDEs and HBCD are associated with endocrine disrupting effects, teratogenicity (i.e. congenital and developmental anomalies) and/or liver and kidney toxicity (Muirhead *et al.*, 2006; Yogui and Sericano, 2009), and are listed by the Stockholm Convention as Persistent Organic Pollutants (POPs). The amount of PBDEs in some plastic products is quite high (e.g. 50 percent by weight) and they can leach to the surrounding environment as they are not chemically bound to the polymer (Engler, 2012). As a result, some microplastics debris sampled from the ocean have sporadically shown high concentrations of PBDEs (Hirai *et al.*, 2011). HBCD is an additive chemical mainly applied to polystyrene products including expanded polystyrene (EPS), extruded polystyrene foam (XPS), extruded polystyrene products, especially construction materials, and electronic housings (Rani *et al.*, 2015). A wide range of HBCDs were found in various polystyrene based consumer products, in EPS buoys used for fisheries and their marine debris (Rani *et al.*, 2014; Al-Odaini *et al.*, 2015).

Nonylphenols

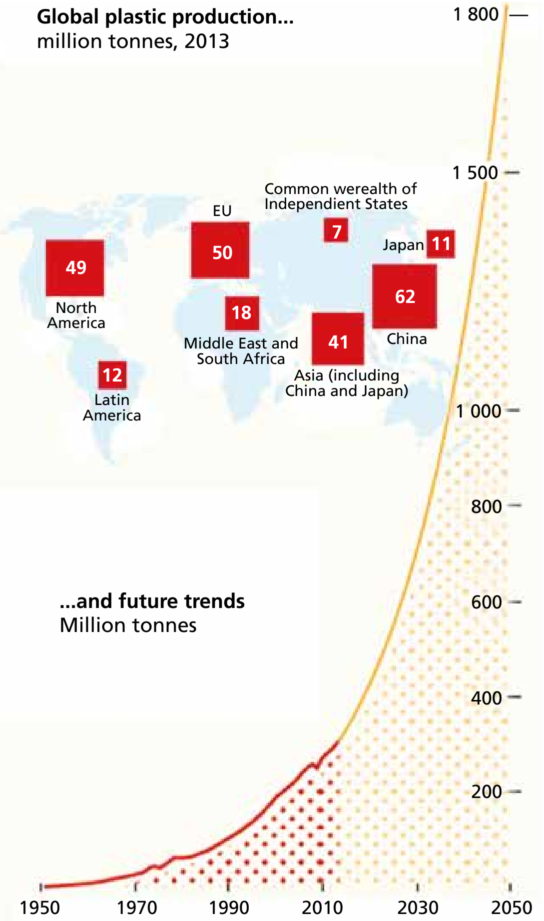
Nonylphenol ethoxylate (NPE) and its precursor and degradation product nonylphenol (NP) are organic compounds that are widely used in detergents, paints, pesticides, personal care products, and in plastics as a stabilizer in food packaging, and as an antioxidant in polymers such as rubber, vinyl, polyolefins, polystyrenes and PVC (USEPA, 2010). Major sources of NP and NPE in the environment are the effluents from wastewater treatment plants (Soares *et al.*, 2008). NPs are alleged to be endocrine disruptors and several studies showed that additive or synergistic effects can be observed from co-occurrence with other compounds such as plasticizers, BPA and PBDEs (Vethaak *et al.*, 2005; Soares *et al.*, 2008). NPs were measured in microplastics collected in remote or urban beaches and in the open ocean suggesting the ubiquitous nature of this contaminant in association with plastic debris (Hirai *et al.*, 2011).

2.3.2 Production data

Worldwide plastic production was estimated to have reached 322 million tonnes in 2015 (Figure 2.1). This value only includes plastic materials (thermoplastics and polyurethanes) and other plastics (thermosets, adhesives, coatings and sealants) but not fibres (e.g. polyacryl-fibres and polyamides). The largest producers of plastic materials (thermoplastics and polyurethane) are China, the European Union and North America, contributing to 27.8 percent, 18.5 percent and 18.5 percent of the total production, respectively (PlasticsEurope, 2016). In the last four years, global plastic production has been estimated to have increased around 3.7 percent per year. Within Europe, the total plastic demand (49 million tonnes) is dominated by six countries: Germany (24.6 percent), Italy (14.3 percent), France (9.6 percent), Spain (7.7 percent), United Kingdom (7.5 percent) and Poland (6.3 percent). It is estimated that by 2025 plastic demand will double present production levels and that by 2050 the demand will exceed three times the present levels (WEF, 2016).

FIGURE 2.1

Trends in global plastic production. The projected exponential increase is the result of predictions based on increasing population and resulting demand and forecasting from the known curve

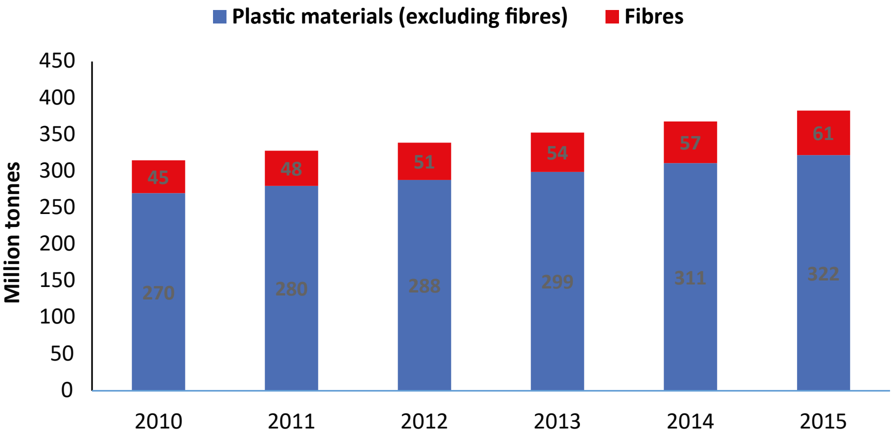


Source: UNEP and GRID-Arendal, 2016.

Petrochemical fibres, namely PET, PA, PP and polyacrylic fibres, are not generally included in the key published data on plastics production, as it is the case for the PlasticsEurope data, even though they are widely used in the fisheries (ropes, nets) and textile industries (clothes, linen, curtains, etc.) (Dris, 2017). For instance, 66.8 million tonnes of fibres were produced worldwide in 2015, of which 60.7 million tonnes (91 percent) were synthetic fibres and 6.1 million tonnes (9 percent) were artificial cellulosic fibres (Industrievereinigung Chemiefaser, 2016). By comparing these figures with the data published by PlasticsEurope, it appears that the exclusion of fibres actually causes an underestimation of 15 percent to 20 percent of the total plastics production depending on the year (Figure 2.2; PlasticsEurope, 2016; Industrievereinigung Chemiefaser, 2016). In terms of volume, the market worldwide is dominated by a limited number of well-established polymers (PE, PP, PS, PET, PVC, PU).

FIGURE 2.2

Worldwide production of plastics in millions tonnes, including plastic materials (thermoplastics, thermosets, polyurethanes, adhesives, coatings and sealants) and synthetic fibers (PA, PE, PP, PU, PET, acrylic and polyester fibers)



Source: PlasticsEurope, 2016; Industrievereinigung Chemiefaser, 2016.

2.4 USE OF PLASTICS IN FISHERIES AND AQUACULTURE

2.4.1 Use of plastics in fisheries

The plastic industry has greatly influenced the development of fisheries and aquaculture. Fisheries techniques have advanced with the industrial manufacture of plastics and equipment is now primarily made from synthetic or semisynthetic materials. When they were first introduced synthetic fibres offered greater strength and durability than natural fibre ropes, they also decreased the overall weight of the net (Valdemarsen, 2001). Synthetic fibres are cheap, durable and easier to handle than their natural counterparts. Most fisheries activities use lines, cages or nets suspended from buoyant structures. Plastic materials are used in boat construction (including painting and anti-fouling coats), boat maintenance, fishing gears (nets, trawls, dredges, traps, floats, lures, hook and lines), fish hold insulation and fish crates (FAO, 2016a). Nets and floats are made from a range of plastics, including PP, PE, PVC, PS and PA, and the choice of fishing method or gear type is critical for both its use in fishing and its impact on the environment (UNEP, 2016). Once fish are caught, plastic fish boxes and industrial packing crates are used on vessels for transportation and distribution of fish. Plastics are also used to make avoidance devices, such as pingers and streamer lines, to reduce the interactions of non-target species (e.g. seabirds and marine mammals) with the fishing gear (Dietrich, Melvin and Conquest, 2008; Gazo, Gonzalvo and Aguilar, 2008).

Trawl and dredge nets

Trawling and dredging are common worldwide fishing methods used to collect organisms from a variety of estuarine and marine habitats. Fundamentally, trawling is the process of towing a net to catch fish and invertebrates; the type and size of gear used is scaled to match the available horsepower of the fishing vessel. Nets can be towed on the seabed or at any depth in the midwater, by one or multiple vessels. Most nets are funnel shaped with a cod end, made of plastic (primarily PA, PP, PE), with floats (either spherical plastic or aluminium) on the headrope, and weights on the footrope. Dredging utilizes a scoop shaped metal frame with a holding bag. The frame is covered with a heavy chain mesh and polymer netting laced to the frame to retain the catch. Dredges are designed to be towed, or dragged, along the seabed to collect bottom dwelling species. Examples of species captured by dredging include benthic fish, scallops, oysters, clams and crabs.

Gill and seine nets

Gillnetting is a common fishing method for commercial and artisanal fisheries in oceanic, freshwater and estuarine areas. Gillnetting usually involves vertical panels of netting set in a straight line. Nets are strung between a headrope with plastic floats attached and a weighted footrope. Originally, the nets would have been made from natural fibres, but in the 1960s synthetic fibres were introduced to the fishing gear industry which led to the expansion of commercial gill net production. The synthetic fibres are cheap, durable and easier to handle than their natural counterparts. Monofilament lines are almost invisible in water and have a high tensile strength. Gill nets can be used to target pelagic, demersal and benthic species worldwide. There are several different forms of gillnets such as:

Set gill nets: a single netting wall (made primarily of plastic monofilament) kept vertical by a floating headrope and a weighted footrope. The net is set on the bottom and kept stationary by anchors or weights at both ends.

Driftnets: a string of gillnets that are kept vertical by floats (plastic or cork) on the headrope and weights on the ground-rope. They are set adrift to float on currents, either near the surface or in midwater.

Trammel nets: nets made of two/three layers of netting with a slack small mesh between the outer layers of netting in which fish entangle. They are kept vertical by floats on the headrope and weights on the ground-rope. The floats on the headrope are usually cylindrical or egg-shaped plastics.

Fixed (staked) gillnets: nets stretched between stakes that are driven into the seabed usually in coastal zones in the intertidal area.

Seine netting involves a very long net set from the shore or from boats. The length of the seine and hauling lines determines the size of the catch. A synthetic rope is used for hauling and the net is commonly made from PA and PE. This fishing method is primarily used to catch demersal species in coastal areas, lakes, rivers and in the open ocean for pelagic species (purse seines).

Traps and pots

Traps and pots are gears in which fish can enter voluntarily and will be prevented from escaping by a non-return device. They are frequently utilized to catch bottom-dwelling organisms such as crustaceans and fish. There are many different forms that were originally made from wood, bamboo and natural netting, although it is now more common for them to be produced from synthetic plastics. Traps will usually be hauled by hand on synthetic lines, but pot haulers are commonly used in deep water. Stalls, barrages or nets can be used for migrating fish species whereas pots are primarily for lobsters, crabs, shrimp, octopus and eels. These fishing methods are used worldwide in different depths of water: traps are usually set in coastal areas whereas pots can be set to a few hundred metres depth in the open ocean. There are several different forms of traps including:

Aerial traps: designed to catch jumping or gliding fish. Nets can be set from small boats and fish can be frightened by fishers into the traps.

Fyke nets: are usually cylindrical or cone shaped net bags mounted on rings or rigid structures. They can be fixed to the bottom by anchors, ballasts or stakes.

Stow nets: stationary gear, made from nets in the form of a cone or a pyramid. Usually used in areas with strong currents, they can be fixed to the seabed using anchors or stakes.

Barriers/fences/weirs/corrals, etc: usually made of a variety of material, and installed in tidal waters. They have a narrow opening leading to an enclosed catching chamber.

Pots were originally made from natural cane, but are now constructed with plastic piping frames with synthetic netting covering a plastic bucket at the entrance and a plastic base which is protected by rope or car tyres (Galbraith, Rice and Strange, 2004). In addition, creels are a type of pot used mainly for catching invertebrate species (crabs, lobsters, octopus, etc.) They are made with a steel rod frame dipped in plastics for corrosion protection and covered in synthetic netting. The apex of the net contains a funnel which is either held open with plastic rings (hard eye) or synthetic netting (soft eye). There is a hinge attached to the frame by rubber and hooks, and usually ropes and old tyres are attached to the base and sides to prevent damage.

Line gears

Longlines, handlines and pole and lines are methods of fishing in which fish are attracted to natural or artificial bait (lures) on hooks. These diverse methods can be deployed from a boat or from the shore. Depending on the scale of the fishery, the size of the handling equipment varies. Longlines consist of several strings, on a main line with baited hooks and marker buoys at each end and may be anchored on the seabed. The larger, commercial longline fisheries employ heavier main lines, often in deep water to target large demersal and pelagic species. Plastic floats are often used to support the fishing gear (Watson and Kerstetter, 2006). The different forms of line gears include:

Set longlines: baited or unbaited mainlines and snoods set on or near the bottom. The number of hooks and set distance depend on the target species and gear used.

Drifting longlines: the mainline is kept near the surface by means of regularly spaced floats with long snoods and baited hooks. Some drifting longlines are set vertically with each line hanging from a float at the surface.

Trolling line: a single line with natural or artificial baited hooks can be trailed behind a vessel. Rubber is often attached to each line to act as a shock absorber.

Handlines: can be used with or without a pole and fishing in deep water usually involves reels.

Pole and lines: a hooked line attached to a pole, common in sport fisheries and some commercial fisheries. Poles are made of wood and more recently fibreglass. The fishing line is usually a monofilament or PA.

Fish aggregating devices

In the open water, fish naturally gather under flotsam. Fish Aggregating Devices, or FADs, are permanent, semi-permanent or temporary structures, which attract and aggregate pelagic fish. These artificial objects can either be anchored or set adrift on ocean currents. The earliest FADs were natural driftwood and trees, but are now commonly constructed from synthetic materials. Usually the surface floats are made of bamboo poles, synthetic material and buoys with subsurface netting that can reach from 10 m to 300 m below the surface. To capitalize on fish aggregations, circular nets or purse seines are set.

2.4.2 Use of plastics in aquaculture

As the world's population continues to grow, demand for seafood has been increasing proportionally. Aquaculture was developed to support consumer demand for fish and shellfish, and the methods of production have continued to expand with the growing consumer market. World animal aquaculture production reached 73.8 million tonnes live weight with an estimated first-sale value of 160.2 billion US$ in 2014 (FAO, 2016b). Although shellfish have always been cultured in mariculture systems in the intertidal or semi-enclosed coastal areas, the development of the fish mariculture industry is relatively recent. Mariculture developed to move away from land-based aquaculture systems as the demand for space intensified. Originally, land based facilities were used for fish, but a lack of suitable sites inland saw an expansion into open water bodies including rivers, lakes, brackish coastal bodies and marine offshore waters. Systems for aquaculture are changing rapidly as consumer demand increases and technologies for production facilities advance: this includes structures to contain cultured organisms in open waters. The expansion of the industry and the diversity of materials used to build and maintain the culture systems have paralleled the development of synthetic polymers over the last 50 years.

Fish and bivalves can be cultured in several different ways from traditional rope cultures to intensive cage farming. Plastic materials are used for seafood packaging and transportation, ropes, floats, fish crates and boxes, fish cages, pond lining, fish feeders and fish tanks. Mariculture structures (primarily made of plastics) are kept afloat by buoyant plastics (often EPS or plastic buoys) and held in place with lines and ropes (mostly non-buoyant plastic lines). Plastics are used for cages, nets, ropes, lines and buoys: this includes from small domestic facilities to highly technical systems. Plastics are used because they are a cheap and durable material, and can be easily sourced and maintained. Aquaculture practices and systems vary widely around the world according to the species and environment.

Bivalves

Mariculture of bivalve molluscs is increasing globally, particularly in the culture of oysters, clams, scallops, mussels and cockles. The requirements of these species restrict such facilities to coastal areas, within several kilometres of the shoreline; and locations are usually chosen based on the presence of suitable conditions for spat collection and cultivation. Plastics are used in all stages of bivalve mariculture including PA ropes for line culture, and plastic crates and frames for bottom culture. Spat are collected using oyster shells or similar materials for settlement, as well as plastic collectors, and then strung on synthetic twine or monofilament PA attached to trays of bamboo or wood (Baluyut, 1989). A wide variety of materials are used for scallop spat collection including PE mesh bags, and PA or PP rope (Lovatelli, 1988). It is more common for natural rope to be used as it attracts more larvae than PE or PP, but because they break down quickly, a hybrid mixture of natural and synthetic material is often used. The nursery and on-growing of bivalves can be carried out in either suspended subtidal cultures (e.g. ropes), tressil cultures in the intertidal and by way of on/in- ground cultivation.

Bivalve mariculture in estuarine conditions involves several methods. Some species are stake and line grown (mussels and oysters), grown in plastic trays (oysters) or bottom grown (cockles and clams). The stake method is used in soft sediment and shallow waters (<1m at low tide); bamboo trunks or mangrove branches are spaced apart and used as a substrate for spat attachment. Bivalves can be set directly on rope, in bags or in plastic trays, or in the intertidal culture where shellfish are maintained in mesh bags or other plastic enclosures. Plastic mesh screens protect organisms from predators (e.g. crabs and birds) and also protect them in areas subjected to extreme heatwaves. Bottom growth can occur in the intertidal and offshore marine environment (scallops and oysters) along with deep water longline culture. Several configurations can be used for shellfish longlines, but they usually consist of lines hanging from floats, rafts or a line strung between floats. EPS floats are commonly used in East Asia to support flotation in shellfish cultures. Other buoys are made from acrylonitrile butadiene styrene (ABS). For some species, PE trays will be hung from longlines in a similar way to the hanging method for oyster culture. Baskets and collectors for shellfish at harvest are usually made of plastics (such as loops of black flat PP, knitted onto a 30 cm PE monofilament line) (Andrefouët, Thomas and Lo, 2014).

Crustaceans

As with bivalve culture, crustaceans can be cultured in ponds and intertidal environments. Crustaceans are usually maintained in plastic mesh bags or plastic lined enclosures (ponds) in estuaries. Crustaceans can be cultured on the seafloor in deep or shallow water and can include fencing and/or anti-predator nets or mesh screens made from a variety of polymers including PE and PP. Plastics are also used for nursery cages for smaller life history stages including PVC pipes for water drainage and flow, pond liners (often made of high density polyethylene (HDPE)) and mesh screens to prevent undesirable organisms entering the ponds (Park and Kim, 2013).

Fishes

The production of fish from cages is increasing globally, and technologies are now well developed in Europe, parts of South America (predominantly Chile) and Asia (predominantly China) (FAO, 2016a; Funge-Smith and Phillips, 2001). Traditionally, fish were cultured in ponds or other closed water bodies on land, or penned in estuaries. As the demand for fish and aquaculture has increased, it has stimulated the development and expansion of mariculture facilities in coastal and open water locations to compensate for the demand for space. Modern marine cages can be either floating or submersible (Cardia and Lovatelli, 2015). Fish cages and pens are generally comprised of a net stretched over a framework structure. Tanks, pens, nets, floats and pontoons are primarily made of plastic material (Kumar and Karnatak, 2014). Nets are often made from PE and PA monofilament twine and several methods can be used for flotation of cage structure including PVC and HDPE pipes and containers, plastic drums and PS floats. HDPE pipes are widely used for cage structures because of their versatility, resistance to UV degradation and cost-effectiveness. The pipes are filled with PS cylinders to increase buoyancy. Pipes can be formed in various shapes, colours and sizes and the frame held together with plastic brackets, and these frames form the main structure for the fish net pen to be secured. Gravity cages (nets) attached to the frame are held in place with weights on the bottom of the net. Nets must be maintained and controlled for chemical degradation, biofouling and corrosion, and are regularly inspected to ensure strength and stability. Cage structures may break away or be lost at sea; therefore, it is paramount that the broken or damaged nets are repaired and replaced, and that the broken item is removed from the environment before it becomes marine debris. Other plastic materials on culture cages include:

- **Ropes:** used for mooring lines, the grid system and the netting frame. They are primarily made of polysteel which is a blend of PP and PE, which has a 25 percent higher tensile strength than PP. Although, polyester and PA can be used, they are more expensive and more elastic than polysteel.
- **Nets:** common polymers for nets and ropes include PA, PP, PET and PP. PE and PP can be braided together for netting on the cage. The most common plastic rope is usually PA with UV stabilizer to reduce degradation. Without stabilizers exposure to UV will cause depolymerization, brittlement and subsequent breakages. The colour of the net can depend on the species of fish being cultured in the facilities.

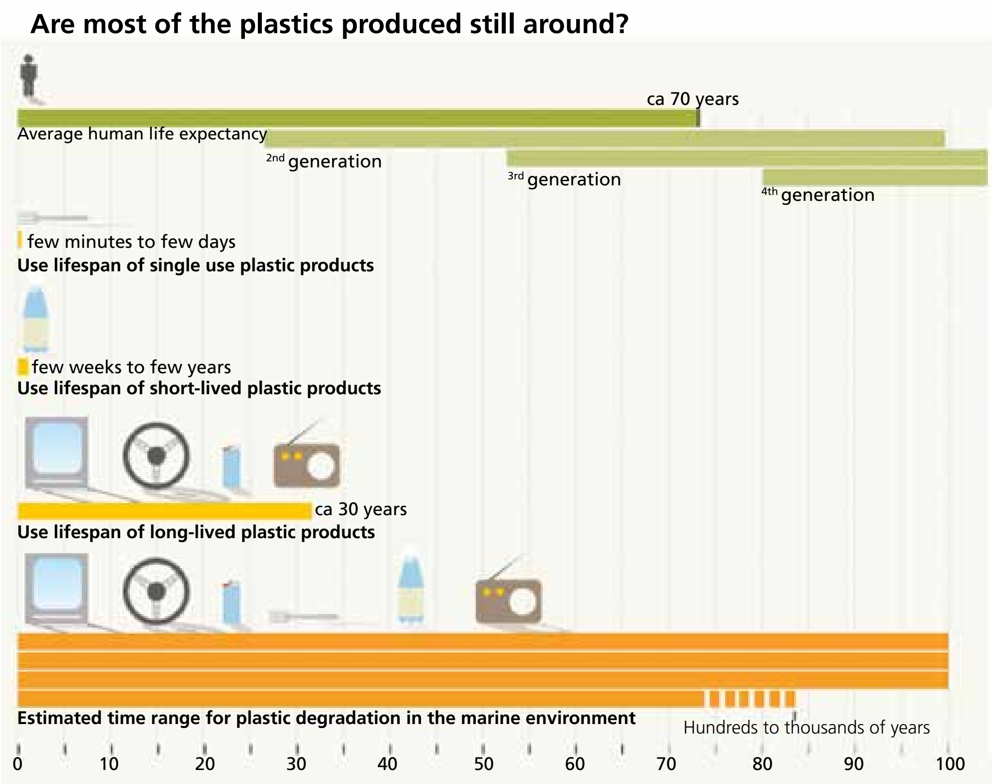
- **Marker buoys:** made of moulded PE and filled with PU or PS for additional buoyancy.

2.5 PLASTIC WASTE AND MANAGEMENT

The production and use of plastics necessitates waste management. Many countries have developed sophisticated waste management schemes; however, the waste problem is acute in developing countries (Hoornweg, Bhada-Tata and Kennedy, 2013). Thus, in this context the persistence of plastics and the short life of many disposal products pose serious problems for waste management.

Nowadays, plastics make up at least 10 percent of solid waste by mass in 58 percent (61 out of 105) of countries with available data (Hoornweg and Bhada-Tata, 2012). Plastics may be buried in landfills, recycled, incinerated (with or without energy generation), accidentally lost or deliberately littered. Except for the incinerated component of waste, it is estimated that all plastics produced since the beginning of mass plastic production remain in the environment in whole or in fragmented forms (Figure 2.3). If inappropriately handled plastic waste may escape waste streams, enter the environment and eventually reach the sea. The percentage of recycled plastics has increased every year since at least 1990, but it is far behind other materials including paper (58 percent) and iron and steel (between 70 percent and 90 percent) (WEF, 2016).

FIGURE 2.3
Lifespan of plastic items



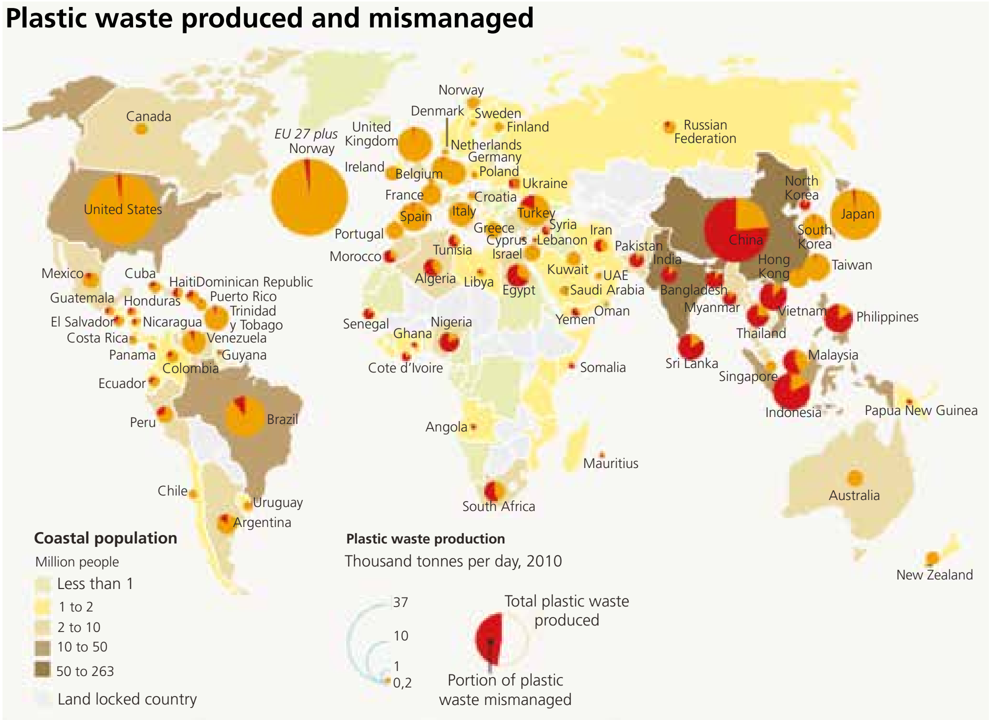
Source: UNEP and GRID-Arendal, 2016.

Several management options have been implemented to manage the waste stream of large plastic items. Current management plans for solid waste include: open dumps or tips into landfill, incineration, waste to energy and recycling (Bernardo, Simões and Pinto, 2016). However, not all plastics are recyclable or recycled which can be the result of insufficient waste streams. In Europe in 2014, 25.8 million tonnes of post-consumer waste ended up in official waste streams, 69.2 percent was recovered through recycling and energy recovery process (29.7 percent and 39.5 percent, respectively) and 30.8 percent ended up in landfill. With specific reference to plastic packaging materials (78 million tonnes in 2013), 14 percent was recycled, 14 percent incinerated, 40 percent landfilled and 32 percent reached the environment globally (WEF, 2016). Of the available waste streams, recycling is widely regarded as the preferential treatment option. It allows end of life items to have a value rather than becoming waste. Treatment of waste is different by country, some rely heavily on landfill, whereas other focus on recycling and energy production (in the form of heat, steam and electricity). However, this requires sophisticated and expensive separation infrastructure which is less available in developing countries. According to estimations, 5 of 192 countries contribute more than 50 percent to the mismanaged plastic waste (Jambeck *et al*., 2015). These are mainly developing countries, experiencing rapid economic growth, but not yet having efficient waste-management infrastructures.

Mismanaged plastic waste is either littered or inadequately disposed and could eventually enter the ocean via inland waterways, wastewater outflows and transport by wind or tides. Additionally, even if plastic litter is retained at sewage treatment plants, in conditions of heavy rainfall, sewage systems can become overwhelmed and the volume of water passing through them can force plastic items into water courses. This results in items of sewage related debris entering the waterways and reaching the ocean, evidenced by the large number of sewage related and domestic items commonly reported in marine litter surveys (e.g. Duhec *et al*., 2015). Current estimates suggest that between 4.8 million tonnes to 12.7 million tonnes of plastics entered the ocean in 2010 (Jambeck *et al*., 2015). The authors further evaluated that if no waste management infrastructure improvements are made in the next decade, the estimated quantity of mismanaged plastic waste entering the marine environment is predicted to be multiplied by ten. It must be noted that all these estimations are based on rough calculations and several assumptions whose accuracy is difficult to evaluate (Figure 2.4).

FIGURE 2.4

Worldwide Input of plastics from solid municipal waste and wastewater



Source: UNEP and GRID-Arendal, 2016.

2.6 THE MARINE LITTER PROBLEM

Defining the marine litter problem is complex as there are many different sources and forms of litter that can enter the oceans. Plastic items are consistently the most abundant type of marine debris identified around the globe, and can amount to more than 80 percent of reported debris (e.g. UNEP, 2016). Both sea and land-based activities are responsible for the continued input of plastic, making it a ubiquitous pollutant as it has been reported globally. Land-based sources include unprotected landfills and dumps located near the coast or to riverine systems, general public litter in shorelines, accidental loss, harbour activities, overflow of sewage systems, and extreme events (i.e. storms). Oceanic based plastic litter can be generated from all types of boats, ships and offshore platforms in the ocean. This can be through accidental loss, indiscriminate disposal or illegal dumping (Allsopp, Santillo and Johnston, 2006). Prior to the 1980s, the occurrence of compounded plastics in the open ocean was most probably due to the routine solid waste disposals by individual ships. However, MARPOL, the International Convention for the Prevention of Pollution from Ships, now prohibits the disposal of waste from vessels. Even with international regulation, deliberate waste emission to the ocean may still be occurring but it is almost impossible to monitor and regulate. Domestic litter, riverine litter and litter from shipping and fishing are generally considered as chronic sources, as they represent an almost continuous input. Chronic sources of mismanaged waste as well as extreme weather events contribute to a large amount of plastic waste entering the marine environment. Events can include floods, storms (high winds), cyclones, high rainfall and tsunamis (e.g. Calder *et al.*, 2014; da Silva *et al.*, 2016; Goto and Shibata, 2015; Lebreton and Borrero, 2013; Moore, Lattin and Zellers, 2011; Rech *et al.*, 2014; Swanson *et al.*, 2016).

As mentioned earlier in this report, one of the primary properties of plastics is their durability; unfortunately, this supports their persistence in the marine environment long after they have been introduced. Depending on physical properties as well as environmental conditions plastics can be found in the five different compartments of the marine environment: coastlines, water surface, the water column, the seafloor and biota. Plastic marine debris has been reported in surveys of shore based litter and oceanic litter worldwide. Movement of plastic debris items within the environment can be facilitated by their physical properties as well as environmental variables. The properties of plastic materials also influence their distribution in the environment. The density of an item in relation to that of the surrounding water can influence whether it is positively, neutrally or negatively buoyant. There are many other factors which can affect an item's buoyancy, such as entrapped water, water currents, turbulence, encrusting biota and microbial growth; often originally buoyant plastics may sink over time to be deposited on the seabed.

Coastal waters and shorelines often contain a considerable amount of plastics debris. They are subjected to anthropogenic pressures including: land based input of debris, tourism, aquaculture, shipping, fisheries and high coastal population. Beach surveys, while by far the most numerous, are often difficult to compare due to differences in surveying and recording protocols (Cole *et al.*, 2011). Most studies demonstrate densities in the 1 item per m^2 range, except for those with very high concentrations as a result of local conditions, after storms or flooding events (Browne *et al.*, 2015a; Galgani, Hanke and Maes, 2015; Liu, Wang and Chen, 2013; Topçu *et al.*, 2013).

Open ocean surface waters have persistent ocean currents which passively transport floating items. Buoyant plastics are also moved by winds and generally items accumulate in convergence zones, washed far from their point of entry. Floating marine debris can reach densities over 600 items per km^2 (Galgani, Hanke and Maes, 2015 and references therein). Using observations onboard vessels to calculate the abundance, composition and distribution of floating debris, Hinojosa and Thiel (2009), identified that 80 percent of the debris was composed of plastics items (PS, plastic bags and plastic fragments). Items may also wash up onto shorelines of mid-ocean islands which tend to lie on the boundaries of accumulation zones (e.g. the Hawaiian Islands and the Equatorial Atlantic Archipelago of Saint Peter and Saint Paul, Ivar do Sul *et al*., 2013).

Water column transport of plastics appears to be a transitional zone, items are usually present if they are sinking to the benthos, or returning to the surface. Turbidity can cause items to sink forcing plastics below the water surface into the water column. Sinking could be facilitated by encrusting biota or microorganisms which affect the buoyancy of a plastic item, and rising through the water column could be through the release of plastic items from the benthos because of disturbance events.

Seabed deposition of plastic is very dependent on location and densities range from 0 items to more than 7 700 items per km^2 (Galgani, Hanke and Maes, 2015 and references therein). Items usually sink to the sea bed if they have a density greater than that of the water or through biofouling.

2.7 CONTRIBUTION OF FISHERIES AND AQUACULTURE TO MARINE LITTER

As already discussed, plastics are a significant component of fisheries and aquaculture. Unfortunately, a proportion of the materials used in fisheries and aquaculture may become marine debris. As with all user products and plastic items, when they reach their end of life they must be removed from the environment and either reclaimed, recycled or disposed in the correct waste streams. Through continued use plastic materials are exposed to direct UV light, wave action, abrasion and temperature changes. These factors can contribute to embrittlement and fragmentation. When they break, the items will no longer be fit for purpose. Additionally, plastics were adopted because of their advantages over traditional natural fibres which would readily degrade in the environment; however, the currently used synthetic polymers do not readily degrade and will take a long time to breakdown. Therefore, losses from fisheries and aquaculture are regularly reported in surveys of marine debris on beaches (Browne *et al*., 2015a; Nelms *et al*., 2017; Slip and Burton, 1991), floating on surface waters (Cózar *et al*., 2014; Thiel *et al*., 2003) and located on the seafloor (Iñiguez, Conesa and Fullana, 2016).

There are no global estimates of the amount of plastic waste generated by the fisheries and aquaculture sector. The first national estimate of fisheries and aquaculture debris input to marine environments has been estimated for the Republic of Korea (Jang *et al*., 2014b), where annual input from lost fishing gears was estimated at 44 081 tonnes, 2 374 tonnes of garbage thrown overboard from fishing vessels, and 4 382 tonnes of EPS floats lost from aquaculture facilities. Debris collected from North Sea coastlines such as Belgium, Denmark, France, Germany, the Netherlands, Norway, and the United Kingdom, has been largely attributed to shipping and fishing activity (Galgani *et al*., 2000; Vauck and Schrey, 1987; Unger and Harrison, 2016; Williams, Tudor and Randerson, 2003). Similarly, fisheries litter are a major component of oceanic litter in the Australian Bight (Edyvane *et al*., 2004), Korean waters (Hong *et al*., 2014; Jang *et al*., 2014a) and in the Celtic Sea (Moriarty *et al*., 2016). Some items of marine debris can be directly sourced to trawling and commercial netting operations, and in some cases labelled pots and bait boxes can identify specific fisheries and home ports.

2.7.1 Abandoned, lost or otherwise discarded fishing gears

In the fisheries and aquaculture sector it is considered that abandoned, lost or otherwise discarded fishing gears (ALDFG) are the main source of plastic waste into the marine environment (FAO, 2016b) but there is substantial variability in its distribution and abundance. Depending on the fishing system used, some methods regularly catch species close to the seafloor which can present problems for fishing gears due to irregularities such as rocks, cracks and crevices which might ensnare fishing gear and lead to breakages and loss. Trawls, dredges and pots will most commonly be affected by irregularities on the seafloor. Gears such as gillnets, trawls, handlines and longlines are a significant component of ALDFG. Loss of fishing gears can be a result of: enforcement on fishers to abandon gears (e.g. illegal fishing or illegal gears), operational pressure (e.g. use of too much gear in restricted time periods) and environmental conditions (e.g. weather, seabed irregularities), lack of/inaccessible/expensive onshore gear and waste disposal facilities (Macfadyen, Huntington and Cappell, 2009; Gilman *et al.*, 2016).

FIGURE 2.5

Crabs caught in an abandoned, lost or otherwise discarded fishing gear (ALDFG), in this case a gill net

Source: FAO.

ALDFG may continue to fish or become entangled in other fishing equipment. Most fishing debris will float because the density of plastics (such as HDPE and PS) are less than that of seawater, whereas, entangled and ensnared gears are more likely to continue fishing on the seafloor until they are removed. Derelict fishing gears dominate the seabed, for example, Pham *et al.* (2014) estimated that overall debris on the seabed of the Mediterranean Sea and Northeast Atlantic was composed mainly of plastics (41 percent) and ALDFG (34 percent), but in some locations, such as north of the Faroe Islands, the Norwegian continental shelf, and several seamounts and ocean ridges, ALDFG represented more than 75 percent of marine debris. A comprehensive analysis of floating macro-debris (> 200 mm diameter) revealed that 20 percent by number and 70 percent by weight was fishing–related, principally floats and buoys (Eriksen *et al.*, 2014). ALDFG is a major issue for fisheries and marine conservation, its presence in the marine environment can have a significant impact on commercial fishing and shellfish industry. ALDFG can lead to ghost fishing, stock depletion, capture of non-target species, conservation concerns, hazards to other vessels and it is costly to remove (Arthur *et al.*, 2014; Bilkovic *et al.*, 2016; Derraik, 2002; Laist, 1987; Wilcox *et al.*, 2015). One example is the derelict crab pots in Chesapeake Bay, Virginia, United States of America. In four consecutive winters (2008-2012) marine debris was removed during a collection programme and blue crab pots were the most abundant form of derelict gear recovered. Bycatch and animal mortality because of derelict pots was estimated to represent an annual potential economic loss of US$ 300 000 (Bilkovic *et al*., 2014).

2.7.2 Plastic debris from aquaculture

As with ALDFG, plastics may be released from aquaculture and mariculture facilities and become marine debris. Aquaculture structures are lost due to wear and tear of anchor ropes, storms, and accidents or conflicts with other maritime users. Severe weather conditions can cause widespread damage to aquaculture structures, at times generating large quantities of marine debris (Lee *et al*., 2015). The shellfish culture industry appears to be a major contributor to shoreline debris including EPS floats, plastic net sheets, bags, ropes and baskets (e.g. Hinojosa & Thiel, 2009; Jang *et al*., 2014a,b; Lee *et al*., 2013, 2015; Liu, Kao and Chen, 2015). Other concerns are the loss of antipredator nets from the shellfish industry (Bendell, 2015) and the continuous removal of biofouling organisms from aquaculture facilities, as this cleaning could release net and rope fibres in the form of microplastics to the environment (Floerl, Sunde and Bloecher, 2016).

Two examples of plastics debris from aquaculture include derelict gear from the pearl oyster aquaculture in French Polynesia (Andréfouët, Thomas and Lo, 2014) and the pollution of Puget Sound by shellfish aquaculture (Bendell, 2015). Lost or discarded materials from these two aquaculture facilities have been reported on the seafloor and beaches near the operations. PVC tubes, net caps, plastic bands, zip ties, oyster bags and nets on shellfish beds have been recorded. The extent and ecological impact is unknown and ongoing concern has highlighted that impacts could include, navigation hazards, ecological disturbances, hazards for animals and hazards for boat traffic and fishers. Finally, the breakdown of these items can lead to the formation of microplastics which could have further impacts on the environment.

3. Microplastics

3.1 DEFINITIONS

There is an ongoing debate about the appropriate definition of microplastics. So far, the most widely used definition is that microplastics are particles less than 5 mm in their longest dimensions. This definition has been adopted in practical terms as it is considered the size under which ingestion by many species of aquatic biota occurs (GESAMP, 2015). This loose definition has been accepted by the National Oceanographic and Atmospheric Administration (NOAA) of the United States of America and the Marine Strategy Framework Directive (MSFD) of the European Union for monitoring and the implementation of directives. Thus, this document will follow the definition that microplastics are particles consisting of a heterogeneous mixture of differently shaped materials referred to as fragments, fibres/filaments, beads/spheres, films/sheets and pellets in the range from 0.1 µm to 5 000 µm in their longest dimensions (Lusher *et al*., 2017; EFSA, 2016), while nanoplastics are defined as plastic particles ranging from 0.001 µm to 0.1 µm (Klaine *et al*., 2012).

Microplastics are largely classified by their morphological characteristics: size, shape and colour. Size is an important factor when studying microplastics as it dictates the range of organisms it may affect. The high surface area to volume ratio of small particles gives them a high potential for leaching and uptaking of chemicals (Velzeboer, Kwadijk and Koelmans, 2014). When reporting microplastic shape, researchers tend to use five main categories, although the nomenclature used varies between research groups (Table 3.1). Colours are often reported across a wide spectrum; colour differentiation is subjective and visual identification of microplastics cannot be based on colour alone. Caution should be given to categorizing microplastics suffering embrittlement, fragmentation or bleaching, or encrusted with biota, as this may skew results.

TABLE 3.1
Categories used when classifying microplastics by shape (adapted from Lusher *et al.*, 2017)

Shape classification	Other terms used
Fragments	Irregular shaped particles, crystals, fluff, powder, granules, shavings, flakes, films
Fibres	Filaments, microfibres, strands, threads
Beads	Grains, spherical microbeads, microspheres
Foams	Polystyrene, Expanded Polystyrene
Pellets	Resin Pellets, nurdles, pre-production pellets, nibs

Once released into the ocean, the environmental fate of microplastics primarily depends on the polymer density (Table 3.2) which influences buoyancy, position in the water column and the consequent possible interaction with biota (Wright *et al*., 2013). Polymers denser than seawater (> 1.027 g/cm^3, e.g. polyvinyl chloride (PVC)) will sink, while those with lower density (e.g. polyethylene (PE) and polypropylene (PP)) will tend to float in the water column. Processes like biofouling and the colonization of organisms onto the plastic surface increase the weight of particles, thus accelerating their sinking to bottom sediments (Lobelle and Cunliffe, 2011); also degradation, fragmentation and the leaching of additives can change the density of objects and their distribution within the water column.

TABLE 3.2

Specific gravity of plastics and different water types, and common applications of plastics found in the marine environment

Plastic type	Common applications	Specific gravity[*]
Polyethylene	Plastic bags, storage containers	0.91–0.95
Polypropylene	Rope, bottle caps, fishing gears, strapping	0.90–0.92
Polystyrene (expanded)	Cool boxes, floats, cups	1.01–1.05
Polystyrene	Utensils, containers	1.04–1.09
Polyvinyl chloride	Film, pipe, containers	1.16–1.30
Polyamide or Nylon	Fishing nets, rope	1.13–1.15
Poly(ethylene terephthalate)	Bottles, strapping, textiles	1.34–1.39
Polyester resin + glass fibre	Textiles, boats	>1.35
Cellulose Acetate	Cigarette filters	1.22–1.24
Pure water		1.000
Sea water		1.027
Brackish water		1.005-1.012

Source: GESAMP, 2015

[*] Specific gravity is the ratio of the density of a material to that of pure water at 4°C

3.1.1 Primary and secondary microplastics

Primary microplastics are plastic particles which are manufactured in a size range below 5 mm (GESAMP, 2015). They include: (1) pre-production resin pellets, often 3-5 mm in diameter, which are used as raw material in plastic manufacture; (2) microbeads incorporated in cosmetic products; and (3) beads used for abrasive blasting of surfaces. The route of entry for primary microplastics into the environment will depend on their application: particles from cosmetic products will usually enter through wastewater; microplastics from abrasive blasting will enter through the atmosphere and wastewater, while primary microplastics used for raw materials may enter the environment through accidental loss during transportation and transshipment, or through runoff from processing plants. When too small for retention by wastewater treatment plants, primary microplastics may be passed directly into the oceans or pass through freshwater watercourses to subsequently enter the marine environment.

Secondary microplastics are the by-product of fragmentation and weathering of larger plastics in the environment (GESAMP, 2015). The generation of secondary microplastics may occur during use of plastic products (for example, textiles, paints and tyres) or once plastics have been disposed of in the environment. There are multiple pathways for the entry of secondary microplastics to the environment, which include (1) particles from textiles may enter through wastewater following washing or through air when drying (Browne *et al.*, 2011; Napper and Thompson, 2016); (2) weathering of plastics used in agricultural applications may enter the environment through surface runoff from soil; (3) abrasion of tyres during use generates microplastics that enter the environment through air and surface runoff; (4) fragmentation and weathering of items in landfills by UV light which may introduce microplastics into the atmosphere, rivers and the ocean by wind and surface runoff and (5), weathering of plastic litter in coastal areas and beaches which may remain in coastal sediments or be transported further offshore. The main environmental factors related to secondary microplastic generation are UV light exposure, temperature and abrasion. In aquatic environments areas with reduced UV exposure and low temperature (such as the deep sea) will slow down the process of secondary microplastic generation (Andrady, 2015).

3.1.2 Primary and secondary nanoplastics

Based on the internationally recognized definition of nanomaterials, nanoplastics can be defined as a material with at least one external dimension in the nanoscale, approximately in the region from 1 nanometre (nm) to 100 nm (EFSA, 2016). As with microplastics, they can be either primary or secondary. The former are engineered within this size range for a variety of industrial applications and, as with smaller primary microplastics, will not be retained by wastewater treatment plants and enter the aquatic environment (GESAMP, 2015). There is little doubt that secondary nanoplastics will be produced during fragmentation or weathering of microplastic debris (Andrady, 2011; Koelmans, Besseling and Shim, 2015; Mattsson, Hansson and Cedervall, 2015). Laboratory experiments showed degradation of polystyrene (PS) disposable coffee cup lids with formation of nanoplastics over time (Lambert and Wagner, 2016). Possibly, microbial degradation could also play a role, because several hydrocarbon degrading microorganisms have been identified to thrive on plastic debris in the oceans (Zettler, Mincer and Amaral-Zettler, 2013). Current methods of microplastic detection do not extend to particles in the nano size range. Thus, little is known about the occurrence and distribution of nanoplastics in the environment (Mattsson, Hansson and Cedervall, 2015). Table 3.3 summarizes information on particle sizes, their sources and examples.

TABLE 3.3

Summary of size definitions of marine plastic litter and common sources

Size category of marine litter	Longest dimension				
	Nano < 0.1 µm	Micro < 5 mm	Meso < 2.5 cm	Macro < 1 m	Mega >1 m
Source	Primary nanoplastics Secondary fragmentation of microplastics	Primary microplastics Secondary fragmentation of larger plastic items	Direct and indirect: including fragmentation	Direct: lost items from maritime activities	Direct: abandoned gears Indirect: land based waste
Examples	PRIMARY: Industrial applications including pharmaceuticals and the medical device industry SECONDARY: microplastic fragments	Primary: Resin pellets Secondary: fragments and fibres	Bottle caps, plastic fragments	Plastic bags, food and other packaging, fishing floats, buoys, balloons	Abandoned fishing nets and traps, ropes, boat hulls, plastic films from agriculture

3.2 FISHERIES AND AQUACULTURE AS A SOURCE OF MICROPLASTICS

As discussed in section 2.7, both recreational and commercial fisheries, and aquaculture facilities directly introduce plastics into the marine environment. The cost implications is such that equipment loss is kept to a minimum through maintenance and gear recovery where possible, and most producers are resolved to appropriately dispose of materials that reach the end of their usable life span (Jensen and Zajicek, 2008). In many cases, best practices and working standards have been produced by governmental bodies, industrial organizations and researchers (e.g. National Research Council, 1995; BCSGA, 2013); unfortunately, environmental weathering, biodegradation, and the wear and tear of in-use plastics still result in the loss of large plastics and the formation of microplastics. It is pertinent, however, to reiterate that it is difficult to estimate the percentage of marine litter that originates from maritime sources, and only a proportion of this will result from the fisheries and aquaculture industry (Jambeck *et al.*, 2015).

3.2.1 Fishing gear contributing to microplastics in the field

The commercial gears which have the potential for greatest contribution to microplastic loads are the nettings used in benthic dredges and trawls and in particular the ground ropes. This rope is dragged in contact with the sea bed for many miles, subject to abrasion from benthic sediments, or snagging and total loss. The use of sacrificial ropes protects the integrity of the net; however, these ropes are often of plastic construction and must be monitored and replaced as they wear away over time. In addition to the in-use degradation of fishing gears, and despite careful maintenance, analysis of the abundance of ALDFG have shown that many kilometres of netting are lost to the marine environment each year (Macfadyen, Huntington and Cappell, 2009). Thus, plastics from the fishing industry are apparent in numerous studies of environmental plastic contamination (e.g. Edyvane *et al*., 2004; Macfadyen, Huntington and Cappell, 2009; Choy and Drazen, 2013; Peters and Braton, 2016). For example, ingested synthetic fibres are similar to those commonly used in fishing gears (Lusher *et al*., 2013), and the forms of microplastic debris (fibres, fragmented debris) have been linked to local fisheries sources (Dantas, Barletta and Costa, 2012; 2015; De Witte *et al*., 2014; Mathalon and Hill, 2014). Trawling activities in the enclosed Clyde Sea have been linked to elevated levels of microplastic fibres in Norway lobster (*Nephrops norvegicus*) populations (Murray and Cowie, 2011). Expanded polystyrene (EPS) particles were found inside mussels inhabiting on EPS buoys and their debris (Jang *et al*., 2016). Finally, in the Goiana estuary, Brazil, 12 percent of Mojarras (family Gerreidae), 23 percent of catfish (family Ariidae) and 64 percent of Acoupa weakfish (Cynoscion acoupa) were seen to contain microplastics which were of a similar description to local artisanal and commercial fishing practices (Possatto *et al*., 2011; Dantas, Barletta and Costa, 2012; Ramos, Barletta and Costa, 2012; Ferreira *et al.,* 2016).

It is not the case that fishing activities are the sole source of fisheries and aquaculture related microplastics. Examination of microplastic loads in Jurujuba Cove, Brazil showed high levels of synthetic fibres believed to originate from local mussel farming (Castro *et al*., 2016), and Mathalon and Hill (2014) revealed elevated levels of microplastic fibres in farmed mussels when compared to wild gathered counterparts, indicating increased uptake from aquaculture sources. Observations of microplastic releases have also been recorded from aquaculture structures. For instance, boring isopods (*Sphaeroma* spp.) burrow in PS floats used in aquaculture facilities and docks. As these small crustaceans burrow into the plastic, they may release thousands of microplastic particles to the marine environment (Davidson, 2012).

3.2.2 Possible indirect pathways

In addition to acting as a local source of marine debris, commercial seafood production may aggregate and disperse microplastics in several ways. One possible route for plastic dispersal is discards. Worldwide discard totals were estimated at approximately 10 million tonnes per year between the years 2000 and 2010 (Pauly and Zeller, 2016), much of which was probably consumed by scavenging benthic species. In the Clyde Sea Area, where benthic trawling for Norway lobster is the primary fishing type, much of the Norway lobster population feeds to some extent on discards. Studies of microplastic contamination in the stomach content of Clyde Sea Norway lobster, revealed both a high abundance of conspecifics (animals of the same species) and other discarded species (Welden and Cowie, 2016a). In this way, animals which contain plastic may be directly fed to the remaining population, increasing the microplastic load per individual. A similar effect may occur in aquaculture if microplastic is accidentally aggregated in the production of feeds from already contaminated individuals. Large numbers of small pelagic fishes, at risk of direct uptake of microplastics mistaken for food items, are converted into fishmeal and fish oil each year (FAO, 2016b), a proportion of which is used in the aquaculture industry as feed. The homogenization and dehydration of these feeds results in a feedstock preloaded with microplastic contaminants, but no data are available about this possible indirect pathway. In addition to the use of contaminated feeds in aquaculture, fisheries may also influence the availability of environmental plastics. Regular trawling also results in a plume of contaminated sediments (Churchill, 1989), which makes previously entrained plastics available to filter feeding organisms, and alters the distribution of sediment microplastics in line with the tidal conditions and currents (Lattin *et al.*, 2004). In pole and line and other baited fisheries, contaminated bait may result in either introduction of microplastic to the target organisms upon capture or to the environment as bait is discarded or lost. Offal may be another source of microplastics as many fish species are gutted onboard during fishing activities and the remains are thrown overboard. This offal is scavenged upon by seabirds, fish and benthic organisms, and it is possible that any microplastics present within the remains may thus re-enter the food web.

3.3 DISTRIBUTION OF MICROPLASTICS IN THE AQUATIC ENVIRONMENT

Although first identified as a potential hazard as early as the 1970s (Carpenter and Smith, 1972), only reports from the early 2000s gained a sufficient following that research into microplastic contamination became a priority (Thompson *et al.*, 2004). Moreover, only in the last 10 years has the subject received increased attention from the research community, mainly as a subset of the more general problem of marine plastic litter, but increasingly as an environmental contaminant in their own right. In recent years, governments and non-govermental organizations (NGOs) started to fund research into standardized and harmonized protocols for the identification and understanding of microplastic impacts (Lusher *et al.*, 2017) and cleanup (Rochman, 2016; Sherman and van Sebille, 2016).

Microplastics represent an increasing proportion of marine debris and to date several modelling studies have attempted to identify numerous sources, distribution and accumulation areas (Clark *et al.*, 2016; Cózar *et al.*, 2014; Eriksen *et al.*, 2014; van Sebille *et al.*, 2015). Microplastics may enter aquatic environments from different sources and follow diverse pathways. Sources on land are related to many activities such as primary microplastic production, terrestrial transportation (i.e. microplastics generated from tyre wear), agricultural activities, wastewater plants, degradation of plastic litter on beaches, and may enter aquatic environments through the atmosphere, coastline or runoff. On the other hand, as has been discussed in more detail in section 2.7, maritime sources are related to shipping, offshore industrial activities, and fisheries and aquaculture (mainly from the degradation of lost gears) (GESAMP, 2016).

3.3.1 Freshwater and estuarine environments

Until recently, the distribution of microplastics in freshwater systems was unknown. There are relatively few studies addressing the presence of microplastics in freshwater environments including lakes, rivers and estuaries. Nevertheless, several recent publications have reviewed current knowledge on occurrence of microplastics in freshwater environments (Dris *et al.*, 2015a; Driedger *et al.*, 2015; Eerkes-Medrano, Thompson and Aldridge, 2015; Horton *et al.*, 2017b; Wagner *et al.*, 2014) and addressed the fate of micro and nanoplastics in freshwater systems (Besseling *et al.*, 2017). Table A2.2 in Annex 2 presents data on the abundances of microplastics in freshwater and estuarine environments. There is large spatial variability in the distribution of microplastics in freshwater environments, with results showing differences of several orders of magnitude within and between studies. In general, sediments show higher concentrations than water samples, and results are comparable to those obtained in coastal marine habitats. On the other hand, in lacustrine environments, microplastics tend to show higher abundance close to urban and industrial centres, but in some riverine systems this relation does not hold and may be a result of river flow dynamics and flooding (Klein, Worch and Knepper, 2015). Inadequate waste management and windblown microplastics could contribute to pollution of relatively isolated freshwater environments (Free *et al.*, 2014).

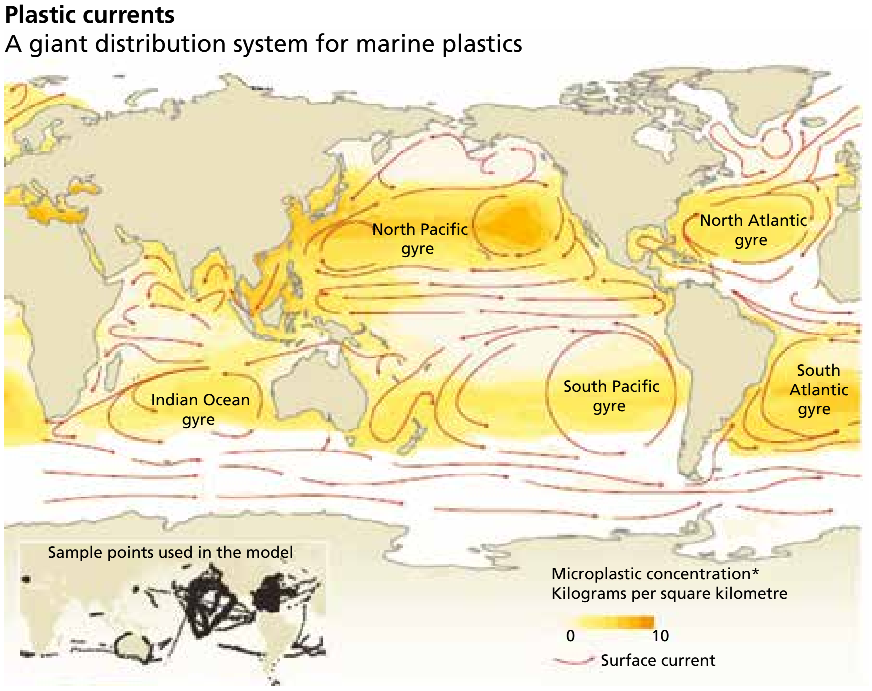
3.3.2 Marine environments

Microplastics have been documented in every habitat of the open ocean and enclosed seas, including beaches, surface waters, the water column and the deep seafloor (Lusher, 2015; Table A2.3 in Annex 2). In oceans, the small size and low density of microplastics contribute to their widespread transport across large distances (Figure 3.1) by ocean currents (Cole *et al.*, 2011; Eriksen *et al.*, 2013b). Oceanic transport can move buoyant microplastics to distant shorelines or entrained particles can accumulate in central ocean regions (e.g. Law *et al.*, 2010). Microplastic distribution is further influenced by wind that can cause the vertical movement of particles within the water column (Kukulka *et al.*, 2012). Wind, surface currents and geostrophic circulation drive the dispersal patterns of microplastics at large scales (Law *et al.*, 2010). Otherwise, at smaller scales, such as turbulent flows, from tides or waves, high-energy oceanographic events, like sea storms, as well as dredging processes to remove heavy depositions of sediment can all lead to the resuspension of benthic particles from sediments (Ballent *et al.*, 2012, 2013). As mentioned earlier, particles which have a greater density than seawater and those which are biofouled will readily sink to the seafloor. The most recent estimate of microplastic abundance in surface oceanic and neritic waters yields a concentration of 51.2 x 10^{12} particles and a mass of 260 000 tonnes (van Sebille *et al.*, 2015). Per this model, the highest concentrations and masses occur in the Mediterranean Sea and the North Pacific Ocean. A recent study on the Mediterranean Sea provides further evidence that microplastic abundances are relatively high compared to other water basins (Suaria *et al.*, 2016).

Most studies on occurrence and abundance of microplastics have been conducted on beaches and coastal surface waters of enclosed or semi-enclosed seas, such as the North Sea, Caribbean Sea, Mediterranean Sea and China Sea. Microplastics have been reported along coastlines of all continents, in remote locations such as the mid-Atlantic archipelago islands (Ivar do Sul *et al.*, 2013; Lima, Barletta and Costa, 2016), the Arctic (Lusher *et al.*, 2015b; Obbard *et al.*, 2014) and Antarctic (Waller *et al.*, 2017).

FIGURE 3.1

Marine global distribution of microplastics in surface oceanic and neritic waters from model by van Sebille *et al.* 2015. The lower left rectangle shows observed sample points



Source: UNEP and GRID-Arendal, 2016.

In general, microplastics are more abundant in beaches and infralittoral sediments and near highly populated areas. Also, the accumulation of microplastics in oceanic gyres and accumulation zones, especially in the North Pacific and the North Atlantic has received much attention (Eriksen *et al.*, 2013b; Law *et al.*, 2010; 2014). Microplastics have also been located in deep sea habitats (Fischer *et al.*, 2015; Van Cauwenberghe *et al.*, 2013a; Woodall *et al.*, 2014).

3.3.3 Relation of microplastics in aquatic environments with fisheries and aquaculture

Areas of coastal mariculture and fishing may be responsible for the presence of microplastics, whereas the sources of microplastics in offshore fishing grounds may be harder to interpret because of the influence of oceanic distribution. However, sources of secondary microplastics from fisheries and aquaculture can include the routine wear and tear of fishing gears and other equipment. For example, ground ropes on bottom trawls will fragment through use and mechanical abrasion against bottom substrata. In addition, it is likely that fisheries play an important role in the resuspension and spatial redistribution of microplastics in marine environments, particularly through bottom trawling and dredges. Plastic cages, floats and buoys that are exposed to air and UV light can become embrittled and fragment forming microplastics. Furthermore, the cleaning of facilities in aquaculture (i.e. rope used in shellfish culture), can generate microplastic fibres during the removal of biofouling organisms.

It can be extremely complicated to directly source microplastics to fisheries and aquaculture operations unless microplastics have the same visual and chemical characteristics as the gears used. Possibly the most obvious microplastics related to fisheries and mariculture is the breakdown of EPS floats; EPS fragments originating from aquaculture buoy debris are found as a dominant item (above 90 percent) of microplastics on beaches of the Republic of Korea (Heo *et al.*, 2013; Lee *et al.*, 2013). Davidson (2012) reported that boring isopods (*Sphaeroma* spp.) may form colonies in EPS floats, which potentially generated millions of PS microplastics per colony. In Lake Hovsgol, Mongolia, Free *et al.* (2014) associated some of the microplastics present in the water column with fragmentation of plastic fishing debris encountered in shoreline surveys. Fragmented ropes and fishing lines may be responsible for the microplastics described as “lines” which were found in the Yangtze River, China (Zhang *et al.*, 2015). Similarly, high densities of microplastics were reported in sediment in the port of Chioggia in the Venice Lagoon, which is the main fishing port on the Adriatic Sea (Vianello *et al.*, 2013).

3.4 CHEMICAL AND BIOLOGICAL INTERACTIONS OF MICROPLASTICS IN THE ENVIRONMENT

Microplastics (and macroplastics) contain a mixture of chemicals added during manufacture, but also sorb (either adsorb or absorb) contaminants from the surrounding environment (Rochman, 2015). These added chemicals include monomers, oligomers, plasticizers, and flame retardants, whereas contaminants sorbed from the environment include polychlorinated biphenyls (PCBs), polycyclic aromatic hydrocarbons (PAHs), chlorinated pesticides, collectively included within the group of persistent organic pollutants (POPs), trace metals, and bacteria. POPs are also classed as persistent bioaccumulative toxic substances (PBTs). The term PBTs will be used throughout the report because it reflects a major concern of these compounds: their ability to accumulate in organisms. PBTs also include, for example, flame retardants added to plastics such as polybrominated diphenyl ethers (PBDEs). This section provides a brief description of some of the major chemicals as well as information on microbes associated with microplastics.

Monomers and oligomers

Polyethylene (PE) is used in large volumes in the production of high-density polyethylene (HDPE) and low-density polyethylene (LDPE). The release of monomers and oligomers from PE are currently not recognized to represent a significant human health threat (Halden, 2010).

Polyethylene terephthalate (PET) is produced by esterification of terephthalate or dimethyl terephthalate, monomeric building blocks that are distinct in structure and toxicity from other, more problematic phthalates. Reports on the release of intermediary monomers, e.g., bis(2-hydroxyethyl)terephthalate, from PET are rare (Bach *et al*., 2013). Extraction of commercial PET products with saline yielded rather weak estrogenic activity displaying compounds in 76 percent of products tested (Yang *et al*., 2011). Similar to PE, toxicity concerns regarding PET mostly centre on secondary contaminants acquired from the environment.

Studies on polypropylene (PP), used in the manufacture of packaging, plastic containers and bottle tops, suggest limited risk to human health from this polymer. Again, similar to PE, migration of PP monomers is not a recognized concern and leaching of additives into surrounding media also appears to be limited (Dopico-Garcia, López-Vilariñó and González-Rodríguez, 2007). Eighty-one percent of 16 unstressed (i.e. not submitted to environmental stressors, such as UV radiation and high temperatures) commercial PP products tested displayed limited estrogenic activity from unidentified leachable chemicals (Yang et al., 2011).

Polystyrene (PS) articles such as packaging, dairy containers, and utensils are known to release styrene monomers and oligomers, which can bind to the oestrogen receptor and exhibit oestrogen-like activity (Klärner *et al*., 1998; Ohyama, Nagai and Tsuchiya, 2001; Park *et al*., 2006). Thirty-eight percent of 16 unstressed commercial PS products tested displayed limited estrogenic activity (Yang *et al*., 2011). However, the potency and importance of endocrine disruptors leachable from PS articles are subject to debate (Halden, 2010).

Polyvinyl chloride (PVC) is manufactured from vinyl chloride monomers. Although vinyl chloride is recognized as a genotoxic and mutagenic toxicant, polymerization of vinyl chloride in PVC removes these adverse properties. Toxicity concerns about chemicals given off by weathered PVC tend to focus not on its monomeric building block but instead on phthalates, specifically di-(2-ethylhexyl) phthalate (DEHP; Halden, 2010).

Plasticizers

In the case of plasticizers, their ease of migration from polymers has classed them as hazardous because they have environmental impacts and are the subject of health concerns (Murphy, 2001; Oehlmann *et al*., 2009; Teuten *et al*., 2009). Bisphenol A (BPA) is best known as the monomeric building block of polycarbonate plastics but it also finds uses as an additive to other plastic polymers (PE, PP, PVC). Potential adverse effects of BPA have already been addressed in section 2.3.1. BPA contents of microplastics in the marine environment are given in section 5.6.1.

Flame retardants

Flame retardants may be mixed in (e.g. polyurethane foams) or reacted into the polymer (e.g. epoxy resins). Brominated flame retardants (BFRs) are of particular concern as they are mixed into plastic resins and are more available to leach out of the plastic. BFRs include polybrominated diphenyl ethers (PBDEs), tetrabromobisphenol A (TBBPA), hexabromocyclododecane (HBCD), and brominated polystyrene. Exposure pathways to BFRs, include household dust, food and breast milk as major sources.

In the marine environment BFRs may leach directly from plastics or be transported from land-based sources. PBDEs are highly persistent and accumulate in aquatic environments (Hermabessiere *et al*., 2017), but to a lesser degree than PCBs (Engler, 2012). PBDEs measured in microplastics in the marine environment are presented in section 5.6.1.

Other persistent bioaccumulative and toxic substances (PBTs)

Of the environmental chemicals that are present in seawater, common compounds include dichlorodiphenyltrichloroethane (DDT), PAHs and PCBs. PBTs may enter aquatic environments through several pathways: dispersion, industrial releases, or release from objects like plastics. PBTs are hydrophobic chemicals which have a long life in the environment as they are resistant to environmental degradation and may persist for several years. Their hydrophobicity drives the partitioning to microplastics and organic particles. As plastics also have hydrophobic properties, microplastics act as a sink for these pollutants present in aquatic environments (Gouin *et al*., 2011; Rochman, 2015). Sorption processes can be influenced by weathering and biofouling. Microplastics collected from around the world contained PBTs (Ogata *et al*., 2009) with amount of sorbed contaminants varying by geographic origin and by polymer type (Endo *et al*., 2005; Mato *et al*., 2001). Published data are available only for a limited number of PBTs in microplastics. Section 5.6.3 presents measured contents of some PBTs in microplastics in the marine environment.

Trace metals

Other chemicals associated with plastics are trace or heavy metals. Some heavy metals are incorporated into plastics during manufacture to act as stabilizers, anti-oxidants and dyes (Murphy, 2001). Metals are also incorporated into antifouling paints which are a source of heavy metal contamination in harbours and marinas through paint deterioration (Almeida, Diamantino and de Sousa, 2007). Metals can adsorb to microplastics under marine (Ashton, Holmes and Turner, 2010; Holmes, Turner and Thompson, 2012), estuarine (Holmes, Turner and Thompson, 2014) and freshwater (Turner and Holmes 2015) conditions. For example, virgin PE pellets absorbed heavy metals (aluminium (Al), iron (Fe), manganese (Mn), copper (Cu), lead (Pb), zinc (Zn), silver (Ag)) from seawater during experimental exposure (Ashton, Holmes and Turner, 2010). Although fragmented plastics accumulated metals better than virgin plastics (Brennecke *et al*. 2016), the degree of accumulation was moderate as compared to PBTs. Theoretically, desorption of metals is possible, but data on this are lacking. Finally, the presence of metals in microplastics has been seldomly studied.

Bacteria and viruses

The opportunities presented for settlement of biota on microplastics provide a unique surface for colonization as is evidenced by biofilms which contain a diverse community of microbes (Carson *et al*., 2013; Caruso, 2015; Osborn and Stojkovic, 2014). This phenomenon, known as rafting, enables the transport of organisms attached to natural or man-made materials; the durability of plastic results in these rafts being transported for much greater distances than their natural counterparts. However, colonization of plastic materials is restricted to a smaller number of organisms. Unlike natural rafting materials, plastic may continue to transport biota for decades, and the extent of plastic pollution in worldwide oceans provides a continuously increasing colonization surface and promotes the odds of microbe-polymer encounters, binding and colonization. Fortunately, plastic debris has been seen to transport a lower number of species due to the difficulties posed by initial colonization, and due to its size, the number of species that may be transported by microplastic debris is reduced. However microplastic debris may facilitate the transport of microorganisms that bind to its surface causing migration over long distances. Microplastics have previously been seen to transport bacteria including *Vibrio* (Amaral-Zettler *et al*., 2015; Harrison *et al*., 2014; Zettler, Mincer and Amaral-Zettler, 2013) and more specifically pathogenic strains (*V. coralliilyticus, V. harveyi, V. splendidus, V. parahaemolyticus, V. alginolyticus and V. fluvialis)* (Kirstein *et al*., 2016), as well as harmful micro-organisms such as toxic dinoflagellates (*Ostreopsis* spp., *Coolia* spp., *Alexandrium* spp.) (Masó *et al*., 2003). To date, no data are available on the potential presence of viruses on plastic debris although it is well known that viruses, like other microbes, attach to particles in the environment through complex reversible and irreversible interactions, and so their fate and transport is associated with that of the particles (Tufenkji, 2007, Dang *et al*., 2008).

3.4.1 Sorption and release of PBTs and additives by microplastics

The hydrophobicity of PBTs drives their partitioning to hydrophobic microplastics and organic particles, which act as a sink for these pollutants present in aquatic environments (Gouin *et al*., 2011; Rochman, 2015). Sorption is enhanced by the large surface area to volume ratio of microplastics, which increases by ongoing fragmentation of the particles. For a spherical particle, this ratio is inversely proportional to its diameter. Under artificial laboratory conditions, microplastics have been shown to readily accumulate waterborne PBTs (Bakir, Rowland and Thompson, 2104; Bakir *et al.*, 2016; Napper *et al*. 2015; Rochman 2015; Wardrop *et al*., 2016). Indeed, high concentrations of PBTs have been measured in microplastics in the marine environment (see section 5.6). Furthermore, as mentioned above, plastics also moderately accumulate metals although their presence on microplastics has been less studied (Ashton, Holmes and Turner, 2010; Holmes, Turner and Thompson, 2012).

Like sorption of chemicals, their release or desorption is potentially enhanced by a large surface area to volume ratio, and both are also dependent on the concentration gradients between the particles and the surrounding water or organisms, temperature and pH (Teuten *et al*., 2009; GESAMP, 2016, Engler, 2012; Rochman, 2015; Jang *et al*., 2016). The ability of a compound to migrate out of microplastics is further dependent on the polymer pore size relative to that of the toxic molecule.

Understanding sorption, accumulation, desorption, release and transport of these chemicals associated with microplastics is a challenging process, because also a variety of organic particles, like plankton, organisms and microorganisms, sediments, etc., in the marine environment will interact with the chemicals. The partitioning of chemicals between microplastics and organic particles is a dynamic process dependent on many variables which are constantly changing (Hartmann *et al*., 2017).

3.4.2 Transfer of PBTs and additives from microplastics to marine organisms

The ability of microplastics to accumulate PBTs has generated considerable concern that microplastics will transfer these hazardous PBTs to marine animals, and has been central to the perceived hazard and risk of microplastics in the marine environment (Gouin *et al*., 2011; Koelmans, Besseling and Foekema, 2014; Koelmans, Besseling and Shim, 2015; Koelmans *et al*., 2013, 2016; Bakir *et al*., 2016; Paul-Pont *et al*., 2016). However, calculations based on published studies showed the fraction of total PBTs sorbed by microplastics to be small compared to that sorbed by other media in the ocean. Taking into account measured PBT transfer from microplastics to organisms in the laboratory, desorption rates for plastics in artificial gut fluids, theoretical desorption rates and simulations in bioaccumulation models containing plastics, it was concluded that overall the amount of PBTs bioaccumulated from natural prey overwhelms the amount from ingested microplastics. This implies that ingestion of contaminated microplastic is not likely to increase the exposure to PBTs in marine organisms (Koelmans *et al*., 2016). However, it has been argued that in some local conditions with high levels of contamination by microplastics (i.e. hotspots) these could represent a significant phase for interaction with PBTs and their subsequent transfer to marine organisms (Hartmann *et al*., 2017).

Although it is often assumed that ingestion of microplastics by aquatic species leads to increased exposure to plastic additives, experimental evidence at environmentally relevant concentrations is lacking (Hermabessiere *et al*., 2017; Koelmans *et al*., 2016). The potential of leaching of nonylphenol (NP) and bisphenol A (BPA) in the intestinal tracts of the lugworm (*Arenicola marina*) and Atlantic cod (*Gadus morhua*) was studied using a biodynamic model (Koelmans, Besseling and Foekema, 2014). A conservative analysis showed that microplastic ingestion had a negligible contribution to the total exposure to NP and BPA.

4. Interactions of microplastics with aquatic organisms

Microplastics have a global distribution and have been found in a range of aquatic and terrestrial environments. Organisms across many trophic levels interact with microplastics and there are a number of routes by which exposure and interaction may occur. Microplastics present in the environment (water or sediment) elicit direct exposure of organisms, whereas microplastics which have been previously ingested by prey items may represent an indirect source of contamination to predators through trophic transfer. The scale and ubiquity of this contamination raises concerns regarding interactions with biota, motivating a number of exposure and toxicological studies under laboratory conditions. In fact, controlled laboratory exposures facilitate monitoring of the uptake, movement and distribution of synthetic particles in whole organisms and excised tissues (e.g. gills, intestinal tract and liver). Laboratory studies have confirmed that a diverse array of marine organisms, across trophic levels, can absorb or consume microplastics. This includes protists (Christaki *et al.*, 1998), copepods (e.g. Cole *et al.*, 2013), annelids (e.g. Besseling *et al.*, 2013, Wright *et al.*, 2013), echinoderms (Della Torre *et al.*, 2014; Kaposi *et al.*, 2014; Nobre *et al.*, 2015), cnidaria (Hall *et al.*, 2015), amphipods (Thompson *et al.*, 2004, Ugolini *et al.*, 2013), decapods (e.g. Watts *et al.*, 2014), isopods (Hämer *et al.*, 2014), molluscs (e.g. Avio *et al.*, 2015a), fish (e.g. Pedà *et al.*, 2016), and birds (Tanaka *et al.*, 2013). Interactions between freshwater invertebrates and microplastics have also been reported, most commonly focusing on model species such as *Daphnia magna* (e.g. Besseling *et al.*, 2014). In comparison, the consumption of microplastics by free living terrestrial organisms is poorly documented; however, laboratory studies indicate earthworms (*Lumbricus terrestris* and *Eisenia andrei*) can consume plastic particles present in soil (Huerta-Lwanga *et al.*, 2016; Rillig, Ziersch and Hempel, 2017; Rodriguez-Seijo *et al.*, 2017).

4.1 MECHANISMS OF INTERACTIONS (UPTAKE, INGESTION) AND IMPACTS (PHYSICAL AND CHEMICAL) OF MICROPLASTICS DETERMINED IN EXPERIMENTAL SETTINGS

A number of studies have explored the mechanistics of biota-microplastic interactions in laboratory settings. Micro- and nanoplastics can adhere to external appendages, including setae, swimming legs and antennules of copepods (Cole *et al.*, 2013; 2015), and the gills of the green crab (*Carcinus maenas*) (Watts *et al.*, 2014) and blue mussels (*Mytilus edulis*) (Paul-Pont *et al.*, 2016).

Possibly the most studied interaction is uptake of microplastics through ingestion (as reviewed in Lusher, 2015; Lusher *et al.*, 2017). Ingestion is the most likely interaction between organisms and microplastics, as their small size can make them indistinguishable from natural prey items, or they may be accidentally ingested while filter feeding (Lusher, 2015). Direct consumption of microplastic is prevalent in suspension feeders including oysters (Sussarellu *et al.*, 2016) and mussels (e.g. Avio *et al.*, 2015a) and deposit feeders, such as sea cucumbers (Graham and Thompson, 2009), annelids (Besseling *et al.*, 2013; Browne *et al.*, 2015b; Wright *et al.*, 2013), crabs (Watts *et al.*, 2014) and Norway lobster (*Nephrops norvegicus*) (e.g. Welden and Cowie, 2016a, b). Green crabs (*Carcinus maenas*) can also uptake microplastics across the gills during ventilation (Watts *et al.*, 2014). These uptake behaviours stem from the organisms' inability to differentiate between microplastics and food. Recent results also indicate that microplastics in marine environments acquire a dimethyl sulphide signature, which acts as a keystone odorant in pelagic foodwebs (Savoca *et al*., 2016). This would imply that some aquatic organisms may also actively search out and ingest microplastic particles.

There is growing concern for the possible trophic transfer of microplastics in aquatic, benthic and pelagic foodwebs. Predatory organisms may indirectly accumulate microplastics during the ingestion of microplastic contaminated prey, which may lead to bioaccumulation at upper trophic levels. Similarly, predators and detritivores may ingest microplastics while scavenging detrital matter containing microplastics. Laboratory experiments have established that green crabs fed with blue mussels containing microplastics accumulated particles in their digestive tract (Watts *et al*., 2014; Farrel and Nelson, 2013); Also, trophic transfer from green algae (*Scenedesmus* spp.) to the planktonic water flea (*Daphnia magna*) and then to several species of fish: Crucian carp (*Carassius carassius)*, Bleak (*Alburnus alburnus)*, Rudd (*Scardinius erythrophthalmus*), Tench (*Tinca tinca)*, Northern pike (*Esox lucius)* and Atlantic salmon (*Salmo salar*) has been observed (Besseling *et al*., 2014; Cedervall *et al*., 2012), as well as transfer of fluorescent polystyrene (PS) microspheres (10 µm) from zooplankton to the mysid shrimp (*Mysis* spp.) (Setälä, Fleming-Lehtinen and Lehtiniemi, 2014). It should be noted however that while laboratory studies may succeed in determining the pathways of microplastics within an organism, the exposure concentrations used to achieve this goal exceed those expected in the field (Phuong *et al*., 2016), such that the results should be treated with care (Box 4.1).

Many laboratory studies have observed interactions of biota with microplastics (as introduced above): in the case of non-commercially targeted species refer to Table A2.4 (Annex 2). Figure 4.1 presents a broader overview of the interactions and scale of the effects of microplastics exposure. Microplastic exposure has been associated with a suite of negative health effects, including increased immune response (von Moos, Burkhardt-Holm and Köhler, 2012), decreased food consumption, weight loss and energy depletion (Besseling *et al*., 2013; Wright *et al*., 2013), decreased growth rate (Huerta-Lwanga *et al*., 2016), decreased fecundity and negative impacts on subsequent generations (Sussarellu *et al*., 2016). Whether these effects will also occur at realistic concentrations of microplastics is largely unknown.

BOX 4.1
General considerations about laboratory exposure experiments

Laboratory exposure experiments are a key aspect of the study of microplastics and they have contributed to a greater knowledge and understanding of their behaviour in the environment and their interaction with biota and other chemical pollutants.

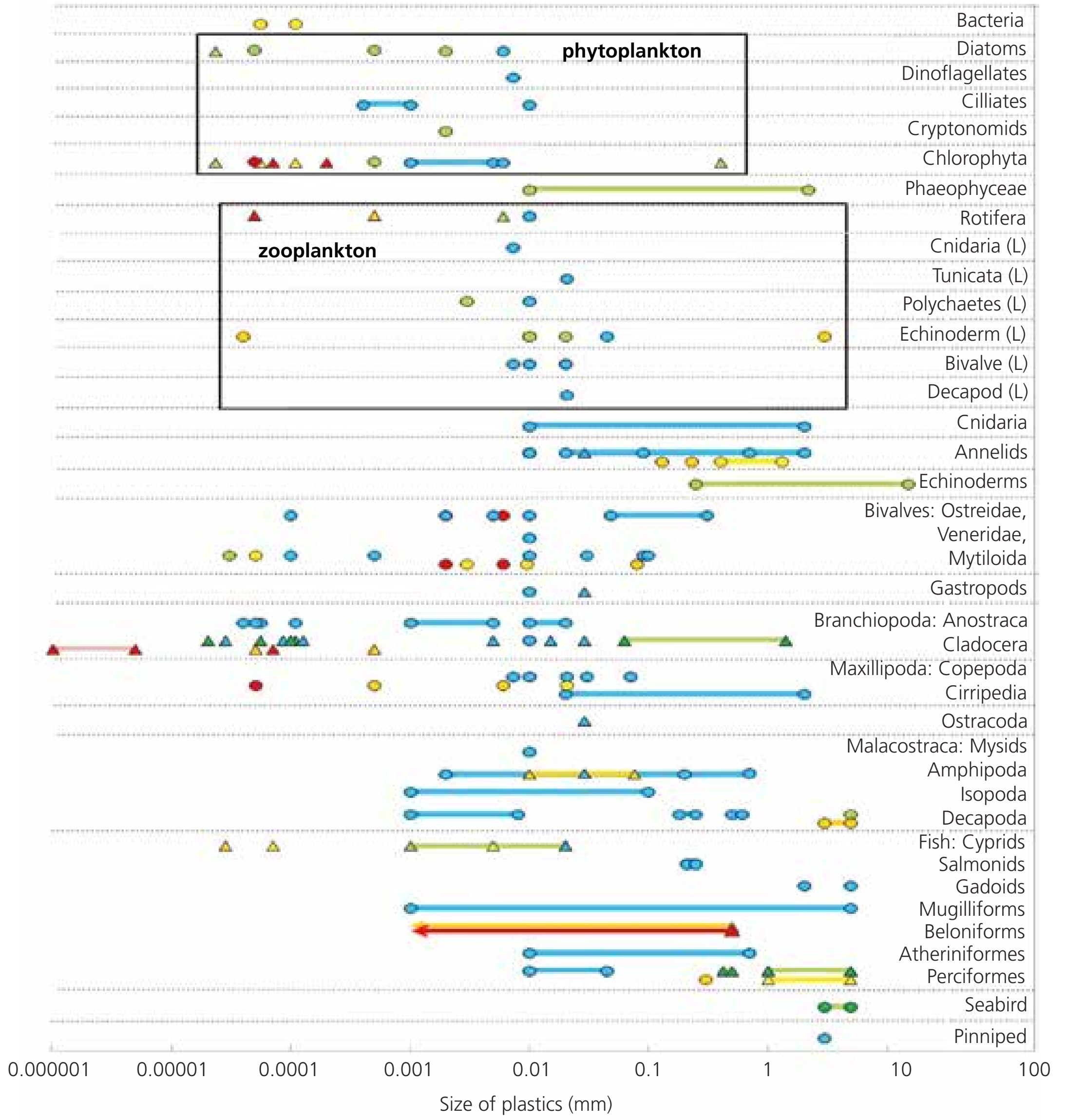
However, often exposure (particle number/mass/concentration) exceeds that observed in the field and results should be treated with care. For example:

- Exposure duration is often acute involving high levels of microplastics over a short period of time.
- Sample sizes are often small and comprised of individuals of one size or life history stage, and cannot be reliably expanded to wild populations.
- Microplastics used for exposure are often spherical in form, while the most commonly found microplastics in the environment are fibres and fragments.

Ongoing research into laboratory exposures have begun to use more environmentally-realistic scenarios such that cumulative negative effects are monitored under chronic exposure scenarios to better replicate environmental conditions.

FIGURE 4.1

Summary of laboratory experiments (published up to the end of December 2016), in which organisms were exposed to micro- and nanoplastics



Details of the studies can be found in Annex 2 Table A2.4. The x-axis shows the size of the plastic used in the exposure (mm) on a log-scale. Freshwater studies are depicted with triangles and marine studies are depicted with circles. The severity of the effect is rated from blue to red, where blue = no observed effect/interaction occurs but organism is unaffected/interaction ends after egestion; green = minor effect, interaction occurs over a short or long period of time/affects assemblage formation/particles are retained/short term energy loss; yellow = marginal effects, interaction causes reduction in function/behaviour and or feeding is affected/some tissue transfer of particles; orange = critical effect, the reduction in function caused by the interaction has subsequent biological effects; and, red = major effect, the biological processes affect growth and development/ mortality. When more than one effect was reported per study, the most severe effect is reported. Where no minimum size range was reported arrows pointing towards a low size range are used. (L)= planktonic larvae

Molluscs

Bivalves are by far the most commonly used organisms in microplastic exposure studies. In the blue mussel microplastics were retained in the digestive tract and some studies reported the transfer to the haemolymph[2] and lysosomal system[3] triggering an

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[2] Haemolymph is a fluid, analogous to blood in vertebrates, which circulates in arthropod and molluscan bodies.

[3] Lysosomes are organelles found in animal cells that, among other functions, act as a waste disposal system of unwanted materials within the cells.

inflammatory response (Browne *et al*., 2008; von Moos, Burkhardt-Holm and Köhler, 2012). Whereas these studies demonstrate possible pathways of microplastic uptake by and distribution within a bivalve species, it is possible that under lower, environmentally relevant exposure conditions the efficiency of uptake and internal distribution may differ. Cellular effects associated with intake included alterations of immunological responses, neurotoxic effects and the onset of genotoxicity (Avio *et al*., 2015a). Beside these effects, a modulation of the transcriptomic profile of exposed organisms was also observed, indicating the onset of early molecular changes even in the absence of visible cellular alterations (Avio *et al*., 2015a). Paul-Pont *et al*. (2016) used a high concentration (0.032 mg L^{-1}) of micro PS particles (2 µm and 6 µm diameter) to study the accumulation of fluoranthene, as a model polycyclic aromatic hydrocarbon (PAH), in mussels (*Mytilus* spp.). After a seven day exposure and a seven day depuration period, a higher fluoranthene concentration was detected in mussels exposed to microplastics and fluoranthene, as compared to mussels exposed to fluoranthene alone. The authors also found that highest histopathological damages and levels of anti-oxidant markers were observed in mussels exposed to microplastics together with fluoranthene. The authors concluded, that under the experimental conditions of their study, the results suggested that microplastics led to direct toxic effects at tissue, cellular and molecular levels, and modulated fluoranthene kinetics and toxicity in marine mussels. Furthermore, microplastics significantly affected the reproduction (and subsequent population growth) of the Pacific cupped oyster (*Crassostrea gigas*) when adults were experimentally exposed to virgin 2 µm and 6 µm PS microspheres (0.023 $mg \cdot L^{-1}$) for two months during a reproductive cycle. Researchers observed feeding modifications in the form of a shift in energy allocation from reproduction to structure growth with elevated maintenance costs, which had significant impacts on offspring by reducing larval yield and development (Sussarellu *et al*., 2016). At present, only one study observed the interaction of gastropods with microplastics. A grazing species of gastropod mollusk, the periwinkle (*Littorina littorea*), was observed to feed on contaminated algae and subsequently microplastics were identified in the stomach, gut and faeces indicating that the particles did not accumulate in individuals (Gutow *et al*., 2016).

Crustaceans

Planktonic copepods, isopods and decapods have all been exposed to microplastic in laboratory trials. Work carried out by Cole *et al*. (2013) has clearly demonstrated the uptake of fluorescent microspheres by zooplankton from the water column. Analysis of the impacts revealed a reduction in feeding after plastic ingestion but also egestion of the microspheres (Cole *et al*., 2013). When chronically exposed to levels of microplastic over successive generations increased mortality rates were observed (Lee, K. W. *et al*., 2013). Jeong *et al*. (2017) observed developmental delays and reduced fecundity in the marine copepod *Paracyclopina nana* exposed to 0.05 µm nanoplastics, whereas individuals exposed to microplastics (0.5 µm and 6 µm) did not show overall developmental delays or reduced fecundity. The marine isopod *Iotea emarginata* was unable to distinguish between clean and plastic contaminated foods; however, plastics were readily egested and did not accumulate in the gut or negatively affected growth or survival (Hämer *et al*., 2014). Sandhoppers (*Talitrus saltator*) fed on microplastics mixed with food, were also seen to ingest microspheres; again, plastics were eliminated over a 24 hour to 1 week period, and no detrimental effects were observed (Ugolini *et al*., 2013). Farrell and Nelson (2013) fed green crabs homogenised samples of pre-exposed blue mussels. The microspheres (0.5 µm) appeared to have translocated from the crab gut into its tissues. However, only 0.3 percent of the exposed particles were found in the crab's haemolymph. Interestingly, green crabs also take up microplastics through the gills (Watts *et al*., 2014), and plastic ingestion has been seen to reduce food consumption and growth (Watts *et al*., 2015). Another decapod crustacean, the Norway lobster (*Nephrops norvegicus*), ingested and retained microplastic fibres present in food. Long-term exposure reduced the nutritional health and availability of energy stores (Welden and Cowie, 2016b).

Fishes

Laboratory investigations of the interactions of fish and microplastics are less common than those on bivalves. It can be difficult to obtain ethical licences to carry out some investigations (for example, there is a European Directive on animal experimentations on vertebrates 2010/63/CE). Fish may ingest microplastics for several reasons, they may confuse microplastics with prey items, or ingest them during filter feeding. Once ingested microplastics (3 mm-5 mm) have been observed to pass directly through the digestive system and be excreted (Dos Santos and Jobling, 1991), and during a recent study it was proposed that microplastics (50 µm - 500 µm) in goldfish (*Carassius auratus*) would not accumulate over successive meals as microplastic retention times were similar to those of other contents of the gastrointestinal tract (Grigorakis, Mason and Drouillard, 2017). However, gastric obstruction has been proposed (Mazurais *et al*., 2015). Smaller particles (< 600 µm) translocated from the digestive tract to liver tissues of Flathead grey mullet (*Mugil cephalus*) (Avio *et al*., 2015b). However, the numbers of particles in the liver were two orders of magnitude lower than in the corresponding gastrointestinal tract. Furthermore, microplastics were isolated in the gills, liver and digestive tract of the Zebra danio (*Danio rerio*), which caused inflammation, oxidative stress and disrupted energy metabolism (Lu *et al*., 2016). Food chain transfer of microplastics is the process by which a prey item containing microplastics is consumed by a predator. Dietary exposure in this way may be responsible for microplastic presence in predatory fish (Batel *et al*., 2016). Mattsson *et al.* (2015) observed behavioural and metabolic effects of pre-exposed prey on predatory Crucian carps (*Carassius carassius*), these predators displayed reduced feeding and activity.

Physiological effects of microplastic exposure include intestinal tract alterations and compromised intestinal function as was observed in the European seabass (*Dicentrarchus labrax*) after 90-day exposure through feed pellets (0.1 percent polyvinyl chloride (PVC)) (Pedà *et al*., 2016). Fish behaviour is also affected by microplastic exposure: the common goby (*Pomatoschistus microps*) displayed reduced predatory performance, abnormal swimming behaviour and lethargy (De Sá, Luís and Guilhermino, 2015; Ferreira P. *et al*., 2016; Oliveira *et al*., 2013).

Only two studies used environmentally relevant concentrations of microplastics and associated contaminants. Microplastic exposure (concentration of 0.008 mg L^{-1} of polyethylene (PE)) induced liver toxicity, hepatic stress and changed endocrine function, as well as gene expression in Japanese medaka, *Oryzias laticeps* (Rochman *et al.*, 2013; 2014). Remarkably, Rummel *et al.* (2016a) did not observe any effect related to microplastic exposure in rainbow trout (*Onchorhynchus mykiss*).

4.2 OBSERVATIONS OF BIOTA INTERACTING WITH MICROPLASTICS IN THE ENVIRONMENT

4.2.1 Ingestion by commercially important species from field observations

Observations of microplastic uptake by environmentally exposed organisms have now been reported in a range of habitats, including the sea surface, water column, benthos, estuaries, beaches and aquaculture (e.g. Lusher 2015; GESAMP, 2016), as well as the deep sea (Taylor *et al*., 2016). The diversity of the organisms studied and the habitats from which they are sampled require a range of collection techniques: the sampling method employed is determined by the research question, available resources, habitat and target organism (Lusher *et al*., 2017; see Annex 1). Over 220 different species have been found to consume microplastic debris *in natura* (GESAMP, 2016; UNEP, 2016). Of this number, 58 percent were commercially targeted species. It is also important to understand microplastics ecology (interactions with fauna and flora) above the individual level, and develop study strategies that allow for the assessment of populations (including different ontogenetic phases), assemblages and communities. Studies must also consider that habitat use and type, as well as age and length are strong drivers of interactions with environmental variables, including contaminants, and that a long-lived individual might be contaminated at different times and rates during ontogeny (e.g. Ferreira *et al.* 2016). In the following sections information is provided for commercially important shellfish (bivalves and crustaceans) and fishes. Relevant general information regarding microplastics occurrence in seabirds and sea mammals is presented in Box 4.2.

BOX 4.2
Seabirds and marine mammals

The occurrence of microplastics in seabirds and marine mammals has been documented and may be of ecological and human health concern.

Many species of birds are consumed by humans. Almost every seabird species studied presented one individual with microplastics in their digestive tracts. The method used to quantify plastic in the stomach contents of Northern Fulmars (*Fulmarus glacialis*) is being rapidly adopted as an internationally standardized method of monitoring spatial and temporal trends in plastic pollution (van Franeker *et al.*, 2011).

Microplastics can be ingested by marine mammals, although the source of microplastic is hard to identify. Microplastics have been found in the stomachs of harbor seals, *Phoca vitulina* (Bravo Rebolledo *et al.*, 2013), beaked whales, *Mesoplodon mirus* (Lusher *et al.*, 2015a), and baleen whales, *Megaptera novaeangliae* (Besseling *et al.*, 2015). Furthermore microplastics were found in the scats of fur seals, *Arctocephalus* spp (Eriksson and Burton, 2003). Microplastics in marine mammals may have originated from feeding in the water column or through ingestion of prey already containing microplastics.

Indeed marine mammals, in particular the baleen whales, may be considered as a sentinel of ocean health concerning microplastic ingestion and exposure to plastic additives (Fossi *et al.*, 2012, 2014, 2016; Baini *et al.*, 2017). Furthermore, these long-lived top predators may be effective indicators for chronic or slow developing pathologies. Some species of marine mammals may provide an early indication of potential adverse health effects and provide insight into the toxic mechanisms of a given hazardous agent, such as microplastics or their associated chemicals.

4.2.2 Shellfish (bivalves and crustaceans)

Of the invertebrates examined in studies of wild and laboratory cultured animals, comparatively few have been of economic or dietary importance. This is pertinent when the annual volumes of such species produced and consumed are considered. One such species is Norway lobster which appears to retain plastics for long periods; this retention of plastic may not only affect the health of the animal, but the profitability of the fishery. Examination of wild caught Norway lobsters from stocks around Scotland has shown large variability, with greatly increased plastic abundance in the highly-impacted Clyde Sea area (Murray and Cowie, 2011; Welden and Cowie, 2016a) and low plastic uptake around the Minch and Orkney Islands (Welden and Cowie, 2016a). Another decapod crustacean, the common shrimp (*Crangon crangon*), sampled from around the North Sea demonstrated microplastic uptake by 63 percent of sampled animals; however, it is not clear what effect this has upon the organism (Devriese *et al.*, 2015). Also, a study of microplastic ingestion by the invasive Chinese mitten crab (*Eriocheir sinensis*) from Baltic Sea coastal waters in Poland and the Tagus Estuary in

Portugal showed that as many as 13 percent of the 302 analysed individuals contained microplastics in the form of strands and balls (Wójcik-Fudalewska, Normant-Saremba and Anastácio, 2016). Several mollusc species have also been seen to take in microplastics. Both farmed and wild blue mussels contained microplastic fibres (e.g. Li *et al.*, 2016; Mathalon and Hill, 2014). The brown mussel (*Perna perna*) has also been reported to take up plastics, with occurrence recorded in 75 percent of individuals sampled from the Santos Estuary, Brazil (Santana *et al.*, 2016); similar results were observed for the Pacific cupped oyster reared in Atlantic Ocean coastal waters (van Cauwenberghe & Janssen, 2014). Manila clams (*Venerupis philippinarum*) both farmed and wild caught did not appear to have any difference in microplastics concentration (range from 0.07 to 5.47 particles per g) which suggests that shellfish aquaculture does not appear to be increasing microplastic concentrations in the farmed organisms (Davidson and Dudas, 2016). In Chinese waters wild blue mussels contained higher levels of microplastics than farmed mussels, which was attributed to better water quality at aquaculture sites (Li *et al.*, 2016). Another method of assessing microplastic contamination in species for human consumption has been the collection of samples at markets and other suppliers. Comparisons between shop-bought and wild caught blue mussels revealed plastic contamination in both samples (Van Cauwenberghe and Janssen, 2014). Furthermore, studies of nine species of bivalves in Asian markets revealed extensive microplastic contamination (Li *et al.*, 2015). In these studies, fibres were the primary source of microplastic contamination; while this form of microplastic is widely reported throughout the marine environment, the scale and diversity of possible sources prove hard to identify or control. Quantitative data on microplastics contents in bivalves are given in Table 5.1.

4.2.3 Fishes

Several studies have identified the presence of microplastics in the digestive tracts of wild-caught fish. Increasingly, researchers have been targeting commercial species due to the potential implications of microplastics on the commercial market. Field studies have demonstrated microplastic ingestion by many commercial (benthic and pelagic) fish species, for example, from the English Channel (Lusher, McHugh and Thompson, 2013), the North Sea (Foekema *et al.,* 2013), the Baltic Sea (Rummel *et al.,* 2016b), the Indo-Pacific Ocean (Rochman *et al.*, 2015; Jabeen *et al.*, 2016), the Mediterranean Sea (Bellas *et al.*, 2016; Güven *et al.*, 2017), the Adriatic Sea (Avio *et al.*, 2015b) and the North Eastern Atlantic (Neves *et al.*, 2015). However, the quantities observed in fish guts are generally very low, usually in the range < 1- 2 particles per individual that ingested microplastics. The percentage of individuals that ingested microplastics ranged from 2.6 percent in the North Sea (Foekema *et al.*, 2013; Rummel *et al.*, 2016b) to 18 percent in the Central Mediterranean (Romeo *et al.*, 2015), to 28 percent in the Adriatic Sea (Avio *et al.*, 2015b) and 41 percent in the Eastern Mediterranean (Güven *et al.*, 2017). Interestingly, fish collected from fish markets contained microplastics, including 25 percent of fish caught off the west coast of the United States of America and 28 percent of fish caught in Indonesian waters (Rochman *et al.*, 2015). Microplastics have been similarly observed within market-purchased freshwater fish including Nile Perch (*Lates niloticus*) and Nile Tilapia (*Oreochromis niloticus*) from Lake Victoria (Tanzania), with 20 percent of each species containing microplastics in their digestive tracts (Biginagwa *et al.*, 2016). Twenty-seven species of fish collected from Shanghai fish markets (Jabeen *et al.*, 2016) originating from marine (Yangtze Estuary, South China Sea and East China Sea) and freshwater (Lake Taihu) sources also had varying degrees of microplastic contamination (from 1.1 to 7.2 items per individual). Recently, microplastics have also been found in the digestive tract of wild fish larvae (2.9 percent of individuals sampled) belonging to commercially important species from the English Channel (Steer *et al.*, 2017). Although it is clear that microplastics are ingested by many species of commercial fish, we know little about the impact of their consumption. Microplastics may be egested along with faecal material or retained within the digestive tract. Translocation to other tissues does not occur or is very low for the smaller microplastics (< 600 µm) (see Chapter 5).

4.3 ECOLOGICAL IMPLICATIONS

Despite the increasing literature on marine plastic contamination, there is very little information on its effects at ecosystem, habitat, population or even individual level. Laboratory experiments do not identity the effects of microplastics on individuals in nature because exposure (microplastic shape, size, concentrations) and habitat conditions are different to the wild and include other stressors. Most of the known interactions and effects relate to pre-existing environmental concerns such as the transport of persistent bioaccumulative and toxic compounds (PBTs) and invasive species. Many of the projected outcomes of increasing microplastic abundance remain academic speculation based on an understanding of the physiology of the organism or group in question. This is further complicated by the limited knowledge of the physiological effect of microplastics *in natura*.

Few laboratory studies have been conducted with environmentally relevant concentrations of microplastics and associated contaminants. Furthermore, studies should also be relevant in terms of all exposure pathways. For example, Besseling *et al*. (2013) used naturally contaminated sediment as well as environmentally realistic spiked sediment while accounting for all possible exposure pathways: water, sediment and polystyrene (PS) microparticle ingestion by the lugworm (*Arenicola marina*) under pre-equilibrated conditions. A low PS concentration increased bioaccumulation of contaminants in lugworms which may have been due to physiological stress and/or PS acting as a vector for contaminants; however, at higher concentrations of PS microparticles decreased bioaccumulation occured which was partially adscribed to PS acting as a sink for contaminants. Furthermore, Sussarellu *et al.* (2016) exposed Pacific cupped oysters to virgin PS microplastics (2 µm and 6 µm diameter) for two months during a reproductive cycle at a concentration of 0.023 mg L^{-1}, which is within the upper range observed for >333 µm microplastics in the environment. The authors observed that oysters preferentially ingested the 6 µm micro-PS over the 2 µm diameter particles. Also, the consumption of microalgae and absorption efficiency were significantly higher in exposed oysters, suggesting compensatory and physical effects on both digestive parameters. After two months, the exposed oysters displayed significant decreases in oocyte number (–38 percent), oocyte diameter (–5 percent) and sperm velocity (–23 percent). The D-larval[4] yield and larval growth of offspring derived from exposed parents decreased by 41 percent and 18 percent, respectively, when compared with offspring of non-exposed oysters.

Hence, in principle, microplastic ingestion by bivalves and fish may affect individual physiology, metabolism, body condition, growth, contaminant body burden and reproductive success, but the evidence has to be considered very weak (Ziccardi *et al*., 2016). Population level studies have not been conducted and at present there is no evidence of negative effects from microplastic ingestion in wild and farmed aquatic organisms. Studies at the population level are hindered not only by the problem of scale and associated costs, but also by the multiple environmental and human induced stressors to which farmed and wild animals are subjected, which may veil the possible role played by microplastics in this multifactorial context. Farmed aquatic animals in relatively closed systems (i.e. fish ponds) at different levels of ambient exposure to microplastics may be an adequate starting point for these studies.

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[4] The D larval stage is an early development stage of molluscan bivalve larvae with a characteristic D shaped shell.

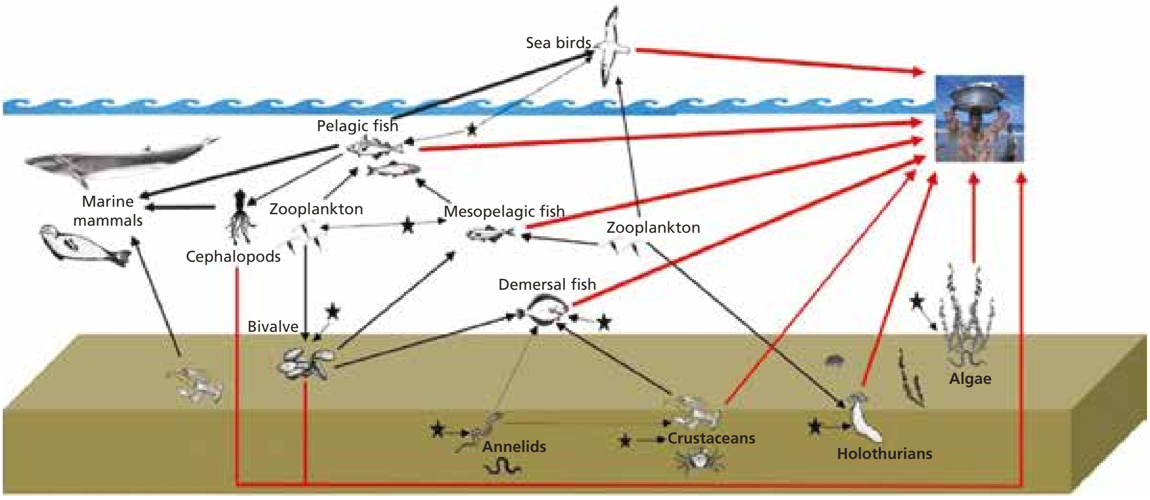
A risk analysis framework is a useful tool for studying the possible impacts of microplastics at ecological levels of organization. This framework allows the use of different data (qualitative, semi-quantitative and quantitative) sources to analyse the possible impacts and risks associated with a particular hazard, as well as identify information and data gaps (UNEP, 2016).

4.3.1 Accumulation of microplastics through trophic transfer

A common concern about many persistent pollutants is the possibility of trophic transfer and bioaccumulation. PBTs build up in the tissues of organisms and accumulate up the food chain, leading to increased body burdens in higher trophic levels. However, in contrast to PBTs, most microplastics ingested will not translocate into the tissues of their hosts, and thus would not be retained and concentrated at subsequent higher trophic levels. If trophic transfer were to occur regularly, animals at higher trophic levels would be at increased risk of negative impacts, such as damage to and irritation of the gut lining and reduced nutrient uptake. To date there is only initial evidence which suggests potential of trophic transfer of microplastic in wild caught organisms (Eriksson and Burton 2003; Romeo *et al.*, 2015; Welden and Cowie 2016a). Most laboratory evidence for distribution of microplastics has been obtained in mesocosm studies at highly exaggerated exposure scenarios, which may not directly apply to natural much lower exposure. One exception to this is animals feeding by scavenging, which may ingest portions of prey larger than themselves. Examples of species where this may occur include large decapod crustaceans such as Norway lobster (Welden and Cowie 2016a). Microplastics have been observed in large pelagic fish and it has been hypothesized that microplastics present in these species may have originated from prey items rather than from directly feeding in the environment (e.g. Romeo *et al.*, 2015). Interactions of marine biota and microplastics and possible trophic pathways are depicted in Figure 4.2.

FIGURE 4.2

Interaction of microplastics with marine organisms and potential trophic transfer to humans



Black stars represent microplastic particles and the black dotted arrows indicate an observed interaction between organism and particle (direct ingestion/uptake). The black arrows indicate indirect ingestion of MPs (potential trophic transfer). The red lines indicate potential route of microplastics to humans following ingestion of seafood. Finally, arrow thickness represents potential bioaccumulation of particles through the foodweb

4.3.2 Effects at the population and community levels

Most of the available literature on the effects of microplastics to biota pertains to individuals rather than on a population level. This is a critical factor that requires further investigation since there is no theoretical reason why trophic transfer would not occur. Browne *et al.* (2015b) raised the notion that the level of disturbance caused by microplastics may not reach a level beyond that of an individual if the effects are restricted to physiological functioning. As has been evidenced in this document, as well as numerous reviews, most work focus on effects of microplastics (and any associated chemicals) on the lower levels of organization: cellular to individual, without investigating ecological effects on populations or assemblages (Rochman *et al.*, 2016). Without information on these higher levels of organization, ecological assessments lack critical information about the capacity of microplastics to alter ecological structures and processes (assemblages and food webs, competition and predation) (Browne *et al.*, 2015b). However, it might be necessary to conduct long-term studies in order to detect the phenomenon in the wild, since the large number of ecological variables, and the input from mixed sources will possibly cause some level of noise in detecting microplastics signal of exposure and damage to more complex biological systems.

Population level effects resulting from reduced condition indexes, lowered fecundity and larval survivorship are touched upon in a few experimental studies. Welden and Cowie (2016b) note the potential effect of lowered body mass in females on egg production could affect populations. Experiments of nanoplastics and microplastic exposure on larval development resulted in developmental defects (Della Torre *et al.*, 2014; Nobre *et al.*, 2015). Additionally, the potential negative impacts of microplastics on recruitment may have an effect on populations and the assemblages that rely upon them; however, there is currently no evidence of this from environmental observations.

Only two studies are available on the potential effects of microplastics at the community level. In these investigations, conducted through outdoor mesocosm experiments, authors observed an effect of both biodegradable (polylactic acid) and conventional plastics (PE and PVC) on benthic assemblage structures and on the nutrient cycle. Concluding that the high doses of either tested polymers can decrease the species richness and the total number of organisms, as well as directly or indirectly reduce primary productivity of these habitats (Green, 2016; Green *et al.*, 2017).

One potential adverse population impact is the ability of microplastics to increase the availability of chemical contaminants, which may result in further ecotoxicological impacts. Models of the partitioning of contaminants between the water, sediments, plastics and organisms have indicated the potential for increased polychlorinated biphenyl (PCB) loads in lugworms (*Arenicola marina*) (Teuten *et al.*, 2009). However, an important factor should be considered, viz. the difference between the concentration of the contaminant in the microplastics and in the organism (i.e. the diffusion gradient). With fresh microplastics having a low level of contamination, the net movement of chemicals may be reversed: from an organism into the microplastic (Koelmans, Besseling and Foekema, 2014).

5. Microplastics in foods

Aside from a few reports on the occurrence of microplastics in beer (Liebezeit and Liebezeit, 2014), sugar and honey (Liebezeit and Liebezeit, 2013), and table salt (Yang *et al.*, 2015; Karami *et al.*, 2017), the majority of reports focus on their occurrence in seafood (EFSA, 2016). Thus, at present seafood appears to be the best studied source of dietary intake of microplastics. Because analytical methods for nanoplastics in foods are currently lacking (see section 5.2), there are no data on the occurrence of nanoplastics in foods.

5.1 FISH AS FOOD

The increased production of fisheries and aquaculture products has resulted in increased global per capita consumption. In 2013 fishery products represented around 17 percent of animal protein intake by the world´s population (FAO, 2016c). Also, in the last decades there has been a substantial increase in the proportion of fishery products used for direct human consumption. For example, in 2014, 87 percent of total marine capture production was used for human consumption and the remaining 23 percent, which accounted for 21 million tonnes, was destined to non-food products, especially fishmeal and fish oil (FAO, 2016b).

5.1.1 Fish consumption

World per capita apparent consumption (see Box 5.1) of fishery products averaged around 9.9 kg in the 1960s, increasing to 14.4 kg in the 1990s and reached a value of 19.7 kg in 2013; preliminary data indicate that this increasing trend has continued for 2014 and 2015 (FAO, 2016c). However, FAO data (FAO, 2016c) show that there are significant regional contrasts in fish consumption. For example, in Asia fish consumption has increased around three-fold from 1961 to 2013, from 7.8 kg/cap in 1961 to over 23 kg/cap in 2013, while in Africa the increase has been about two-fold from 4.8 kg/cap to 9.9 kg/cap. Fish consumption is highest in Oceania with 24.8 kg/cap in 2013, but this value is the result of a recent decreasing trend that started in 2004 when per capita consumption was around 26.3 kg. Moreover, there are significant contrasts within each continental area, for example, a sample of the three most populated countries by continent shows that in Asia consumption in 2013 was around 38 kg/cap in China, 31.8 kg/cap in Indonesia and 6.1 kg/cap in India. This contrast is particularly significant between the most populated countries in Africa, where per capita consumption in 2013 reached 22.1 kg in Egypt, 13.3 kg in Nigeria and 0.25 kg in Ethiopia. In America, the series shows more or less increasing trends from 1961 to 2013 and reached 21.6 kg/cap in the United States of America, 13.2 kg/cap in Mexico and 9.74 kg/cap in Brazil at the end of the time series. In Europe, among the three most populated countries, France has the highest per capita rate at 33.5 kg in 2013, while the Russian Federation consumed 22.9 kg and Germany 12.6 kg. Finally, in Oceania, Australia and New Zealand show very similar consumption rates in 2013 at 26.3 kg and 25.2 kg, respectively, while apparent consumption in Papua New Guinea was 15.4 kg.

BOX 5.1

Estimation of apparent fish consumption

Apparent fish consumption, as the name implies, is not a direct measurement of ingestion of fishery and aquaculture products. It is an estimate of the quantity in kg (wet-weight equivalent) of animal fishery and aquaculture products available for human consumption divided by the human population in a specified country or region.

1. Definitions used

   Live weight: The weight of finfish and shellfish at the time of their capture or harvest. Calculated on the basis of conversion factors from landed to nominal weight and on rates prevailing among national industries for each type of processing.

   Production: Refers to FAO capture and aquaculture statistics of all fish, crustaceans, molluscs and aquatic organisms excluding aquatic plants.

   Non-food uses: Including utilization of aquatic products for reduction to meal and oil, for feed and bait, for ornamental purposes, withdrawals from markets and any other non-food use of fish production (e.g. fertilizers, medical uses).

   Imports/exports: In accordance with the internationally-recommended practice, imports and exports statistics have been adjusted to include as imports fish caught by foreign fishing vessels and landed in domestic ports and as exports fish caught by domestic fishing vessels and landed directly in foreign ports. Exports include re-exports.

   Stock variations: Information on changes in stocks occurring between the production and the retail levels, or in levels of inventories.

2. Estimation of fish supply

The total fish available for apparent human consumption is derived by using the following equation:

Total food supply = production - non-food uses + imports – exports ± variation in stocks (all data expressed in terms of live-weight equivalent).

The total food supply is divided by country/region human population to obtain per capita apparent fish consumption.

Fish consumption is particularly relevant in island states. For example, in Iceland and the Faroe Islands apparent per capita consumption in 2013 was 92 kg and 87.3 kg, respectively. Also, in Oceania, small-island developing states such as Samoa and Tuvalu presented consumption rates in 2013 of 46.8 kg/cap and 43.3 kg/cap, respectively. Nonetheless, the consumption rates in these last two oceanic developing states is still below that observed in developed mainland states with high fish consumption, such as Portugal (53.8 kg/cap) the Republic of Korea (53.5 kg/cap), Norway (52.1 kg/cap) and Japan (48.9 kg/cap).

5.1.2 Benefits of seafood consumption

Consumption of fisheries and aquaculture products have well-established health benefits due to its unique nutritional composition, but in some cases fish accumulate significant levels of contaminants from the environment, resulting in some fish products potentially being harmful if consumed. In such cases the question is how to maximize the positive consequences of seafood consumption while minimizing the concurrent negative consequences.

The risks of consuming potentially contaminated foods have traditionally received greater attention than the benefits of eating them. However, there is now a growing focus on the risks of not consuming certain foods, and among them fish products, given their components known to benefit human health. The nutritional benefits of fish consumption are not only due to the long-chain polyunsaturated fatty acids (LCPUFAs) – docosahexaenoic acid (DHA) and eicosapentaenoic acid (EPA) – but also from amino acids and peptides derived from their proteins, micronutrients (vitamins, minerals) and possibly from other nutrients (e.g. taurine), all found in fish (Lund, 2013).

Fish consumption has for a long time been recognized in helping to prevent coronary heart disease (CHD). However, there is an increasing focus on fish as a natural source of DHA and iodine, both essential for the early development of the brain and neural system. For this reason, fish is of particular importance in diets for pregnant women and for children up to the age of two. These nutrients are almost exclusively found in foods from the aquatic environment. The role of fish in mitigating mental disorders, including depression and dementia (Morris *et al.*, 2016), is also receiving increased attention from scientists.

However, the presence of contaminants in some fish and fish products and other foods is of increasing concern to consumers. Some fish products are known to contain contaminants such as methyl mercury and other persistent, bioaccumulative and toxic compounds (PBTs) like dioxins, polychlorinated biphenyls (PCBs), furans and pesticides. There is a concern that microplastics could act as an agent for the transfer of many fat-soluble pollutants, such as PBTs, from the environment and into organisms such as fish that end up on our plates. The presence of persistent contaminants in seafood is a potential concern. In particular in fish caught in polluted waters or in large, long-lived predator species, the levels of contaminants might exceed the levels regarded as safe for consumption. The role of microplastics on the levels of pollutants in fishery and aquaculture products frequently consumed, and on potential adverse health effects is highlighted further below and in Chapter 6.

It is well known that fish can be a source of PBTs such as dioxins in populations that consume fish frequently. However, the occurrence of dioxins among individuals in these populations is generally not higher than in populations having low fish consumption (Sasamoto *et al.*, 2006; Mazet, Keck and Berny, 2005). In these cases, reducing the consumption of fish will likely not reduce the exposure to dioxins, which will probably be the same for individuals even if they significantly reduce their consumption of fish. When consumption of a food can be associated with both potential health risks and benefits, risk managers try to identify an intake level that minimizes risks and maximizes benefits. It is particularly important to establish such a guideline as to when consumption may result in a potentially harmful level of exposure (EFSA, 2007).

Advice on limiting the consumption of fish for vulnerable groups, such as children and pregnant women, is being given by many public health authorities. While the intention is only to limit consumption of products believed to have elevated levels of contaminants, the effect in some cases has been a significant reduction in seafood consumption. However, such a reduction could result in a diet that contains insufficient levels of essential nutrients. Both children and adults run this risk. As LCPUFAs are essential in the early development of the brain and neural system in children, advice aiming to limit the consumption of contaminated fish must be couched in such terms that not all fish is given a "bad name". Similarly, as seafood consumption reduces cardiovascular diseases among the adult population, messages intended to reduce the exposure of fish products to contaminants should go hand in hand with the promotion of safe fish products.

5.1.3 Balancing the risks and benefits

Emerging concerns such as the potential human health impacts of microplastics in seafood should not be seen in isolation, but in the context of the potential health benefits of consuming the same food. In light of this, an ideal solution is one that consists of sound, science-based advice that weighs the benefits and costs for human health of consuming fish.

Some studies (EFSA, 2015b; Mozaffarian and Rimm, 2006) have tried to balance the positive and negative sides of consuming foods of high nutritional value which may also be a source of contaminants. However, the procedures used have been controversial, and experts in this field maintain that new procedures need to be developed in order to carry out assessments of the risks and benefits to human health of consuming fish and other seafood (EFSA, 2010).

On the request from Codex Alimentarius, FAO and the World Health Organization (WHO) held an Expert Consultation on the Risks and Benefits of Fish Consumption in 2010 (FAO, 2011), the experts concluded that consumption of fish provides energy, protein and a range of essential nutrients, and that eating fish is part of the cultural traditions of many peoples. In some populations, fish and fishery products are a major source of food and essential nutrients, and there may be no alternative and affordable food sources for these nutrients.

To minimize risks in target populations, the Expert Consultation recommended that states should emphasize: (i) that fish consumption reduces CHD mortality in the adult population; and (ii) that fish consumption improves the neurodevelopment of foetuses and infants and is therefore important for women of childbearing age, pregnant women and nursing mothers. In order to provide sound advice to different population groups, it will also be important to develop, maintain and/or improve regional databases of the specific nutrients and contaminants in the fish available for consumption. Risk management and communication strategies that aim to minimize risks and maximize benefits from eating fish should be developed and evaluated (FAO, 2011).

5.2 ANALYTICAL METHODS FOR MICROPLASTICS IN FOODS

EFSA (2016) reviewed analytical methods for microplastics in foods. In summary, methods for microplastics include the following steps: extraction and degradation of biogenic matter; visual detection and quantification (enumeration); confirmation and/or characterization of the plastic identity of the particles. If visual-assisted microscopy is used, the lower size limit of detection is in the low micrometre range. Crucial factors to consider are adequate precautions to prevent the contamination of samples with microplastics from air, clothes, equipment and/or reagents used in the analysis, and confirmation that the particles detected are indeed microplastics. Reference methods for sampling or analysis of microplastics in foods have not yet been described.

For nanoplastics, microscopic methods which are used for microplastics are inadequate because of the small size of the nanoplastics. As a result, the resolution or contrast between nanoplastics and the food matrix is too low, which hampers their detection. Nanoplastics have only been detected in simple model systems, e.g. degradation of polystyrene (PS) into nanoplastics in pure water (Lambert and Wagner, 2016). Analytical methods that are adequate to detect and quantify nanoplastics in the environment or food chain have not been sufficiently developed (Koelmans, Besseling and Shim, 2015; EFSA, 2016).

5.3 MICROPLASTICS IN BIVALVES

There is only a limited number of publications on the occurrence of microplastics in marine bivalves and they mostly refer to wild-caught mussels (Table 5.1). Except for the study by Li *et al.* (2016), sample sizes of the studies are rather limited. Reports are restricted to Europe, North America, Brazil and China. Overall, the lowest contents of microplastics, less than 0.5 particles per g of soft tissue, were found in Europe. The highest contents were observed in Newfoundland, Canada, about 100 fold higher than the levels measured in Europe (Mathalon and Hill, 2014). In this last study, ambient particles in laboratory air were very high and blank samples indicated that laboratory contamination could contribute 25 microplastic particles per g soft tissue. Adjusting for this background, would lead to contents comparable with those found in bivalves from China (Li *et al.*, 2015).

TABLE 5.1
Occurrence of microplastics in species of bivalves destined for human consumption

Species[a]	Location	Average number of particles per g soft tissue (SD)	Type of particles	Reference
M. edulis	Germany	0.36 (0.07); n = 72 Size: 5–25 µm (85%), > 25 µm (15%)	Fragments, spheroids	Van Cauwenberghe and Janssen, 2014
C. gigas	Germany	0.47 (0.16); n = 21 Size: 5–25 µm (55%), > 25 µm (45%)		
M. edulis	Belgium France Netherlands	0.20 (0.30); n = 6 Size: 20–90 µm	Partial fragments	Van Cauwenberghe *et al.*, 2015a
M. edulis	Belgium	0.37 (0.22); n = 9 Size: 200–1500 µm	Fibres	De Witte *et al.*, 2014
M. edulis	New Foundland	34 (14)[b]; n = 45 Size: no data	Fibres, spheroids	Mathalon and Hill, 2014
M. edulis	China	2.2 (0.9–4.6)[c]; n~1100 Size: 33–4700 µm (fibres)	Fibres Fragments	Li *et al.*, 2016
9 different species	China	4.0 (2.1–10.5)[d]; n = 9 Size: 5–250 µm (60%), 250–5000 µm (40%)	Fragments, fibres, pellets	Li *et al.*, 2015
Perna perna	Brazil	75% contained ≥ 1 particle; n = 30 Size: < 5000 µm	Fragments	Santana *et al.*, 2016
V. philippinarum	British Columbia	Wild: 0.84 (0.85) Farmed: 1.13 (0.66) n= 54 Size: no data	Fibres, films, fragments	Davidson and Dudas, 2016

[a] *M. edulis – Mytilus edulis; C. gigas – Crassostrea gigas; V. philippinarum – Venerupis philippinarum*

[b] only microfibres, spherical particles not quantified, values calculated from Mathalon and Hill (2014) using a weight of 4 g of soft tissue per mussel

[c] average and range

[d] median and range

5.4 MICROPLASTICS IN OTHER INVERTEBRATES

So far, microplastic contents of crustacean species, viz. the common shrimp (*Crangon crangon*) and the Norway lobster (*Nephrops norvegicus*) have only been determined in coastal waters of the North Sea and Irish Sea. Sampling locations for common shrimps included the coastal waters of Belgium, France, the Netherlands and the United Kingdom. Contents ranged from 0.03 to 1.92 microplastic particles per gram wet weight, depending on location and sampling date (Devriese *et al.*, 2015). It was shown that only temporal differences were significant with higher concentrations in October compared to March. An average content of 0.75 microplastic particles per gram wet weight (SD = 0.53) was calculated from the data of 165 samples presented. Comparing whole and peeled shrimps, it was revealed that no particles were present in the abdominal muscle tissue, which constitutes usually the edible part (Devriese *et al.*, 2015). Peeling will remove most of the digestive tract, the head and the gills, which apparently contain the major part of the microplastics. Because peeling not always completely removes the digestive tract, Devriese *et al.* (2015) proposed to use 90 percent as a peeling factor.

Stomach contents of Norway lobsters from the Irish Sea contained microplastics (Murray and Cowie, 2011; Welden and Cowie, 2016a). Murray and Cowie (2011) found that 83 percent of the animals were contaminated. However, only Welden and Cowie (2016a) determined the contents of microplastics in the gut and found, in samples from the Irish Sea area, an average weight of microplastics per individual of 0.40 mg with a maximum of 0.80 mg. Because the gastrointestinal tract is generally removed before consumption, microplastics detected in the guts of Norway lobsters will present a limited threat to human health.

Echinoderms are highly consumed in some regions and ingestion of nylon and polyvinyl chloride (PVC) microplastic fragments have been observed under laboratory conditions in four species of sea cucumbers (Graham and Thompson, 2009). In most cases, the gastrointestinal tract of echinoderms is ingested but in wild populations microplastics have only been observed in one specimen of unidentified sea cucumber species from deep-sea samples (Taylor *et al.*, 2016).

5.5 MICROPLASTICS IN FINFISH

Ingestion of microplastics has been observed in a relatively large number of fish species used for human consumption from the Pacific, Atlantic and Indian oceans, as well as the Mediterranean Sea, albeit at relatively low concentrations of around one to two items per individual (Lusher 2015, GESAMP, 2016). For example, microplastics have been observed in the gastro-intestinal tract in 11 out of the 20 most important species and genera of finfish that contribute to global marine fisheries (FAO, 2016b). These species are chub mackerel (*Scomber japonicus*), Atlantic herring (*Clupea harengus*), Atlantic mackerel (*Scomber scombrus*), Japanese anchovy (*Engraulis japonicus*), Atlantic cod (*Gadus morhua*), European pilchard (*Sardina pilchardus*), blue whiting (*Micromesistius poutassou*), European sprat (*Sprattus sprattus*), king mackerel (*Scomberomorus cavalla*) from the *Scomberomorus* spp group, shortfin (*Decapterus macrosoma*) and Amberstripe (*Decapterus muroadsi*) scads from the *Decapterus* spp group, and Indian oil sardine (*Sardinella longiceps*) from the *Sardinella* spp group (Brate *et al.*, 2016; Collard *et al.*, 2015, 2017; Foekema *et al.*, 2013; Güven *et al.*, 2017; Liboiron *et al.*, 2016; Lusher, McHugh and Thompson, 2013; Miranda and Freire de Carvalho-Souza, 2016; Neves *et al.*, 2015; Ory *et al.*, 2017; Rochman *et al.*, 2015; Rummel *et al.*, 2016b; Sulochanan *et al.*, 2014; Tanaka and Takada, 2016).

Species of local/regional importance also present microplastics in their gastro-intestinal tract, and this trend needs to be examined especially at estuaries of rivers formerly reported as contaminated (e.g. Dantas *et al.* 2012; Ferreira *et al.*, 2016; Vendel *et al.*, 2017). However, microplastics have only been observed in the gastro-intestinal tract of fish and, as most fish species are gutted before consumption by humans; direct human exposure to microplastics from fish will be negligible. Nevertheless, small marine pelagic fish species, such as sardines, anchovies, and a number of other small sized freshwater fish are sometimes eaten whole, including the digestive system. Small indigenous freshwater species such as mukene (*Rastrineobola argentea*) and kapenta (*Limnothrissa miodon*) are eaten whole and are important in local diets in East African countries. Likewise, small species like mola (*Amblypharyngodon mola*) and darkina (*Esomus danricus*) are also eaten whole and are commonly consumed in rural diets in Bangladesh. Presence of microplastics has been found in small freshwater species such as the gudgeon *Gobio gobio* (Sanchez, Bender and Porcher, 2014) and in small marine pelagic species from the Pacific and Atlantic oceans such as silversides (*Stolephorus commersonnii*, Kripa *et al.*, 2014*; Stolephorus heterolobus*, Tahir and Rochman, 2014; *Atherinella brasiliensis, Alves et al., 2016*), Pacific anchovy (*Engraulis japonicus*, Rochman *et al.*, 2015, Tanaka and Takada, 2016) and European anchovy (*Engraulis encrasicolus*, Collard *et al.*, 2017) . However, little knowledge on levels of microplastics in small sized fish exists.

5.6 CONTAINMANTS AND ADDITIVES ASSOCIATED WITH MICROPLASTICS

5.6.1 Plastic additives

All additives together account for around 4 percent of the total weight of plastics produced. (Andrady and Neal, 2009; Lambert, Sinclair and Boxall, 2014). However, the percentage of additives can vary significantly. In some cases additives make up half of the total material (Lithner, Larsson and Dave, 2011). Microplastics in oceans, coastal regions and deposited at beaches have been found to contain these additives: polybrominated diphenyl ethers (PBDEs) from 0.03 nanograms per gram (ng/g) to 50 ng/g, bisphenol A (BPA) from 5 ng/g to 200 ng/g, nonylphenol (NP) from 20 ng/g to 2 500 ng/g and octylphenol (OP) from 0.3 ng/g to 50 ng/g (Teuten *et al.*, 2009). Data on other additives are only sparingly available.

5.6.2 Plastic monomers and polymers

Apart from being an additive, BPA is also used as a building block of polycarbonate plastics, and in the previous section (5.6.1) contents of BPA in microplastics have been presented. Globally, styrene monomers, dimers, and trimers have been detected in seawater and sand from coastal regions, likely originating from polystyrene (PS) litter (Kwon *et al.*, 2014; 2015) which is a frequently found contaminant in aquatic environments.

5.6.3 Persistent, bioaccumulative and toxic compounds (PBTs)

Because of their hydrophobic properties, persistent organic pollutants accumulate heavily in microplastics floating in the marine environment (see section 3.4). Indeed, microplastics in oceans, coastal regions and deposited at beaches have been found to contain high concentrations of these pollutants: non-dioxin like polychlorinated biphenyls (PCBs) from 0.01 ng/g to 2 970 ng/g; polycyclic aromatic hydrocarbons (PAHs) from 4 ng/g to 44 800 ng/g; Dichlorodiphenyltrichloroethane (DDT), and analogues DDD and DDE, from 2 ng/g to 2 100 ng/g (Teuten *et al.*, 2009; Pelletwatch, 2016, Antunes *et al.*, 2013; Gauquie *et al.*, 2015). Remarkably, concentrations of dioxins in microplastics have not been published.

5.6.4 Metals

Pellets collected along the southwestern shores of England contained metal concentrations ranging from low ng/g for cadmium (Cd), nickel (Ni) and chromium (Cr), to high 7.7 micrograms per gram (µg/g) for copper (Cu), 10.3 µg/g for lead (Pb), 171 µg/g for aluminium (Al), 290 µg/g for zinc (Zn), 308 µg/g for manganese (Mn), and 314 µg/g for iron (Fe) (Holmes, 2013). In a more recent study, also along the Southwestern English shore, pellets showed maximum values of 3 390 µg/g for Cd and 5 330 µg/g for Pb (Massos and Turner, 2017).

5.6.5 Pathogens

It has been documented that plastic debris can act as a substrate for diverse microbial communities (Harrison *et al.*, 2011, 2014; Zettler, Mincer and Amaral-Zettler, 2013; McCormick *et al.*, 2014). Microorganisms, including plastic decomposing organisms and pathogens have been shown to colonize microplastics. Furthermore, in the ocean such communities have been shown to be distinct from microbial communities in the surrounding surface water (Zettler, Mincer and Amaral-Zettler, 2013; Debroas, Mone and Ter Halle, 2017). However, the relevance to seafood and the consequences to human health are unknown.

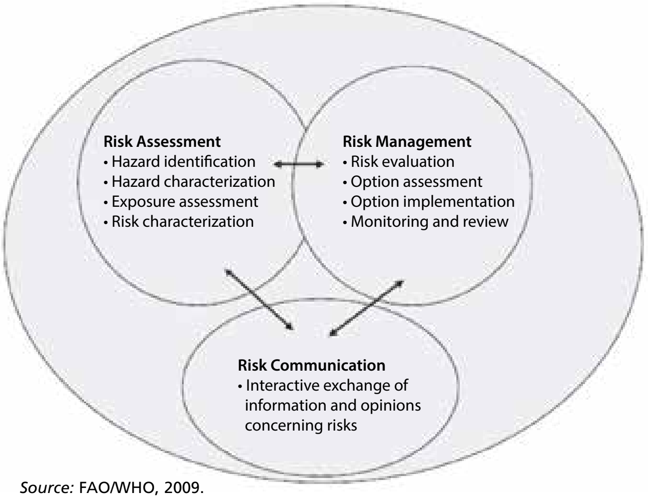
6. Risk profiling of microplastics in aquaculture and fishery products

6.1 INTRODUCTION

Risk analysis is used to develop an estimate of the risks to human health and safety, to identify and implement appropriate measures to control the risks, and to communicate with stakeholders about the risks and measures applied. It can be used to support and improve the development of standards, as well as to address food safety issues that result from emerging hazards or breakdowns in food control systems. It provides food safety regulators with the information and evidence they need for effective decision-making, contributing to better food safety outcomes and improvements in public health. Regardless of the institutional context, the discipline of risk analysis offers a tool that all food safety authorities can use to make significant gains in food safety.

For instance, risk analysis can be used to obtain information and evidence on the level of risk of a certain contaminant in the food supply helping governments to decide which, if any, actions should be taken in response (e.g. setting or revising a maximum limit for that contaminant, increasing testing frequency, review of labelling requirements, provision of advice to a specific population subgroup, issuing a product recall and/or a ban on imports of the product in question). Furthermore, the process of conducting a risk analysis enables authorities to identify the various points of control along the food chain at which measures could be applied, to weigh up the costs and benefits of these different options, and to determine the most effective one(s). As such, it offers a framework to consider the likely impact of the possible measures (including on particular groups such as a food industry subsector) and contributes towards enhanced utilization of public resources by focusing on the highest food safety risks. Risk analysis is comprised of three components: risk management, risk assessment and risk communication (Figure 6.1). Each of these components has been applied

FIGURE 6.1
The FAO/WHO risk analysis framework



Source: FAO/WHO, 2009.

worldwide for a long time and this chapter provides important information compiled during a risk profiling exercise and a preliminary risk assessment with the available data on microplastics including certain monomers and associated plastic contaminants and additives in fishery and aquaculture products. These preliminary data will be useful for a future risk assessment.

Whereas plastic contamination is not a new phenomenon, characterizing the risks posed by micro- and nanoplastics is an emerging issue. Risk is typically defined as the product of the hazard posed by an agent multiplied by the exposure incurred. With respect to small-sized plastics existing hazards are listed in Figure 6.2.

FIGURE 6.2

Hazards resulting from microplastic pollution. Numbers in squares refer to hazards outlined in the text to the left

1. Physical characteristics of microplastic particles (e.g., size, shape, surface area, etc.) can pose hazards.

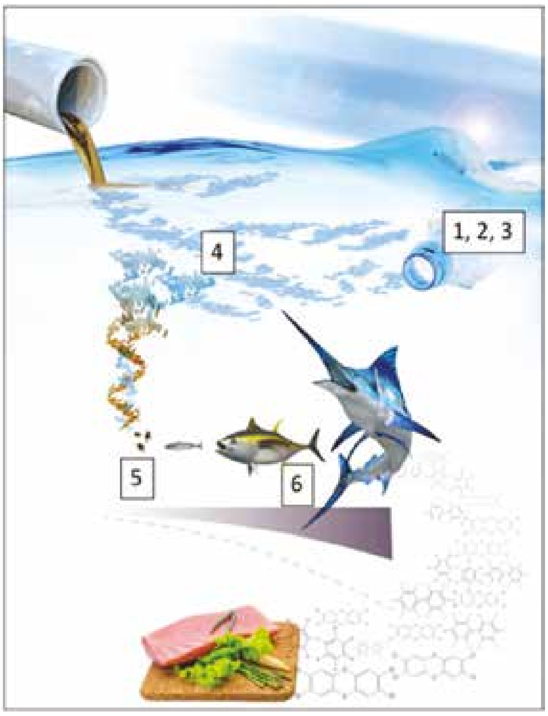
2. Physical characteristics of nanoplastic particles (e.g., size, shape, surface area, etc.) can pose hazards.

3. Additives and coatings intentionally placed on and within the plastic are potential hazards.

4. Unintentional hazards may result from the ability of polymers to selectively attract and concentrate environmental agents, including harmful hydrophobic chemicals, and toxicogenic and pathogenic microorganisms.

5. Environmental durability of plastics creates opportunities of repeated uptake and subsequent release of environmental pollutants into aquatic organisms.

6. Contaminant concentrations can increase through the food chain: from lower to higher trophic levels through predation.



Source: Image Courtesy of Halden, Caspermeyer and Northrop, Biodesign Institute, Arizona State University.

6.2 IDENTIFICATION OF HUMAN HAZARDS FROM MICRO- AND NANOPLASTIC INGESTION

6.2.1 Ingestion, translocation, absorption and excretion of microplastics

Upon ingestion of contaminated foods, micro- and nanoplastics will enter the gastrointestinal (GI) tract. There is a lack of information on the fate of micro- and nanoplastics in the human GI tract. An important question is whether after ingestion micro- and nanoplastics are confined to the gut lumen or whether translocation across the gut epithelium and thus absorption takes place. Translocation would imply exposure of internal organs and tissues to these particles. It is not known whether nanoplastics can be formed upon degradation of microplastics in the GI tract. However, nanoplastic aggregates, when present, may de-aggregate in the gut, thus enhancing absorption. The data available only include translocation and distribution (see below), whereas no information is available on metabolism and little on excretion.

Translocation, absorption and distribution of microplastics

The epithelium of the gut wall represents an important barrier to microplastics, excluding direct transcellular transport. The paracellular route (i.e. transport through space between cells) of uptake is also not possible, given that the maximal functional pore size of the connecting tight junction channels is only about 1.5 nm (Alberts *et al.*, 2002). However, uptake via lymphatic tissue might be possible, specifically via the microfold (M) cells in the Peyer's patches[5] (Galloway, 2015), after which phagocytosis may occur, or via endocytosis[6]. Specific data for microplastics are limited. Particle size is one of the most important factors in determining the extent and pathway of uptake. The upper particle size limit for endocytosis is about 0.5 µm (Yoo, Doshi and Mitragotri, 2011). Phagocytosis by macrophages is believed to occur with particles > 0.5 µm (Yoo, Doshi and Mitragotri, 2011). The upper size limit for phagocytosis obviously is dictated by the volume of the macrophage.

Phagocytosis of 1 µm, 5 µm and 12 µm polymethylacrylate (PMMA) and polystyrene (PS) particles was demonstrated in peritoneal macrophages after intraperitoneal injection in mice (Tomazic-Jezic, Merritt and Umbreit, 2001). Probably the Peyer's patches rich in M-cells are the predominant sites of absorption of microplastics (Galloway, 2015; Wright and Kelly, 2017).

Translocation across the mammalian gut into the lymphatic system of various types and sizes of microparticles of various composition, ranging from 0.1 µm to 150 µm, has been demonstrated in humans (size of particles: from 0.2 µm to 150 µm), dogs (from 3 µm to 100 µm), rabbits (from 0.1 µm to 10 µm) and rodents (from 30 µm to 40 µm) (reviewed by Hussain, Jaitley and Florence, 2001). Polyvinyl chloride (PVC) particles (from 5 µm to 110 µm) have been detected in the portal vein of dogs (Volkheimer, 1975).

The intestinal absorption of microplastics appears to be small. In various rodents, absorption of latex particles measuring 2 µm across was only between 0.04 percent and 0.3 percent of the administered dose (Carr *et al.*, 2012). Similar limited absorption (about 0.2 percent) of polylactide-co-glycolide microparticles (3 µm) was measured *in vitro* using human mucosal colon tissue mounted in an Ussing chamber[7] (Schmidt *et al.*, 2013). The mucosal colon tissue of patients with inflammatory bowel disease showed increased transport (0.45 percent as compared to 0.2 percent in healthy controls) of these particles due to increased permeability of the gut (Schmidt *et al.*, 2013).

Not much is known on the distribution of microplastics after absorption, but it is known that microparticles > 0.2 µm that appear in lymph will be eliminated through the splenic filtration system into the gut (Yoo, Doshi and Mitragotri, 2011), whereas microparticles in the blood will be removed in the liver by bile, and finally excreted via faeces. In mice, daily oral gavage of 5 µm and 20 µm fluorescent polystyrene microplastic particles resulted in accumulation of both types of particles in the liver and kidney (Deng *et al.*, 2017). In these experiments very high doses of the particles were used: from 2×10^4 to 150×10^4 items per animal per day. These results for mice challenge earlier work that concluded that particles measuring > 1.5 µm were not expected to enter the capillaries of organs (Yoo, Doshi and Mitragotri, 2011).

Considering the many factors affecting absorption, such as size, composition, surface charge and hydrophilicity, it is difficult to predict the uptake of the particles.

Summarizing (Table 6.1), *in vivo* human data on the absorption of microplastics are scarce or not available. Mammalian studies have detected microparticles with sizes up to 150 µm in lymph, whereas one study detected PVC particles (110 µm) in the portal vein. It is very likely that microplastics >150 µm are not absorbed. Consequently, only

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[5] Peyer´s patches are lymphoid follicles found in the lower small intestine and represent an important part of the gut associated lymphoid tissue. Their role is to identify and destroy foreign particles in the intestinal lumen.

[6] Endocytosis is a general term used to describe the process by which cells engulf and incorporate liquids (pinocytosis) or solids (phagocytosis). In higher animals these processes play an important role in the immune system.

[7] An Ussing chamber is an apparatus for measuring epithelial membrane properties. It can detect and quantify transport and barrier functions of living tissue.

local effects in the gut might be expected, viz. immune and inflammatory responses. Smaller microplastics (<150 µm) may lead to systemic exposure, but available data show that absorption was limited (≤ 0.3 percent). Only the smallest fraction (size ≤ 20 µm) may penetrate into organs.

Translocation, absorption and distribution of nanoplastics

In contrast to microplastics, the epithelium of the gut wall does not form an impenetrable barrier to nanoparticles. Therefore, a special concern regarding nanoparticles is their ability to translocate across the gut epithelium, resulting in systemic exposure. However, most of the uptake data have been obtained with a large variety of nanoparticles, and not specifically with nanoplastics. Only polystyrene (PS) nanoparticles have been used as model particles in mammalian *in vivo* and in *in vitro* studies. *In vivo* human data on the absorption of nanoplastics are not available.

The estimated oral bioavailability of 50 nm PS nanoparticles varied between studies from 0.2 percent to 2 percent (Walczak *et al.*, 2015) to 7 percent (Jani *et al.*, 1990). As with microplastics, there does not seem to be a simple relation between uptake, size and composition of the nanoparticles (Jani *et al.*, 1990, Jani, McCarthy and Florence, 1992; Hillery, Jani and Florence, 1994; Hillery and Florence, 1996; Hussain, Jani and Florence, 1997; Hussain, Jaitley and Florence, 2001; Walczak *et al.*, 2015). Highly variable uptakes of PS nanoparticles (from 50 nm to 500 nm) have been reported in various *in vitro* intestinal models ranging from 1.5 percent to 10 percent. The extent of absorption depended on particle size, surface chemistry and type of *in vitro* model (des Rieux *et al.*, 2007; Kulkarni and Feng, 2013; Walczak *et al.*, 2015). In a direct comparison of the movement of engineered carboxylated polystyrene nanoplastics measuring 50 nm and 200 nm across a coculture of Caco-2 (enterocyte-like), HT29-MTX (goblet cell-like) and Raji B (M cell-like) cells, it was found that the transport of 50 nm particles was about two orders of magnitude faster than that of 200 nm particles (Mahler *et al.*, 2012). Furthermore, transport of 200 nm PS particles was dependent on temperature and greatly dependent on the presence of M cells, while movement of 50 nm PS particles was independent of these variables (Mahler *et al.*, 2012).

Once nanoparticles have been absorbed, whole body distribution has been shown. For example, after intravenous injection of various sized gold nanoparticles (from 10 nm to 250 nm) in rats, the smallest particles appeared to be widespread and were found in the liver, spleen, heart, lungs, thymus, reproductive organs, kidney, and even in the brain (i.e. crossed the blood–brain barrier). The largest particles were mainly found in the liver and spleen (De Jong *et al.*, 2008). Some nanoparticles were capable to cross biological barriers and potentially access, for example, the brain, the testes and the foetus. Using an *ex vivo* human placental perfusion model, fluorescent PS particles with diameters from 50 nm to 240 nm were found to be taken up by the placenta in a size-dependent manner (i.e. the transplacental transfer was greater for smaller particles) (Wick *et al.*, 2010). Particles sized 500 nm were mainly retained in the maternal circulation or placental tissue, with a low concentration detected in the foetal circulation.

Summarizing (Table 6.1), translocation across the epithelium has been demonstrated for many types of nanoparticles which may result in access to many organs such as the liver, spleen, heart, lungs, thymus, reproductive organs, kidney and the brain. In addition to the blood-brain barrier, the placental barrier may also be crossed. Nanoplastics, other than polystyrene particles, have not been studied yet, and *in vivo* human data on the absorption of nanoplastics are not available (Bouwmeester, Hollman and Peters, 2015; EFSA, 2016). It should be realized that uptake and distribution of nanoparticles very much depend on the chemical nature of the material along with size, shape and other physicochemical properties (EFSA, 2009). Thus, extrapolations from studies on one kind of nanomaterial should be made with caution.

TABLE 6.1
Fate of microplastic and nanoplastics in mammalian bodies as a function of particle size

Microplastics (0.1–5000 μm)		**Nanoplastics (1–100 nm)**
> 150 μm	no absorption	
< 150 μm	in lymph absorption ≤ 0.3%	
= 110 μm	in portal vein	
≤ 20 μm (≤20000 nm)	access into organs	
		≤ 100 nm access to all organs, translocation of blood-brain and placental barrier
		Absorption up to 7%

6.2.2 Toxicity of micro- and nanoplastic particles

In general, after oral ingestion the largest fraction (> 90 percent) of the ingested micro- and nanoplastics will be excreted via faeces (EFSA, 2016). As described in Section 6.2.1, only plastic particles smaller than 150 μm (by definition the smallest microplastics and all nanoplastics) may translocate across the gut epithelium, causing systemic exposure.

It may be expected that micro- and nanoplastic particles will most likely interact with the immune system, not in the least because they can be taken up by phagocytic cells. Experiments in rodents on intraperitoneally injected or inhaled microplastic and nanoplastic beads collectively showed that they activate T-cells[8] and are phagocytized by macrophages, which traffic the particles to the lymph nodes (e.g. Tomazic-Jezic, Merritt and Umbreit, 2001; Blank *et al.*, 2013).). A mouse study with very high doses of 5 and 20 μm polystyrene particles (from 2 x 10^4 to 150 x 10^4 items per animal per day) observed liver inflammation, and changes in metabolic profiles suggesting effects on energy and lipid metabolism, and oxidative stress (Deng *et al.*, 2017). Some of the aforementioned effects have been corroborated by *in vitro* studies (Seydoux *et al.*, 2014) and were more pronounced with smaller plastic beads and differed between various polymers. Interactions of micro- and nanoplastics with the immune system may potentially lead to immunotoxicity and might be associated with several adverse outcomes: 1) immunosuppression (decreased host resistance to infectious agents and tumours); 2) immune activation (increased risk of developing allergic and autoimmune diseases); and 3) abnormal inflammatory responses (chronic inflammation, tissue or organ damage and dysfunction) (EFSA, 2016). However, such effects have so far not been reported in humans. Furthermore, it may be expected that diseases related to the GI tract could potentially be worsened, since most of the particles will be deposited in the GI tract and may interact with bioprocesses at that site (Powell, Thoree and Pele, 2007; Handy *et al.*, 2008), including those in microbiota.

The medical literature (listed in Table 6.2) related to air quality and surgical plastic materials contains information on the impact of micro- and nanoplastics in humans. The inhalation of small particles of plastic may be relevant to the ingestion of plastics (Wright and Kelly, 2017), because the same type of epithelial tissue is involved. Micro- and nanoplastics released from surgical materials may mimic effects of absorbed particles that have entered the bloodstream and tissues. The medical literature documents impacts from polyethylene (PE), PS, Poly(methyl methacrylate) (PMMA) and polycarbonate (PC) particles ranging from 20 nm to 55 μm in size (Table 6.2). Effects are demonstrated at several levels of suborganismal biological organization, from oxidative stress at the cellular level to lesions in organs. When researchers

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[8] T-cells are a type of lymphocyte (i.e. white blood cell) that play an important role in cell-mediated immunity.

TABLE 6.2

Medical literature on impact of microplastics and nanoplastics originating from inhalation and surgical materials at various levels of biological organization

Level of biological organization	Particle type and size	Effect	Reference
Macromolecules	PE 100 nm–30 µm PS 50 nm–4.7 µm PMMA 1 µm–2 µm PC 1 µm–55 µm	DNA damage, changes in gene and protein expression	Gelb *et al.*, 1994; Brown *et al.*, 2001; DeHeer *et al.*, 2001; Gretzer *et al.*, 2002; Petit *et al.*, 2002; Ingram *et al.*, 2004; Clohisy *et al.*, 2006; Kaufman *et al.*, 2008; Markel *et al.*, 2009; Huang *et al.*, 2010; Hallab *et al.*, 2012; McGuinness *et al.*, 2011; Samuelsen *et al.*, 2009; Smith and Hallab 2010; Pearl *et al.*, 2011
Organelles*	PMMA 10 µm	more micronuclei	Zhang *et al.*, 2008
Cells	PS 20 nm–4.7 µm PE 300 nm–10 µm PMMA 2 µm–35 µm PS 20 nm–200 nm PS 60 nm–200 nm	cell clotting, necrosis, apoptosis, proliferation and loss of cell viability Oxidative stress Increased Ca ions	Gelb *et al.*, 1994; Brown *et al.*, 2001; Gretzer *et al.*, 2002; Bernard *et al.*, 2007; Fröhlich *et al.*, 2009; Samuelsen *et al.*, 2009; Hallab *et al.*, 2012; McGuinness *et al.*, 2011
Tissues	PE 600 nm–21 µ, PMMA 1 µm–35 µm	inflammation and bone osteolysis	Gelb *et al.*, 1994; Clohisy *et al.*, 2006; Markel *et al.*, 2009; Pearl *et al.*, 2011
Organs	PMMA 1 µm–10 µm	lesions	Zhang *et al.*, 2008; Pearl *et al.*, 2011

*An organelle is a specialized subunit within a cell (e.g. mitochondria) with a specific function.
PE (Polyethylene), PS (Polystyrene), PMMA (Poly(methyl methacrylate)), PC (Polycarbonate).

investigated impacts of inhalation, they observed oxidative stress, changes in calcium ion concentrations, changes in gene and protein expression, proliferation of cells and cellular necrosis. Studies with plastic prostheses observed changes in expression of genes and proteins, damage to DNA, compromised immune cells, cellular proliferation and necrosis, inflammation in tissues, osteolysis, and lacerations of tissue. In addition, *in vitro* studies that exposed human cells to these types of micro- and nano- plastics have found impacts related to oxidative stress, changes in protein expression, damage to DNA, cellular apoptosis (i.e. programmed cell death) and necrosis, and cell clotting.

In contrast to nanoplastics, the toxicity of engineered nanomaterials, such as metal and metal oxide particles, have been more widely studied and various toxic effects have been found, (reviewed by Bouwmeester *et al.*, 2009). However, comparisons between nanoplastics and engineered nanomaterials are very tricky owing to the roles played by the chemical nature of the material along with size, shape, surface chemistry and charge (reviewed by Bouwmeester *et al.*, 2009). These characteristics differ substantially between these types of particles. Examples of toxic responses of engineered metal/metal-oxide nanomaterials are the production of reactive oxygen species (ROS) and associated inflammation, liver and kidney damage, secondary genotoxic effects and immune effects, which result from the release of metal ions (dissolution) and are therefore not applicable to nanoplastics.

In conclusion, because of a general lack of experimental data, the risk of toxicity of micro- and nanoplastic particles after oral uptake in humans cannot be evaluated.

6.2.3 Contaminants and additives associated with microplastics in seafood

Plastic additives

Toxicity

Toxicity of major plastic additives has been extensively studied, and there is some concern on disruption of human endocrine function by phthalates, bisphenol A (BPA), and polybrominated bisphenols, such as Tetrabromobisphenol (TBBPA) (reviewed in Meeker, Sathyanarayana and Swan, 2009; Talsness *et al*., 2009; Halden, 2010). Data on toxicity has already been briefly described in section 3.4. Besides the above mentioned plastic additives, a diversity of chemical classes of endocrine disrupting chemicals, including persistent organic pollutants are identified in a United Nations Environment Programme (UNEP) and World Health Organization (WHO) report (UNEP/WHO, 2013). UNEP/WHO (2013) concluded that there is a high incidence of many endocrine-related disorders in humans, whereas increasing trends are noticeable. Although chemicals with endocrine disrupting properties affected disease outcomes in animal models, the extent to which observations in these experimental models are relevant to humans is still unclear (UNEP/WHO, 2013).

Exposure from the whole diet

Total dietary exposure to flame retardants in Europe was studied by the European Food Safety Authority (EFSA) (EFSA, 2011). For average consumers the intake of Polybrominated diphenyl ethers (PBDEs) (BDE-47, -209, -153, and -154) ranged, in nanograms (ng) per kg of body weight (b.w.) per day, from 0.70 ng/kg to 5.7 ng/kg b.w. per day. The Joint FAO/WHO Expert Committee on Food Additives (JECFA) reported a dietary intake of total PDBE congeners of approximately 4 ng/kg b.w. per day for adults and up to 100 ng/kg b.w. per day for breasfed infants. JECFA recognized the preliminary nature of the data on PDBE concentrations in food and human milk which added considerable uncertainty to the dietary intake estimates (WHO, 2006).

EFSA measured the total dietary intake of bisphenol A (BPA) and found an average of 130 ng/kg b.w. per day. A temporary Tolerable Daily Intake (TDI)[9] of 4 µg BPA/kg b.w. per day was proposed (EFSA, 2015a). Data published by FAO/WHO showed that the exposure of adults to BPA was from less than 0.01 to 0.4 µg/kg b.w. per day at the mean and 0.06–1.5 µg/kg b.w. per day at the 95th/97.5th percentile. For young children and teenagers, mean exposure was 0.1–0.5 µg/kg b.w. per day, and exposure at the 95th/97.5th percentile was 0.3–1.1 µg/kg b.w. per day (WHO, 2011).

Plastic monomers and polymers

Toxicity

The extent to which plastics tend to disintegrate into subunits is a function of multiple factors, including the type and age of the plastic polymer as well as the environmental conditions of use and weathering, such as temperature, irradiation, pH, etc. (Akbay and Özdemir, 2016; Yang *et al*., 2011). Release of monomers and oligomers from the most widely used plastics and their toxicity have already been described in section 3.4.

Exposure from the whole diet

Data for BPA have been given in the above section on plastic additives. For other monomers and polymers there are no data available on exposure from the diet.

Persistent, bioaccumulative and toxic compounds (PBTs)

Toxicity

PBTs have been reported to cause a variety of effects including immunologic, teratogenic, reproductive, carcinogenic and neurological effects (Kodavanti *et al*., 1998), and extensive programs have been set up throughout the world to monitor their presence in the environment and in foods.

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[9] A Tolerable Daily Intake (TDI) is an estimate of the amount of a substance in air, food or drinking water that can be taken in daily over a lifetime without appreciable health risk.

Exposure from the whole diet

Trophic transfer of these persistent pollutants within the marine food webs is well-documented (see section 4.3.1, and reviewed in EFSA, 2016), and factors have been studied that determine the extent of trophic transfer (Wan *et al.*, 2005).

EFSA monitored 6 indicators for non-dioxin like Polychlorinated biphenyls (PCBs) in food available on the European market (EFSA, 2012), and estimated an average total dietary exposure between 4.3 ng PCBs/kg and 25.7 ng PCBs/kg b.w. per day; the 95 percentile was between 7.8 ng/kg and 53.7 ng/kg b.w. per day. Fish, meat and dairy products appeared to be the highest contributing food groups to the dietary exposure of PCBs. Data of JECFA on the total dietary intake of these indicator non-dioxin like PCBs were similar, and ranged from 1 ng to 83 ng PCBs/kg b.w. per day (WHO, 2016).

The median dietary exposure of Polyaromatic hydrocarbons (PAHs) across European countries was calculated by EFSA both for average and high dietary consumers. For average European Union consumers a median exposure of 28.8 ng/kg b.w. per day was found (EFSA, 2008). The two highest contributors to the dietary exposure were cereals and seafood. JECFA estimated the mean intake of benzo[a]pyrene (as a marker for PAHs) to be 4 ng/kg b.w. per day, whereas the high intake was estimated at 10 ng/kg b.w. per day (WHO, 2006).

The mean total dietary intake of Dichlorodiphenyltrichloroethane (DDT) and related compounds (DDE and DDD) in Europe ranged from 5 ng/kg b.w. per day to 30 ng/kg b.w. per day. Food of animal origin was the major source of DDT exposure (EFSA, 2006).

Pathogens

Although it has been documented that plastic debris can act as a substrate for diverse microbial communities (see sections 3.4 and 5.6.5), sufficient data on the occurrence of pathogens on microplastics are lacking to include pathogens in the risk profiling.

6.3 CASE STUDY: BIVALVE CONSUMPTION

6.3.1 Rationale for species selection

Seafood is one major source of dietary microplastics to humans. Although finfish is the predominant seafood, it is likely not a large source of microplastics, because in most cases the gut is not consumed (see section 5.5). Similarly, peeling of shrimps removes the gastrointestinal tract together with the microplastics (see section 5.4). For Norway lobster (*Nephrops norvegicus*) very little quantitative information on microplastics exists, and in general the gastrointestinal tract is also removed before consumption. Thus, bivalves are likely the largest source of microplastics from seafood to humans because they are consumed whole. As data on occurrence of microplastics were only available for mussels to some extent, mussels were selected as a case study.

6.3.2 Exposure assessment

Human intake of microplastics from seafood has been estimated from 1 particle per day (Vandermeersch *et al.*, 2015) to 30 particles per day (Van Cauwenberghe and Janssen, 2014) depending on seafood consumption habits. A worst case exposure scenario was based on a portion of 225 g of mussels eaten by French volunteers (EFSA, 2016). According to Table 5.1, Chinese bivalves contained the highest number of microplastics: median value of 4 particles/g of tissue (Li *et al.*, 2015). Thus, consumption of such a portion of mussels (225 g) would lead to ingestion of about 900 microplastic particles. Assuming spherical particles with an average particle size diameter of 25 µm (Van Cauwenberghe and Janssen, 2014) and a density of 0.92 g/cm^3 (density of low density polyethylene (LDPE), the most common polymer type of microplastics (Bouwmeester, Hollman and Peters, 2015)), these 900 plastic particles would represent 7 µg of plastics, or 0.1 µg/kg b.w., assuming an average human individual of 70 kg. In the following, the highest reported contents of contaminants and additives in microplastics were used (see section 5.6) to calculate exposure via microplastic ingestion. Furthermore, the total dietary intake of these chemicals has also been described in these sections. Based on these assumptions and data, the contribution of microplastics in mussels to the total dietary exposure to contaminants and additives was estimated (Table 6.3).

As a result, it can be concluded that microplastics contribute only a very small fraction of the total dietary intake of contaminants: PAHs from 0.02 percent to 0.1 percent, PCBs from 0.007 percent to 0.03 percent, and DDT from 0.0000002 percent to 0.004 percent (Table 6.3). The contribution of microplastics to the dietary intake of additives is even smaller: BPA from 0.000005 percent to 0.00002 percent, and PBDEs from 0.0007 percent to 0.003 percent (Table 6.3).

TABLE 6.3

Comparison of the calculated intake of contaminants and additives (worst case scenario) directly from microplastics in seafood and the total dietary intake of these compounds

Compound	Highest concentration in microplastics (see section 5.6) (ng/g)	Calculated intake from microplastics (pg/kg bw/day)	Total intake from the diet (pg/kg bw/day)	Ratio intake microplastic/total dietary intake (%)
Contaminants				
Non-dioxin like PCBs	2 970	0.3		
EFSA, 2012			4 300[a]	0.007
JECFA, 2016			1 000[a]	0.03
PAHs	44 800	4.5		
EFSA, 2008			28 800[b]	0.02
JECFA, 2006			4 000[c]	0.1
DDT	2 100	0.2		
EFSA, 2006			5 000[d]	0.004
JECFA,1960			100 000 000[j]	0.0000002
Additives/monomers				
Bisphenol A	200	0.02		
EFSA, 2015a			130 000[e]	0.00002
FAO/WHO, 2011			400 000[f]	0.000005
PBDEs	50	0.005		
EFSA, 2011			700[g]	0.0007
JECFA, 2006			185[h]	0.003
NP	2 500	0.3	NA[i]	
OP	50	0.005	NA[i]	

[a] lowest intake of 6 indicators of non-dioxin like PCBs, representing about 50 percent of all non-dioxin like PCBs
[b] median intake (EFSA, 2008)
[c] mean intake of benzo[a]pyrene (JECFA)
[d] lowest intake, DDT and related compounds (EFSA, 2006)
[e] average intake adults (EFSA, 2015a)
[f] lowest intake FAO/WHO
[g] lowest intake, sum of BDE-47, -209, -153, -154 (EFSA, 2011)
[h] lowest intake JECFA
[i] NA: dietary intake not available from EFSA or JECFA
[j] provisional tolerable daily intake (JECFA)

Note: PCBs (Polychlorinated biphenyls), PAHs (Polycyclic aromatic hydrocarbons), DDT (Dichlorodiphenyltrichloroethane), PBDEs (Polybrominated diphenyl ethers), NP (Nonylphenol), OP (Octylphenol)

6.3.3 Hazard and risk characterization

Only very recently one peer-reviewed toxicity study of microplastics in mice was published (Deng *et al.*, 2017), whereas toxicity studies on nanoplastics in rodent species are lacking. These types of studies usually form the basis of a risk assessment for humans. As these data are very incomplete or lacking, a formal hazard and risk characterization of the health effects of the plastic particles is not possible. However, concerning contaminants and additives, even if a worst case scenario is followed, the estimated resulting exposure via microplastics in seafood is less than 0.1 percent of the total dietary intake of these compounds (Table 6.3). This low contribution of seafood may even be overestimated considering that the diet is not the only route of exposure. For several of the chemicals considered, also inhalation (e.g., PCBs and PBDEs via indoor dust) and/or dermal uptake (e.g. BPA from thermal paper used in cash registers or credit card terminals) are relevant. De Boer *et al.* (2016) argued that for flame retardants in humans, dust may pose a bigger risk than food. Note that inhaled dust partly consists of polymeric material originating from clothing and textiles, among other sources (Wright and Kelly, 2017).

In these calculations a worst-case assumption was made that when the microplastics are ingested, these chemicals are completely released from the particles. In addition, the highest content of microplastics in bivalves (see section 5.3) and the highest reported contents of contaminants and additives in microplastics (see section 5.6) were used, whereas the lowest or median dietary intakes of these chemicals from the diet were chosen (see section 6.2.3). For additives, a previous EFSA worst case calculation resulted in a higher dietary contribution, because an average content of additives in microplastics of 4 percent (EFSA, 2016) was used. The one presented in this report used actually measured concentrations of PBDEs and BPA in microplastics, leading to lower more realistic ratios. From Table 6.3, it can be concluded that the effect of consumption of microplastics on the intake of contaminants and additives is very likely negligible.

6.4 LIMITATIONS AND FURTHER RESEARCH

The risk profiling of micro- and nanoplastics in seafood made in the previous sections is hampered by a number of limitations.

First of all, data on the occurrence of microplastics in seafood are rather limited and most data originate mainly from European countries. A very important aspect of the occurrence of microplastics in seafood is the particle size distribution. It is evident that particles (>150 µm) are not absorbed and research programs should therefore primarily focus on quantifying microplastics < 150 µm, which certainly is not common practice yet.

Although it is evident that nanoplastics are present and can be formed out of microplastics, no data are available on their occurrence in seafood. Before data on nanoplastics in seafood can be generated, analytical methods for nanoplastics will have to be developed. This will require a focused research program to develop novel methods, because methods for microplastics cannot be adapted to measure nanoplastics in food. Data on nanoplastics are essential, given the fact that nanoplastics are very likely to translocate across the gastrointestinal wall and also will result in a widespread tissue distribution.

Toxicological data on the effects of micro- and nanoplastics as such, are essentially lacking and will be indispensable for food risk safety assessment. The available data on toxicokinetics of micro- and nanoplastics only include absorption and distribution, whereas no information is available on metabolism and little on excretion. There is a lack of knowledge on the local effects of micro- and nanoplastics in the human GI tract, and their effects on microbiota. It is not known whether ingested microplastics can be degraded into nanoplastics.

Also, no data are available on the potential impact cooking and/or processing seafood at high temperature may have on the toxicity of microplastics in seafood products.

Comparison of the calculated intake of contaminants and additives (worst case scenario) directly from microplastics in bivalves, with published data on the total dietary intake of these compounds in human populations, showed that the contribution of microplastics is very likely negligible (<0.1 percent). This estimation is based on information for non-dioxin like PCBs, PAHs, DDT and related compounds, BPA and PBDEs. No sufficient information was available to extend this evaluation to dioxins, Nonylphenol (NP), Octylphenol (OP), phthalates, and other plastic monomers.

The aforementioned limitations in our knowledge define the areas where further research is needed: analytical methods, especially for nanoplastics, occurrence of microplastics (<150 µm) and nanoplastics in seafood, and toxicological research on microplastics (<150 µm) and nanoplastics.

7. Conclusions

7.1 AQUATIC ENVIRONMENTS

Microplastics are currently a ubiquitous contaminant in aquatic (sea, freshwater and estuarine) environments, and have been detected in all compartments: beach sediments, sublittoral and deep sea sediments, water surface and water column. There is a large spatial variability in the distribution of microplastics in aquatic environments and differences amount to several orders of magnitude between studies.

Plastic contamination of aquatic environments has been increasing since industrial high volume polymer production commenced in the 1950s. As production levels continue to increase, it is likely that plastic related contamination will continue to increase in the foreseeable future. As this plastic debris degrades, fragmentation will continue to shift the particle distribution from large plastics to smaller micro- and nanoplastics in aquatic environments, even in the absence of new inputs. Thus, the problem of micro-nanoplastic pollution will only increase in future years.

The entry rate of primary microplastics and the production rate of secondary microplastics in the marine environment have not been quantified adequately on a global or local scale. Very little information exists on the relative contribution of different types of plastic and associated chemicals, and how this has varied with time. This limits the degree to which projections can be made of future trends of the potential effects of microplastics on fisheries and aquaculture.

The distribution of floating microplastics in surface waters shows considerable variation at all spatial scales. Such variations can be explained by ocean circulation patterns and mixing process, combined with the geographical and temporal variations in probable input rates (derived from proxy data such as coastal population densities and main shipping routes). The relative distribution of floating microplastics has been modelled on a global scale based on well-constrained ocean circulation models and input assumptions, but the model outputs have only been partially validated using observational data.

Very little information exists on the distribution of microplastics in the water column, where many species of fish feed (i.e. mesopelagic fish). Plastic particles that were originally buoyant, may have been incorporated with other denser material, such as faecal pellets, or developed biofilms that increased particle density.

Many types of plastic are denser than seawater and will sink once any initial buoyancy is lost (e.g. fishing gears), and there is evidence that microplastics settle in bottom sediments, from the littoral zone down to deep ocean depths. This may be relevant for many species of fish and shellfish which either live in or feed at the sediment surface. In general, sediments show higher concentrations of microplastics than water samples. However, data collection is problematic, especially in deeper waters. More data are available for the littoral zone, which has revealed the presence of relatively large quantities of plastic fibres near urban centres, for example. Overall there is very limited information on the distribution of microplastics in sediments.

The Fisheries and aquaculture sector contributes to microplastic contamination but its overall quantitative contribution is as yet poorly understood. Abandoned, lost or otherwise discarded fishing gears (ALDFG) are thought to be the main contributor of this sector to the generation of microplastics in aquatic environments.

7.2 AQUATIC ORGANISMS

Microplastic ingestion by aquatic organisms has been confirmed in laboratory and field work, including in commercial species. In aquaculture only bivalves have been surveyed. The number of microplastics observed in the gut contents of farmed and wild aquatic animals, including commercial fish species, is usually low. However, observations are not yet comprehensive in terms of the range of species of interest and their geographical distribution.

Trophic transfer of microplastics has been observed under laboratory conditions, but at present there is no direct evidence of trophic transfer in natural conditions. It is not likely that microplastics are accumulated through trophic transfer because the majority of the microplastics will not translocate into the tissues of their hosts.

A critical evaluation of the scientific literature regarding transfer of Persistent Bioaccumulative and Toxic compounds (PBTs) from microplastics to marine animals, including model calculations based on experimental studies, concluded that microplastic ingestion by marine organisms is not likely to increase exposure to PBTs in these organisms. Similarly, transfer of additives from ingested microplastics appears to have a negligible effect on the total exposure to additives as was demonstrated for Nonylphenol (NP) and bisphenol A (BPA).

Metabolic and physiological negative responses have only been observed under laboratory conditions, where in most cases very high levels of microplastics were tested under exposure scenarios that were not representative of natural environmental conditions.

7.3 SEAFOOD SAFETY

The amounts of microplastics ingested by humans as a result of consuming seafood are poorly quantified. This is in contrast to well established practices to estimate the intake of a wide variety of contaminants by the human population from seafood consumption.

The digestive tract of marine organisms contains the largest quantities of microplastics. However, this part is normally discarded before consumption, except for most bivalves, some echinoderms and several small species of fish which are eaten whole. As an example, a worst case estimate of exposure to microplastics after consumption of a portion of mussels (225 g) would be 7 µg of plastics. Based on this estimate and considering the highest concentrations of additives or contaminants reported in microplastics, and assuming complete release from microplastics, the microplastics will have a negligible effect on the total dietary exposure to PBTs and plastic additives. These contaminants are estimated to contribute only <0.1 percent of the total dietary exposure to these compounds.

As far as it is known, only microplastics smaller than 150 µm may translocate across the mammalian gut epithelium causing systemic exposure. However, the absorption of these microplastics is expected to be limited (≤ 0.3 percent). Only the smallest microplastic fraction (size ≤ 20 µm) may penetrate into organs and cause systemic exposure. Most likely, microplastics will interact with the immune system but reports in humans are lacking. Adverse effects in humans have only been reported after inhalation and by plastic prostheses. Therefore, research should focus primarily on the smaller (< 150 µm) microplastic fraction.

Nanoplastics can be produced during fragmentation of microplastics and can originate from particles used in industrial processes. However, nothing is known on the occurrence of nanoplastics in aquatic environments and in fishery and aquaculture products, although their presence can be speculated from knowledge of industrial uses, laboratory experiments on degradation, and hydrocarbon-degrading microorganisms.

Nanoparticles can translocate across the gut epithelium resulting in systemic exposure, and a very wide distribution in all organs is likely. However, specific data for nanoplastics are not available. Toxicological data for nanoplastics are lacking, even though some adverse effects in humans have been reported after inhalation and by plastic prostheses. Finally, it is likely that ingested nanoplastics will interact with the immune system.

Although nanoplastics may pose significant concerns, the data that are needed to perform a full food safety risk assessment of nanoplastics in seafood are completely lacking.

Whereas the overall human health risks posed by microplastics in seafood at present appear to be low, it is important to consider the unavoidable increase of micro- and nanoplastics in the future as a result of degradation of plastics already released in the environment as well as future inputs.

8. Policy-relevant recommendations

The following recommendations are presented for the consideration of stakeholders, including policy makers, decision makers, government authorities, fisheries, aquaculture, and seafood industry and retailers, civil society and private sector organizations, consumers, academics and researchers, as concerned with, and interested in the occurrence and impacts of microplastics on fisheries and aquaculture resources, associated seafood safety and human health aspects:

1. Recognize that occurrence and potential impacts of microplastics in fisheries and aquaculture deserve to be studied.
2. Consider applying environmental risk assessment approaches to potential microplastic contamination impacts on fisheries resources and aquaculture operations.
3. Recognize potential impacts of microplastics on seafood quality and safety.
4. Recognize that data and knowledge gaps exist for risks of small microplastics (less than 150 µm) and nanoplastics in seafood, and consider applying risk analysis approaches.
5. Adopt food safety risk analysis frameworks (i) to evaluate hazards and risks to consumers of seafood contaminated with microplastics and (ii) to determine decisions and measures to ensure effective consumer protection and viable seafood trade.
6. Facilitate and promote cost-effective and well-targeted monitoring of microplastics in the environment, biota and seafood products (market sampling) and promote capacity building and implementation of best practices in monitoring and the review of (i) microplastics contamination of seafood and (ii) microplastics contamination effects on fish resources.
7. Develop and promote the most appropriate approaches for monitoring microplastic contamination levels in commercial fish resources and the impacts of microplastic contamination on fish and fisheries products.
8. Strengthen and harmonize analytical methods used for detection and quantification of microplastics and nanoplastics, and ensure appropriate risk-based interpretation of results.
9. Communicate actively and adequately about hazards and risk management of microplastic contamination of seafood to the general public, food safety and consumer protection authorities, fisheries and aquaculture agencies as well as to the seafood industry.
10. Promote cooperation between national and regional authorities, industry and stakeholders concerned with effects of microplastic and nanoplastic contamination and pollution impacts in fisheries, aquaculture and seafood supply chains.
11. Raise awareness of the appropriate authorities (central, regional and municipal), sectors (industry, transport, etc.) and consumers of microplastic issues (impacts, sources and mitigation), and links to industry practices and other sources.
12. Recognize responsibilities of contributors to microplastic contamination including fisheries and aquaculture as well as other sources such as industry, sewage, transport, etc.

13. Raise awareness of fisheries and aquaculture sector stakeholders on the importance of managing plastic-based gears, equipment use and inputs and preventing loss or release of microplastic-generating gears.
14. With a view to reducing and avoiding impacts on fisheries, aquaculture and seafood supplies and consumers, other sectors contributing to microplastic pollution should consider addressing the following selection of recommendations (UNEP, 2016) that were developed for the 2016 United Nations Environment Assembly (UNEA-2), such as:
    - i. While prevention is key, improving waste collection and management is the most urgent short-term solution to reducing plastic inputs, especially in developing economies.
    - ii. Long-term solutions include improved governance at all levels as well as behavioural and system changes, such as a more circular economy and more sustainable production and consumption patterns.
    - iii. Review existing regulatory frameworks, institutional arrangements and other instruments related to marine litter and their enforcement to identify synergies and gaps as well as potential solutions to address gaps globally and regionally.
    - iv. Strengthen and increase cooperation at all levels, including international multi-stakeholder initiatives such as the Global Partnership on Marine Litter.
    - v. Quantify the relative contributions of all critical land-based and sea-based sources and investigate pathways of marine litter, including macrolitter and microliter.
    - vi. Prioritize actions for marine litter mitigation, including through the identification of hotspots and the examination of future scenarios, by the use of best available technologies (e.g., models and simulations).
    - vii. Develop cost-effective monitoring and assessment strategies with regard to marine litter at all levels, taking into account existing programmes, especially at the regional level.
    - viii. Using the precautionary principle and taking into account that there is unequivocal and quantified evidence of the degree of impact of marine plastic debris, reduce marine litter sources through measures such as market-based instruments and regulatory frameworks.
    - ix. Support efforts to promote a life cycle approach to plastic products, including the consideration of the degradation of different polymers and the rate of fragmentation (in the marine environment).
    - x. Strengthen education and awareness measures on marine litter by: (a) Introducing elements into educational curricula at all educational levels; and (b) Providing educational and outreach materials targeted to specific interest groups and range of ages to promote behavioural change.
15. International organizations (such as the Food and Agriculture Organization of the United Nations (FAO), the United Nations Environment Programme (UNEP), the Intergovernmental Oceanographic Commission (IOC), the World Health Organization (WHO), others) as well as regional organizations (regional fisheries organizations, regional seas programmes, others) should collaborate on increasing knowledge and addressing microplastic and nanoplastic contamination of aquatic environments and potential impacts on food safety and fishery and aquaculture resources. Advisory Bodies such as the Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection (GESAMP) and the Joint FAO/WHO Expert Committee on Food Additives (JECFA) should work on specific aspects of contamination of the environment and seafood with microplastics and nanoplastics.

9. Research Needs

Microplastic research is a relatively new and rapidly growing area of scientific interest. There is still much to be done in order to increase knowledge on, for example, the entry rate of primary microplastics and the production rate of secondary microplastics, as well as their fate in aquatic environments, the relative contribution of different plastic compounds and their associated chemicals to microplastic contamination, the relative contribution of the fisheries and aquaculture sector to microplastics contamination, the possible impacts of microplastics at the population and community levels of ecological organization, and the possible impacts on fisheries and aquaculture resources. While recognizing that research needs exist in both realms of ecology and seafood safety, this section lists specific needs with a high emphasis on food safety.

9.1 MICROPLASTICS:

1. Analytical methods for the detection and quantification of microplastics in the environment (water, sediments and biota) and food should be standardized, with a focus on the smaller (less than 150 µm) particles. After this, occurrence data, including particle size, must be generated, to be used for exposure assessment of dietary intake.
2. Toxicological data on microplastics must be generated and the most common polymers need to be considered during this process. The smaller particles (less than 150 µm) are potentially more hazardous and their study should be prioritized.
3. Further data on translocation of microplastics containing the most common polymers should be generated for aquatic organisms and humans; and studies on microplastics as sources of pathogens to fishery and aquaculture products and humans need to be carried out.
4. No data are available on the impact of cooking or processing seafood at high temperature on the potential toxicity of microplastics. Data are required on the resultant physical and chemical changes in microplastics, as well as on the chemical interactions between nutrients and microplastics.

9.2 NANOPLASTICS:

1. Analytical methods for the detection and quantification of nanoplastics in the environment (water, sediments and biota), food, human tissues and blood should be developed and standardized. After this, occurrence data must be generated, which could be used for exposure assessment of dietary intake.
2. Toxicological data on nanoplastics need to be generated where the most common polymers must be considered and the intake data of those should be further developed.
3. Further data on translocation of nanoplastics composed of the most common polymers should be developed for aquatic organisms and humans; and studies on nanoplastics as sources of pathogens to fishery and aquaculture products and humans need to be carried out.
4. No data are available on the impact of cooking or processing seafood at high temperature on the potential toxicity of nanoplastics. Methods should be developed to analyse the resultant physical and chemical changes in nanoplastics, as well as on the chemical interactions between nutrients and nanoplastics.

Annex 1. Sampling of microplastics in aquatic environments and organisms

A1.1 SAMPLING CONSIDERATIONS FOR ABIOTIC ENVIRONMENTS

Before commencing the identification of microplastics in the environment several sampling factors should be taken into account. First, replicability of the study should be considered. For example can replicable results be obtained if samples are collected a week, a month or even a year later? Second, is the method used similar to previous research and will it allow comparisons between research groups in other geographical regions? Third, can spatial and temporal differences in environmental conditions of the survey be accounted for? Fourth, does the method of choice take steps to prevent contamination? Several concerns have been raised regarding the assessment of microplastic distribution due to a vast array of sampling methods utilized as well as inconsistencies in the units reported (Lusher, 2015). Quantification of microplastics in environmental samples is complicated by the size of the oceans, coastlines and sample locations compared to the size of plastics being assessed (Cole *et al*., 2011). Furthermore, microplastic distribution is confounded by patterns in spatial and temporal variability (Doyle *et al*., 2011), which raises the question is the result a true representation of microplastic contamination in the environment or merely a snapshot in time? (See for example, Lima, Costa and Barletta, 2014 and Lima, Barletta and Costa, 2015 for more detail on the impact on seasonality).

There are several matrices in the environment (coastlines, water surface, water column, seafloor and biota) where microplastics could be found, and there are several different methods available to researchers to sample these matrices. There has been some attempt to provide guidelines for sampling (e.g. Hanke *et al*., 2013; Masura *et al*., 2015), although researchers regularly publish “novel” methods. Raising a final question, how comparable are studies utilizing different methods? Results of studies have been reported in different units, e.g. the number of microplastics in a known volume of water (particles per m^3) or an area measurement (particles per km^2). This discrepancy presents a problem when comparing between studies as it is not possible to compare the results directly (Lusher, 2015). Therefore, it is paramount to gain a better understanding of what variables can influence sampling and to implement standardization of methods as far as possible.

Sampling of microplastics in different environments has been approached using a variety of methods: samples can be selective, bulk, or pre-treated to reduce their volume (Hidalgo-Ruz *et al*., 2012). This document will not go into detail as there is ample literature available on methods for extraction of microplastics from water samples (surface and water column), benthic sediment samples and beach samples (e.g. Hanke *et al*., 2013; Hidalgo-Ruz *et al*., 2012; Masura *et al*., 2015; Nuelle *et al*. 2014). In short, selective sampling consists on direct extraction of particles from the environment that are recognizable with the naked eye, such as the collection of pre-production resin pellets from beaches (Fernandino *et al*., 2015). However, selective sampling is only suitable if there is an aim to monitor specific items that are easily recognizable as sampling can be conflicted by the presence of other debris, or lack of distinctive shapes.

Bulk sampling is the collection of an entire volume of sample (water or sediment) without reducing it. This enables a concentration of microplastics based on the sample mass or volume. Bulk samples can be pre-treated to reduce their volume (by way of sieving, density separation or filtering). Most studies use a combination of these steps and follow with a validation step to visually accept particles based on characteristics or through analysis of their molecular structure (e.g. Löder and Gerdts, 2015).

- **Water samples** are generally collected by way of sampling gears (such as Neuston, manta or bongo nets) towed horizontally, vertically or obliquely through the water column or at the water surface for a set period of time. The sampling gears contain a mesh to capture particles. The sample collected in the net will usually be separated into different size fractions by way of sieving. The material can be separated by density, air dried or digested to remove organic material. Remaining particles can be subjected to visual examination or chemical analysis to identify and verify particles of synthetic origin (such as FTIR and Raman spectrometry).
- **Beach samples** usually consist of a given area or volume of sediment being collected (by spade, spoon or with a sediment corer) and sieved to remove large items of debris. The small fraction can be separated by density to remove organic material. The remaining material is visually analysed under a microscope and potential plastics can be analysed chemically to verify their molecular structure.
- **Benthic**, or **subtidal sediments** can be sampled by way of core or grab sampler (e.g. Reineck core or Van Veen grab). This can be conducted in shallow coastal areas (Vianello *et al.*, 2013) to the deep sea (Van Cauwenberge *et al.*, 2013a). The retrieved sample can be processed in much the same way as beach samples.

A1.2 BASIC SAMPLING CONSIDERATIONS FOR BIOTA

Sampling organisms from the wild needs to consider how the capture, handling and processing steps could alter the presence of microplastics in individuals. Benthic invertebrate species such as Norway lobster (*Nephrops norvegicus*) may be collected using grabs, traps, and creels, or by bottom trawling (Murray and Cowie, 2011; Welden and Cowie, 2016a). Fish species are generally recovered from surface, midwater and benthic trawls. Some species can be collected from the wild by hand, this is common practice for bivalves and crustaceans. Another method is direct collection from shellfish or fish farms, or from commercial fish markets, where the capture method is often unknown. Avoiding contamination and biases during sampling and sample analysis is paramount, and mitigation protocols should be followed.

Handling stress, physical movement, and the physiological and behavioural specificities of the sampled organism, may result in the loss of microplastics prior to animal preservation. Gut evacuation times for animals vary between species, therefore some animals might egest microplastic debris prior to analysis. In such cases, the time between sample collection and the preservation of the animal must be as short as possible. Care must also be taken to minimize handling stress or physical damage. This will reduce the potential for microplastic regurgitation. The main cause of regurgitation in fish is thought to be related to the expansion of gas in the swim bladder and the likelihood of regurgitation increases with depth of capture (see Lusher *et al.*, 2017). To further account for this during sampling, if individuals have an empty stomach that shows signs of recent emptying, the organisms should be discounted from analysis to prevent sampling bias through regurgitation on capture. In addition, when organisms are captured in nets, there is a possibility that they ingest microplastics while they are retained in the codend. This is a concern when the mesh size of the net can collect microplastics, for example, in manta nets (e.g. 0.33 mm).

A1.3 AQUATIC ENVIRONMENTS

A1.3.1 Water column

The presence of microplastics in the water column and surface of freshwater and marine water bodies is reported globally. The methods used are often dependent on the equipment available, the target water depth and sampling conditions. Samples can be collected at the water surface (e.g. manta net) or within the water column (e.g. vertically hauled plankton nets). The Marine Strategy Framework Directive (MSFD) of the European Union and the National Oceanographic and Atmospheric Administration (NOAA) of the United States of America recommend the use of neuston tows or manta nets for routine sampling as they can be used to sample a known area of water which can be standardized (m^2 or km^2) to compare between locations. These methods are simple and can be easily conducted from research platforms, however, they are not without imperfections. Sampling is highly dependent on the weather conditions; high winds or currents can impact the volume of water entering the net as well as the position of the net in the water. Surface sampling should only be conducted in calm sea conditions, with minimal tide or waves. Sampling is further limited by mesh size (commonly used 300 µm) and the likelihood of contamination from numerous sources (deck storage, sampler, airborne fibres, etc.). This can result in both underestimation from particles escaping capture as they pass through the net or vertical mixing moving particles away from the sampling area, and overestimation from contamination. The use of smaller mesh sizes can significantly increase the amount of plastics captured (e.g. Song *et al*. 2014). A manta net is preferable for sampling where the target plastics are > 1 mm, as this method can sample a large volume which increases the potential for capture and contamination issues would be controlled by the size range. Other recommended sampling methods include the Continuous Plankton Recorder (Thompson *et al*., 2004), underway pumps (Lusher *et al*., 2014) and epibenthic sledges (Lattin *et al*., 2004) (Table A1.1).

A1.3.2 Beach sediments

There are different tidal zones on beaches: tide line, last high tide (strandline), back beach and intertidal areas. Usually flotsam and large items of litter are deposited within the high tide zones, but there does not appear to be any uniformity to areas of microplastic accumulation. Methods used for beach sampling vary depending on individual aims of surveys. Surveys targeting resin pellets on beaches can rely on selective sampling by eye, but samples assessing microplastic fragments <1 mm require more substantial methods. For example, methods employed include selective sampling: picking particles with tweezers or by hand and using quadrats sampling a known area of surface sediment or volume, and finally sediment cores for the vertical distribution of plastics within sediment. Unfortunately, many of the methods of collection are inconsistent and incomparable. Results are influenced by lower and upper size limits, the sensitivity of the extraction methods and differences in reported units hampers comparison between studies. As with sampling the water column, the choice of method and approach utilized by researchers will determine the reporting units. Studies utilizing beach quadrats generally report a unit of surface area (m^2), however sampling to a specific depth has a unit volume measurement (m^3). Sometimes researchers report values in counts or weights, however, comparing the number of microplastics in a known mass of sediment is confounded by varying water contents of different beach areas (Van Cauwenberge *et al*., 2015). Sandy beaches and shorelines have been the focus of microplastic abundance in sediments. However, the zone sampled can vary among studies. Some will look at a specific location such as the high tide line, or specific littoral zones, whereas other studies will follow a transect perpendicular to the shoreline (Hidalgo-Ruz *et al*., 2012; Van Cauwenberge *et al*., 2015). As there is a lack of uniformity between beach sediment studies, the distribution of microplastics on beaches is not fully understood. There are several different mechanisms that could affect microplastic distribution on beaches including wind and water currents. Once samples are collected they can be sieved to reduce volume and on return to the laboratory density separation is the recommended step to separate microplastics from sediments (see Table A1.2).

A1.3.3 Bottom sediments

Benthic sediments are suggested as a long term sink for microplastics (Van Cauwenberghe *et al.*, 2015b). There are relatively few studies which identify microplastics in benthic sediments but there is evidence of microplastics in the benthos from coastal areas (Carson *et al.*, 2013) to the deep sea (Van Cauwenberge *et al.*, 2013a; Fischer *et al.*, 2015; Woodall *et al.*, 2014). Several different methods can be employed from grabs to corers. Sediment is collected using ship based apparatus and returned to the vessel. Volume measurements taken by cores can allow for the calculation of microplastics in a known volume of sediment. As with beach sediments, density separation to remove microplastics from denser organic material appears to be the most cost effective and appropriate method and is recommended by the MSFD and NOAA.

A1.4 AQUATIC ORGANISMS

A1.4.1 Microplastic extraction

Several techniques have been developed to detect microplastics ingested by biota. Irrespective of the organisms in question, methods for extracting microplastics from biotic material include dissection, depuration, homogenization and digestion of tissues with chemicals or enzymes (Lusher *et al.*, 2017). In a large proportion of studies researchers target specific tissues for the presence of microplastics. These tissues are usually dissected out and visually assessed under a microscope. A review of published literature found that studies focused primarily on the digestive tract (including the stomach and intestine). Localization of microplastics <0.5 mm can be determined by excising organs, such as the digestive tissues, liver or gills, or, where the research question relates to risks of human consumption: edible tissues, such as tail muscles of shrimp (Devriese *et al.* 2015). Excision of the digestive tract has been used to ascertain consumption of microplastics by vertebrates and invertebrates and is relevant for microplastics > 0.5 mm in size. Microplastics present in dissected tissues can be isolated using saline washes, density flotation, visual inspection or digestion.

Enumerating microplastics present in biota or excised tissues can be challenging if biological material, microbial biofilms, algae and detritus mask the presence of plastic particles. To isolate microplastics, organic matter can be digested, leaving only recalcitrant materials (Table A1.3). Several different digestion protocols have been developed (see Lusher *et al.*, 2017) and include: acids (e.g. nitric acid, formic acid, hydrochloric acid, alkalis (e.g. potassium hydroxide), oxidizing agents (e.g. hydrogen peroxide) and enzymes (Proteinase K). To date the use of Potassium hydroxide (KOH) seems to be the most appropriate strategy (Dehaut *et al.*, 2016; Kühn *et al.*, 2016; Lusher *et al.*, 2017). Following digestion, chemical agents can be filtered to retain any recalcitrant materials (e.g. undigested tissue, inorganic residue, microplastics). Microplastics on filters can be visualized directly, transferred to slides, or extracted.

Density separation can be used in biotic studies to separate plastic particles from organic material (although this is more common in sediment and water samples). Density separation can also be useful in studies following digestion. Saturated salt solutions, such as Sodium chloride (NaCl) (aq) allow the separation of less dense particles where there is large amounts of inorganic matter (e.g. sand, chitin, bone) that has not been dissolved. Density separation has been recommended by the MSFD. NaCl is recommended because it is inexpensive and non-hazardous; however, the use of NaCl could lead to an underestimation of more dense particles (>1.2 g cm^{-3}). Sodium iodide (NaI) and Zinc chloride ($ZnCl2$) solutions have been considered as viable alternatives to NaCl (aq) (Horton *et al.*, 2017a). Their high density makes them capable of floating high-density plastics including polyvinyl chloride (PVC).

A1.4.2 Microplastic identification

Following the processing and preparation of target tissues, the quantity and types of microplastics should be ascertained. Of the methods currently employed, visual identification is most widely utilized; often in combination with one or more follow-up analytical techniques. Researchers can use characteristics, including morphology and density, to identify the presence of microplastics. Visual identification is based on the morphological and physical characteristics of particles whereas chemical characteristics are determined by more advanced analytical techniques. Visual identification is an essential step in classifying microplastics, and is perfectly acceptable when supported by subsequent polymer analysis of sub-samples. Visual identification can be conducted using light, polarized or electron microscopy. Semi-automated methods, including ZooScan, flow, cell sorters and coulter counters to allow for many samples to be analysed; however, these require technical expertise and specialized equipment. Scanning Electron Microscopes (SEM) produce high resolution images and have been implemented in several studies (see Lusher *et al.*, 2017).

Visual identification is rapid, relatively cheap and can be conducted without the need for additional technical staff and consumables; however, accurately differentiating microplastics, particularly in the smaller size ranges, requires training and experience or may not even be possible especially considering particle size < 50 µm. For bigger particles criteria for visually identifying microplastics include: the absence of cellular or organic structures; a homogeneous thickness across the particles; and homogeneous colours and gloss. Plastics are largely classified by their morphological characteristics: size, shape and colour. Size is typically based on the longest dimension of a particle; size categories can be used where appropriate. When reporting microplastic shape, researchers tend to use five main categories: beads, fibres, fragments, foams and films. Finally, colours are often reported across a wide spectrum; colour differentiation is subjective, and visual identification of microplastics cannot be based on colour alone. Caution should be given to categorizing microplastics suffering embrittlement, fragmentation or bleaching, or encrusted with biota, as this may skew results.

Due to the challenges in visually identifying microplastics, especially in the smaller size range, secondary analyses should be used to confirm the identity of suspected polymeric material. For bigger microplastics, the simplest technique is the use of a hot needle to observe melting points. While both cheap and fast, this method does not allow for accurate polymer identification; however, the temperature range at which melting occurs may provide a specific range of potential polymers.

A converse method is to exclude non-plastics rather than identifying the plastics present through oven or freeze drying. Other low-cost techniques involve the examination of microplastics under a polarized light microscope to observe the birefringent properties of the suspected polymer or the use of Nile red to colour plastic polymers (Shim, Hong and Eo, 2017).

More complex - and costly - methods can be used to infer resin constituents, plastic additives and dyes. Often, these techniques require the purification of potential microplastic prior to analysis. The removal of biofilms, organic and inorganic matter adhered to the surface will avoid impeding polymer identification and the removal of non-plastic particles. Indeed, the proportion of particles exhibiting saturated signal could potentially be reduced by applying a pre-treatment of the samples to remove the organic matter (e.g. biofilm, algae, etc.) covering the microplastics, as performed by Imhof *et al.* (2016).

Following such purification, suspected plastics can be subjected to analytical techniques including: non-destructive vibrational techniques such as Fourier Transformed Infra-Red spectrometry (FT-IR) in transmittance or reflectance; Attenuated Total Reflectance (ATR) and Raman spectrometry; and destructive techniques such as Pyrolysis–Gas Chromatography combined with Mass Spectroscopy (Pyr-GC-MS), high temperature gel-permeation chromatography (HT-GPC) with IR detection; SEM–EDS and thermoextraction; and desorption coupled with GC/MS. Raman and FTIR spectroscopies are preferred as they are non-destructive and they are also complementary techniques, as molecular vibrations which are Raman inactive can be FTIR active and vice versa. To date, Raman is able to achieve a better spatial resolution (down to 1 µm) than FTIR (10 µm) (Lenz *et al*., 2015) but the identification of fibres relies mainly on FTIR as Raman analyses did not prove to be efficient so far for this type of microplastics (Dris, 2017)). For more information on these techniques refer to Lenz *et al*., 2015, Wesch *et al*., 2016 and Löder and Gerdts, 2015. It is also noteworthy that automated scanning coupled to microspectrometry could provide a robust method to thoroughly analyse large environmental samples without having to rely on visual identification which is operator dependent and not efficient for the smaller microplastic fraction (<50 µm) (Frère *et al*., 2016).

TABLE A1.1

Examples of methods used for sampling microplastics from the water column

Method	Explanation	Advantages	Disadvantages	Examples of use
Towed Neuston net	• Fine-mesh net attached to a large rectangular frame developed for sampling surface and near-surface waters for plankton, insects and other small biota. • Net length typically 1-8 m. • Mesh size typically 200-333 µm	• Can be deployed from small to large vessels. • Underway sampling	• Use is weather dependant • Care needed to minimize contamination from sampling vessel and tow ropes. • Can only estimate volume of water filtered when flow meter is used • Towing time must be limited to avoid clogging the net and under-sampling surface waters • Under-samples material which have at least one dimension smaller than the sampling mesh size • Vessel speed may need to be restricted	Portuguese coast (Frias *et al.*, 2014)
Towed Manta net	• Similar to the Neuston net with additional 'wings' to restrict sampling to the water surface. • Mesh size typically 330 µm. • Standard deployment configured with long side parallel to sea surface. • An alternative arrangement is to configure the net with the long side vertical (altered this way for high-speed tow).	• Standard manta net can be deployed from small to large vessels. • Samples the sea surface. • Use of flow meter allows estimation of water volume sampled. • Vertically-configured net allows deployment at higher speed. • Underway sampling	• Cannot account for environmental variables • Care needed to minimize contamination from sampling vessel and tow ropes. • Use is weather dependant • Towing time must be limited to avoid clogging the net and under-sampling surface waters • Can only estimate volume of water filtered when flow meter is used • Under-samples material < 330 µm diameter • Vessel speed may need to be restricted	South Pacific (Eriksen *et al.*, 2013b)
Horizontally hauled plankton nets	Cylindrical-Conical shaped, often used for mid-water sampling	• Can be deployed from vessels • Can be used at variable depths • Use of flow meter allows estimation of water volume sampled • Use is not weather dependant	• Risk of sample contamination when the sample is handled on the vessel deck after each sampling procedure. • Under-samples material < 300, 110 and 65µm • Vessel speed may need to be restricted	Goiana estuary (Brazilian east basin); Lima *et al.*, 2014

Method	Explanation	Advantages	Disadvantages	Examples of use
Bongo nets	Paired zooplankton net system	• Can be used for the water column • Paired nets can obtain replicate samples from same area	• High potential for contamination	North Pacific (Doyle *et al.*, 2011)
Underway pumps	Utilizing seawater intakes from vessels	• Can sample a known volume of water over a given time or distance • Can control for contamination on vessel	• Intakes are small and can limit the upper size range • Adverse sea states can affect the position of vessel in water, intake depth variable. • May be contamination from the sampling apparatus including the hose	Northeast Atlantic (Lusher *et al.*, 2014); Northeast Pacific (Desforges et al. 2014)
Submersible pumps	• Deck pump lowered to a known depth	• Can sample a known volume of water	• Vessel needs to be stationary • Intakes are small and can limit the upper size range	Sweden (Norén and Naustvoll, 2011)
Grab method	• Sample collected at surface in a container	• Known volume • Contamination reduced	• Smaller volume • Vessel needs to be stationary	Maine, USA (Barrows *et al.*, 2017)
Bulk sample	• Sampling large volume of water and volume reducing	• Known volume	• Volume reducing sample on a working deck may expose sample to contamination. Care must be taken.	Republic of Korea (Song *et al.*, 2014)
CPR	• Continuous plankton recorder towed from ships underway • Have been in use since 1946	• Can be used over a large distance from vessels of opportunity • Can use archived samples	• Water depth sampled is approximately -10m, i.e. cannot sample surface waters • Restricted size of intake may underestimate larger particles	North Atlantic (Thompson *et al.*, 2004)
Epibenthic sledge	• Sampling gear designed for benthic and bentho-pelagic collection	• Can sample just above the seabed	• Can be difficult to accurately estimate the amount of water sampled • Weather dependant • Contamination concern from deck storage and improper rinsing • Size of particles captured is dependant on the mesh size	California USA (Lattin *et al.*, 2004)

TABLE A1.2

Examples of methods used for sampling microplastics from sediments

Method	Explanation	Advantages	Disadvantages	Examples of use
Littoral zone/beach				
Selective sampling	Use of forceps or similar handheld tool to collect visible particles from sediment surface	Rapid sampling method Easy to include in participatory or citizen sampling initiatives Useful for sampling plastic resin pellets	Sampling efficiency is only as good as the collector. Size collected depends on visual ability of the sampler	United Kingdom (Ashton *et al.*, 2010; Turner and Holmes, 2011)
Separation of microplastics by dry sieving	Use of one or more sieves to extract larger microplastics from finer-grained organic and inorganic material.	Rapid sampling method Easy to include in participatory or citizen sampling initiatives Useful for sampling plastic resin pellets	Limited to relatively coarse mesh size of sieve Unsuitable for wet sediments without provision of water	Frias *et al.*, 2016; Wessel *et al.*, 2016
Separation by wet sieving	Use of one or more sieves to extract larger microplastics from finer-grained organic and inorganic material.		Limited to relatively coarse mesh size of sieve	Dekiff *et al.*, 2014

Method	Explanation	Advantages	Disadvantages	Examples of use
Sub-tidal sediments				
Grab sampler	Lowered to seabed where contact allows one or more jaws to close, capturing a sample of known area, measureable volume and measureable depth penetration. Several different designs available	Ease of use Small samplers can be used from small boats	Sediment surface may be disrupted during operation	Globally (Browne *et al.*, 2011; Claessens *et al.*, 2011)
Box corer	Lowered to seabed where contact captures a sample of known area, measureable volume and measureable depth penetration. Several different designs available	Maintains water-sediment interface Multi-Corers allow replicates at sites	Sediment surface may be disrupted during operation	North East Atlantic (Martin *et al.*, 2017)
Sediment gravity core	Lowered to seabed and allowed to penetrate vertically Several different designs available Capable of being operated at abyssal depths	Tends to better preserve sediment surface Can provide record of microplastics inputs	Relative small surface area sampled. Tends to require heavier lifting gear on vessel	North East Atlantic (Martin *et al.*, 2017)

TABLE A1.3

Optimised protocols for digesting biota or biogenic material to isolate microplastics. Assumptions: 'overnight' given as 12 h; 'room temperature' given as 20°C. Taken from Lusher *et al.*, 2017

Treatment	Exposure	Organism	Reference
HNO_3 (22.5 M)	20˚C (12 h) + 100˚C (2 h)	Blue mussels *(Mytilus edulis)*	Claessens *et al.*, (2013)
HNO_3 (22.5 M)	20˚C (12 h) + 100˚C (2 h)	Blue mussels Oysters *(Crassostrea gigas)*	Van Cauwenberghe and Jansen (2014)
HNO_3 (22.5 M)	20˚C (12 h) + 100˚C (2 h)	Blue mussels Lugworms *(Arenicola marina)*	Van Cauwenberghe *et al.*, (2015a)
HNO_3 (100%)	20˚C (30 min)	Euphausids Copepods	Desforges *et al.*, (2015)
HNO_3 (69-71%)	90˚C (4 h)	Manila clams *(Venerupis philippinarum)*	Davidson and Dudas (2016)
HNO_3 (70%)	70˚C (2 h)	Zebrafish *(Danio rerio)*	Lu *et al.*, (2016)
HNO_3 (22.5 M)	20˚C (12 h) + 100˚C (15 min)	Brown mussels *(Perna perna)*	Santana *et al.*, (2016)
HNO_3 (65%) HClO4 (68%) (4:1)	20˚C (12 h) + 100˚C (10 min)	Blue mussels	De Witte *et al.*, (2014)
HNO_3 (65%) HClO4 (68%) (4:1)	20˚C (12 h) + 100˚C (10 min)	Common shrimp *(Crangon crangon)*	Devriese *et al.*, (2015)
CH_2O_2 (3%)	72 h	Corals	Hall *et al.*, (2015)
KOH (10%)	14-21 d	Fish	Foekema *et al.*, (2013)
KOH (10%)	60˚C (12 h)	Fish	Rochman *et al.*, (2015)
KOH (10%)	14-21 d	Fish	Lusher *et al.*, (2017)
H_2O_2 (30%)	60˚C	Blue mussels	Mathalon and Hill (2014)

Treatment	Exposure	Organism	Reference
H_2O_2 (30%)	20˚C (7 d)	Biogenic matter	Nuelle *et al.,* (2014)
H_2O_2 (15%)	55˚C (3 d)	Fish	Avio *et al.,* (2015b)
H_2O_2 (30%)	65˚C (24 h) + 20˚C (<48 h)	Bivalves	Li *et al.,* (2015)
NaClO (3%) NaClO/HNO_3 (10:1)	20˚C (12 h) 20˚C (5 min)	Fish	Collard *et al.,* (2015)
Proteinase K	50˚C (2 h)	Zooplankton Copepods	Cole *et al.,* (2014)
Corolase 7089	60˚C (12 h)	Blue mussels	Catarino *et al.,* (2017)
Lipex® 100L Savinase® 16L (2:1)	60˚C (12 h)	Mediterranean mussels *(Mytilus galloprovincialis)*	Jang *et al.,* (2016)

Annex 2. Common synthetic polymers, field observations of microplastics in aquatic environments, and interactions between aquatic organisms and microplastics observed in experimental studies

TABLE A2.1

Common synthetic polymers – source, use and degradation properties (UNEP, 2015). Numerals as superscripts in columns 6 and 7 correspond to relevant references in column 8

Abbreviation	**Full name**	**Common source**	**Type**[*]	**Examples of common uses**	**Biodegradation properties in terrestrial environment (including medical applications)**	**Biodegradation properties in aquatic/marine environment**	Reference
ABS	(acrylonitrile butadiene sytyrene) copolymer	Fossil fuel	TP	Pipes, protective headgear, consumer goods, Lego™ bricks			
AC	Acrylic	Fossil fuel	TP	Acrylic glass (see PMMA)			
AcC (CTA, TAC)	Acetyl cellulose, cellulose triacetate	Biomass	TS	Fibres, photographic film base	Biodegradability depends on degree of acetylation[1]		[1] Tokiwa *et al.,* 2009
AKD	Alkyd	Partly biomass	TS	Coatings, moulds			
Cellophane	Cellophane	Biomass (cellulose)	TS	Film for packaging			
DECP	A group of degradable and electrically conductive polymers	Biomass & fossil fuel		Biosensors and tissue engineering	Degradable within living tissues[2]		[2] Guo *et al.,* 2013
EP	Epoxy resin (thermoset)	Fossil fuel	TS	Adhesives, coatings, insulators			
PA	Polyamide e.g. Nylon™ 4, 6, 11, 66; Kevlar™	Fossil fuel	TP	Fabrics, fishing lines and nets,			
PAN	Polyacrylonitrile	Fossil fuel	TP	Fibres, membranes, sails, precursor in carbon fibre production			
PBAT	Poly(butylene adipate-co-teraphthalate)	Fossil fuel	TP	films	Biodegradable[7]		[7]Weng *et al.,* 2013

Abbreviation	Full name	Common source	Type*	Examples of common uses	Biodegradation properties in terrestrial environment (including medical applications)	Biodegradation properties in aquatic/marine environment	Reference
PBS	Poly(butylene succinate)	Fossil fuel	TP	Agricultural mulching films, packaging	Biodegradable[1]	Some degradation after 12 months but retains 95% tensile strength[3]	[1]Tokiwa *et al.*, 2009
						Some degradation after 2 years[4]	[3]Sekiguchi *et al.*, 2011
							[4] Kim *et al.*, 2014a,b
PCL	Polycaprolactone	Fossil fuel	TP	3D printing, hobbyists, biomedical applications	Biodegradable by hydrolysis in the human body	Some degradation after 12 months[3]	[1]Tokiwa *et al.*, 2009
					Biodegradable[1]		[3] Sekiguchi *et al.*, 2011
PC	Polycarbonate	Fossil fuel and renewable resource	TP	Electronic components, construction materials, data storage devices, automotive and aircraft parts, check sockets in prosthetics, and security glazing	Biodegrabale if physico-chemically pretreated [1]	Some degradation after 12 months[2]	[1]Artham and Doble, 2009a [2]Artham and Doble, 2009b
PE	Polyethylene	Biomass & fossil fuel	TP	Packaging, containers, pipes		Extremely limited, potential minor effect in Tropics due to higher temperature, dissolved oxygen and microfauna/ flora assemblages[5]	[5]Sudhakar *et al.*, 2007
PES	Poly(ethylene succinate)	Fossil fuel	TP	films	Biodegradable[1]		[1]Tokiwa *et al.*, 2009
PET	Polyethylene terephthalate	Fossil fuel, fossil fuel with biomass	TP	Containers, bottles, ‘fleece’ clothing			
PGA	Poly(glycolic acid)		TP	Sutures, food packaging	Biodegradable by hydrolysis in the human body		
PHB	Poly(hyroxybutyrate)	Biomass	TP	Medical sutures	Biodegradable[1]		[1]Tokiwa *et al.*, 2009
					Some degradation after 12 months[3]		[3]Sekiguchi *et al.*, 2011
PLA	Poly(lactide)	Biomass	TP	Agricultural mulching films, packaging, biomedical applications, personal hygiene products, 3D printing	Biodegradable[1]		[1]Tokiwa *et al.*, 2009
					Compostable[5]		[5] Pemba *et al.*, 2014

Abbreviation	**Full name**	**Common source**	**Type***	**Examples of common uses**	**Biodegradation properties in terrestrial environment (including medical applications)**	**Biodegradation properties in aquatic/marine environment**	Reference
PMMA	Poly (methyl) methylacrylate	Fossil fuel	TP	Acrylic glass, biomedical applications, lasers			
POM	Poly (oxymethylene) Also called Acetal	Fossil fuel	TP	High performance engineering components e.g. automobile industry			
PP	Polypropylene	Fossil fuel	TP	Packaging, containers, furniture, pipes			
PS	Polystyrene	Fossil fuel	TS	Food packaging			
EPS	Expanded polystyrene	Fossil fuel	TP	Insulation panels, insulated boxes, fishing/aquaculture floats, packaging			
PU (PUR)	Polyurethane	Fossil fuel	TS	Insulation, wheels, gaskets, adhesives			
PVA	Poly(vinyl alcohol)	Fossil fuel	TP	Paper coatings	biodegradable		
PVA	Poly(vinyl acetate)	Fossil fuel	TP	Adhesives			
PVC	Poly(vinyl chloride)	Fossil fuel	TP	Pipes, insulation for electric cables, construction			
Rayon	Rayon	Biomass (cellulose)	TS	Fibres, clothing	biodegradable	biodegradable	
SBR	Styrene-butadiene rubber	Fossil fuel	TS	Pneumatic tyres, gaskets, chewing gum, sealant			
Teflon	Polytetrafluoro-ethylene	Fossil fuel	TP	Electric isolation, non-stick coating			

* Type: TP= Thermoplastic, TS = Thermoset.

TABLE A2.2

Global reports of microplastics in freshwater and estuarine environments dw refers to dry weight

			Range reported and original unit				
Location	**Waterbody**	**Substrate**	**Items per km^2 or kg**	**Items per m^2**	**items/ 100 m^3**	**Standard reporting unit^1**	**Reference**
America							
Lakes Huron, Superior, Erie, Canada/USA	Freshwater lake	Water	0–466 305	0-0.466305		0–0.093 items/m^3	Eriksen *et al.,* 2013a
Lake Ontario, Canada/USA	Freshwater lake	Sediment	20–27 830			0.02–27.83 items/g dw	Ballent *et al.,* 2016
Lake Huron, Canada/USA	Freshwater lake	Sediment				0–34 items/m^2	Zbyszewski and Corcoran 2011
Lakes Erie and St. Clair, Canada/ USA	Freshwater lake	Sediment	0.09–0.47/g			0.18–8.28 items/m^2	Zbyszewski *et al.,* 2014
St. Lawrence River, Canada/ USA,	Freshwater	Sediment	297.97			0–136 926 items/m^2	Castañeda *et al.,* 2014
Coyote, San Gabriel, Los Angeles Rivers, USA	Freshwater	Water				0–12 935 items/m^3	Moore *et al.,* 2011
Chicago, Illinois, USA	Freshwater	Water				1.94–17.93 items/m^3	McCormick *et al.,* 2014
Chicago, Illinois, USA	Freshwater	Sediment				0.018 items/m^2	Hoellein *et al.,* 2014
Chesapeake Bay, USA	Estuarine	Sediment	5 534–297 927			5.53–297.93 items/g dw	Yonkos *et al.,* 2014
Goiana estuary (main channel)	Estuarine	Water			0.3–13.9	0.0037–0.13980 items/m^3	Lima *et al.,* 2014
Goiana estuary (mangrove)	Estuarine	Water			0.218–1.991	0.00218–0.01991 items/m^3	Lima *et al.,* 2015
Europe							
River Seine, France	Freshwater	Water				3–108 items/m^3	Dris *et al.,* 2015b
Lake Geneva, Constance, Neuchatel, Maggiore, Zurich, Basel, Switzerland	Freshwater	Sediment				20–7 200 items/m^2	Faure *et al.,* 2015
Lakes Geneva, Constance, Neuchatel, Maggiore, Zurich, Basel, Switzerland	Freshwater	Water	26 000	0.026		0.0052 items/m^3	Faure *et al.,* 2015
Lakes Chiusi, Bolsena, Italy	Freshwater	Sediment				1 922–2 117 items/m^2	Fischer *et al.,* 2016
Lakes Chiusi, Bolsena, Italy	Freshwater	Water				0.82–4.42 items/m^3	Fischer *et al.,* 2016
Thames River, England	Freshwater	Sediment				185–660 items/m^3	Horton *et al.,* 2017a
Lake Garda, Italy	Freshwater	Sediment				Maximum: 1 108 items/m^2	Imhof *et al.,*2013
Rivers Rhine & Main, Germany	Freshwater	Sediment	228–3 763			0.228–3.763 items/g dw	Klein *et al.,* 2015
River Danube, Austria	Freshwater	Water				0–1.4 x 10^8 items/m^3	Lechner and Ramler 2015
River Rhine, Germany	Freshwater	Water				10 472.8–786 212.4 items/m^3	Mani *et al.,* 2015
Tamar Estuary, England	Freshwater	Water				Mean: 0.028 items/m^3	Sadri and Thompson, 2014
Rivers Rhine, Elba, Mosel, Neckar, Germany	Freshwater	Sediment	0–64			0–0.064 items/g dw	Wagner *et al.,* 2014

Range reported and original unit							
Location	**Waterbody**	**Substrate**	**Items per km^2 or kg**	**Items per m^2**	**items/ 100 m^3**	**Standard reporting unit[1]**	**Reference**
Asia							
Lake Hovsgol, Mongolia	Freshwater	Water	997–44 435	0.000997–0.044		0.0002–0.0089 items/m^3	Free *et al.*, 2014
Yangtze River, China	Freshwater	Water				500–10 200 items/m^3	Zhao *et al.*, 2014
Three Gorges Dam, China	Freshwater	Water	192 500–13 617 000	0.19–13.62		0.038–2.73 items/m^3	Zhang *et al.*, 2015
Pearl River Estuary, China	Estuarine	Sediment				16–258 408 items/m^2	Fok and Cheung 2015
Lake Taihu, China	Freshwater	Water	10 000–6 800 000			0.002–1.36 items/m^3	Su *et al.*, 2016
Lake Taihu, China	Freshwater	Sediment	11–234.6			0.011–0.04692 items/g dw	Su *et al.*, 2016
Lakes * China	Freshwater	Water				2650–10 050 items/m^3	Wang *et al.*, 2016

* Includes 20 urban lakes and urban reaches of the rivers Hanjiang and Yangtze

[1] In order to allow for comparisons between studies with different reporting units we have converted results to uniform units of measurement. However, it must be noted that these conversions are approximations.

TABLE A2.3

Microplastics distribution in marine environments

Range reported and original unit					
Location	**Substrate**	**Items per km^2 or mass**	**Items per m^2**	**Standardised reporting unit[1]**	**References**
Pacific Ocean					
	Water	/	/	0–6.63 x 10^2 items/ha	Uchida *et al.*, 2016
North Pacific					
Bering Sea, Subarctic	Water	80–3 370	0.000080–0.0034	0.000016–0.00068 items/m^3	Day and Shaw 1987;
Bering Sea	Water	/	/	0.004–0.19 items/m^3	Doyle *et al.*, 2011
Hawaiian Islands	Sediment	37.8		Mean: 0.038 items/g	McDermid and McMullen 2004; Corcoran *et al.*, 2009; Cooper and Corcoran 2010; Carson *et al.*, 2011; Young and Elliott 2016
North West Pacific					
Japanese coast and Kuroshio Current	Water	0–3 520 000	/	0.03–11.1 items/m^3	Yamashita and Tanimura 2007; Isobe 2016; Isobe *et al.*, 2014, 2015; Day *et al.*, 1990
Japanese coast	Sediment	/	/	0.52–1 000 items/m^2	Kuriyama *et al.*, 2002; Rios *et al.*, 2007; Kusui and Noda 2003; Endo *et al.*, 2005
Korean coast	Water	/	/	2.6–359 748 items/m^3	Song *et al.*, 2014,2015a, 2015b; Kang *et al.*, 2015; Chae *et al.*, 2015
Korean coast	Sediment	/	/	1.2–285 673 items/m^2	Lee *et al.*, 2013, 2015; Kim *et al.*, 2015; Heo *et al.*, 2013
Korean coast	Sediment	/	/	88 items/L	Song *et al.*, 2015a
North East Pacific					
Canadian Pacific coast	Water	/	/	8 - 9 200 items/m^3	Desforges *et al.*, 2014
Canadian Pacific coast	Sediment	/	/	average: 0.45 items/g	Cluzard *et al.*, 2015
West coast, USA	Water	/	/	0 – > 60 items/m^3	Moore *et al.*, 2002; Lattin *et al.*, 2004; Gilfillan *et al.*, 2009; Sutton *et al.*, 2016
West coast, USA	Sediment	/	/	/	Rios *et al.*, 2007; Ogata *et al.*, 2009; Van *et al.*, 2012

		Range reported and original unit			
Location	**Substrate**	**Items per km^2 or mass**	**Items per m^2**	**Standardised reporting unit[1]**	**References**
North Pacific Central and subtropical gyres	Water	/	/	0–2.23 items/m^3	Moore *et al.*, 2001; Carson *et al.*, 2013; Goldstein *et al.*, 2012, 2013
South Pacific					
South Pacific subtropical gyre	water	0–396 342	0–0.39	0–0.079 items/m^3	Eriksen *et al.*, 2013b
Easter Island, Chile	Beach sediment	/	/	30–805 items/m^2	Hidalgo-Ruz and Thiel 2013
Australia	Water	0–48 893	0–0.049	0–0.0098 items/m^3	Reisser *et al.*, 2013
North Atlantic					
North Atlantic	Water	/	/	0.01–0.04 items/m^3	Thompson *et al.*, 2004;
North West Atlantic	Water	490–3 537	0.00049–0.0035	0.000098–0.0007 items/m^3	Wilber 1987; Law *et al.*, 2010; Colton *et al.*, 1974; Carpenter and Smith 1972
Eastern USA Coast	Water	/	/	14–543 items/m^3	Austin and Stoops-Glass 1977
Nova Scotia, Canada; Maine, Florida, USA;	Sediment	105–800	/	0.105–0.8 items/g	Mathalon and Hill, 2014; Graham and Thompson, 2009; Gregory 1983;
North Atlantic subtropical gyre	Water	/	/	0.01–501 items/m^3	Enders *et al.*, 2015; Ivar do Sul *et al.*, 2013
NE Atlantic (English & Bristol Channel, Portuguese & Spanish coasts)	Water	/	/	0–2.46 items/m^3	Lusher *et al.*, 2014; Cole *et al.*, 2014; Morris & Hamilton 1974; Gago *et al.*, 2015; Frias *et al.*, 2014;
English Channel	Sediment	31–86	/	0.031–0.086 items/g	Thompson *et al.*, 2004;
NE Atlantic (Portugal)	Sediment	/	/	40–1 289 items/m^2	Martins & Sobral, 2011; Antunes *et al.*, 2013; Van Cauwenberghe *et al.*, 2013a
NE Atlantic (Portugal)	Sediment	/	/	0–6.2628 items/g	Frias *et al.*, 2016
NE Atlantic (Canary islands)	Sediment	/	/	0.001–0.04 g/g	Baztan *et al.*, 2014;
Caribbean Sea	Water	60.6–1 414	0.000061–0.0014	0.000012–0.00028 items/m^3	Colton *et al.*, 1974; Law *et al.*, 2010;
Florida Keys, Cape Cod, Mobile Bay & Caribbean Sea	Sediment	/	/	50–10 000 items/m^2	Wilber 1987; Wessel *et al.*, 2016
South Atlantic Bight, USA	Water	/	0.03–0.08	0.006–0.016 items/m^3	Van Dolah *et al.*, 1980
South Atlantic					
Offshore, St Peter and St Paul Archipelago	Water	/	/	0.03–0.26 items/m^3	Ivar do Sul *et al.*, 2013; Lima *et al.*, 2016;
South East Atlantic	Water	1 874.3–3 640	0.0018–0.0036	0.00037–0.00073 items/m^3	Morris, 1980; Ryan, 1988
South Africa	Water	/	/	257–1 215 items/m^3	Nel and Froneman, 2015
South West Atlantic	Sediment	/	/	0–300 000 items/m^3	Costa *et al.*, 2010; Costa *et al.*, 2011; Turra *et al.*, 2014;
South East Atlantic	Sediment	/	/	40–3 308 items/m^2	Nel and Froneman, 2015; Van Cauwenberghe *et al.*, 2013a
South West Atlantic	Sediment	0.73–9.63	/	0.00073–0.0096 items/g	Ivar do Sul *et al.*, 2009;
European seas and the Mediterranean					
Baltic Sea	Water	/	/	100–10 000 items/m^3	Gorokhova 2015

		Range reported and original unit			
Location	**Substrate**	**Items per km^2 or mass**	**Items per m^2**	**Standardised reporting unit**[1]	**References**
Sweden, Finland, Baltic, Mediterranean, Adriatic	Water			0.01–102 000 items/m^3	Norén 2007; Norén and Naustvoll, 2011; Gorokhova, 2015; de Lucia *et al.*, 2014; Gajst *et al.*, 2016; Magnusson, 2014; Fossi *et al.*, 2012; Panti *et al.*, 2015
Spanish Mediterranean	Water		0.01–4.52	0.002–0.904 items/m^3	Collignon *et al.*, 2012; Collignon *et al.*, 2014; Gago *et al.*, 2015; Ruiz-Orejón *et al.*, 2016; Suaria *et al.*, 2016
North Sea	Sediment	1.3–3 800		0.0013–3.8 items/g	Norén, 2007; Liebezeit & Dubaish, 2012; Dubaish & Liebezeit, 2013; Fries *et al.*, 2013; Dekiff *et al.*, 2014; Ogata *et al.*, 2009; Claessens *et al.*, 2011; Van Cauwenberghe *et al.*, 2013b;
Mediterranean	Sediment			0.7–575 items/m^2	Turner & Holmes, 2011; Van Cauwenberghe *et al.*, 2013a; Kaberi *et al.*, 2013;
Mediterranean, Slovenian coast	Sediment	0–2 175		0–2.18 items/kg dw	Vianello *et al.*, 2013; Laglbauer *et al.*, 2014
Asian seas					
East Asian Sea	Water			0.03–491 items/m^3	Isobe *et al.*, 2015
India	Sediment	81 mg/kg		Mean: 0.0081 mg/g	Reddy *et al.*, 2006
Singapore	Sediment	36.8		0.037 items/g	Mohamed Nor and Obbard 2014
Malaysia	Sediment			Mean: 18 items/m^2	Ismail *et al.*, 2009
Hong Kong	Sediment			Mean: 5 595 items/m^3	Fok & Cheung, 2015;
Strait of Hormuz, Persian Gulf	Sediment	2–1 258		0.0002–1.26 items/g dw	Naji *et al.*, 2017
Arabian Gulf and Gulf of Oman	Sediment			50–80 000 items/m^2	Khordagui & Abu-Hilal, 1994;
Polar					
Arctic	Sea ice	/	/	/	Obbard *et al.*, 2014
Barents Sea	Water			0–11.5 items/m^3	Lusher*et al.*, 2015b
Greenland	Water	/	/	Mean: 0.99–2.38 items/m^3	Amélineau *et al.*, 2016
Southern Ocean	Water	/	/	0.0035–0.099 items/m^3	Isobe *et al.*, 2017
Ross Sea, Antarctic	Water	/	/	0.0032–1.18 items/m^3	Cincinelli *et al.*, 2017

[1] In order to allow for comparisons between studies with different reporting units we have converted results to uniform units of measurement. However, it must be noted that these conversions are approximations.

TABLE A2.4

Summary of observed interactions of aquatic organisms with microplastics under laboratory conditions

Taxonomic Group	Exposure scenario	Exposure duration	Interactions	References
Algae	1.10-56.95 particles/ml	< 72 h	Whole organism, adherence	Gutow *et al.*, 2016
Phytoplankton	0.000046–40 mg/ml 0.01% weight/volume 9 x 10^4 particles/ml	1–96 h	Adhesion, absorption, affected growth, inhibition of photosynthesis	Bhattacharya *et al.*, 2010; Cedervall *et al.*, 2012; Long *et al.*, 2015; Davarpanah and Guilhermino 2015; Sjollema *et al.*, 2016; Lagarde *et al.*, 2016
Zooplankton	635–10 000 particles/ml	4 h	Adhesion, ingestion, effects on fecundity, survival and fitness	Cedervall *et al.*, 2012; Cole *et al.*, 2013; 2014; 2015; Lee K.W. *et al.*, 2013; Setälä *et al.*, 2014
Cnidaria	0.395 g/L	48 h	Ingestion	Hall *et al.*, 2015
Protozoans	X	X	Adsorption	Christalki *et al.*, 1998
Annelids	1.5 g/L 0–5% by weight 0–100 particles/L 2 000 particles/ml	20 min to 28 d	Ingestion, egestion, rejection before digestion, no effect, weight loss, reduced feeding, decreased energy reserves, reduced fitness, increased metabolic rate, oxidative stress	Bolton and Haverhand 1998; Thompson *et al.*, 2004; Besseling *et al.*, 2013; Browne *et al.*, 2013; Wright *et al.*, 2013. Huerta-Lwanga *et al.*, 2016; Green *et al.*, 2016; Imhof *et al.*, 2016
Mollusca	1.05–3 000 particles/ml 0.5–2.5 g/L 50 µl in 400 ml 1–199 µg/ml	45 min to 7 d	Ingestion, transfer to haemolymph, cellular effects, reproduction, reduced feeding activity	Lei *et al.*, 1996; Brilliant and MacDonald 2000, 2002; Browne *et al.*, 2008; Ward and Kach 2009; von Moos *et al.*, 2012; Wegner *et al.*, 2012; Cole *et al.*, 2013; Farrell and Nelson 2013; Avio *et al.*, 2015a; Canesi *et al.*, 2015. See also review in Ward and Shumway 2004.
Echinoderms	1–300 particles/ml 10–60 g per 600 ml sand	20 h to 9 d	ingestion, retention, egestion, impact on embryonic development, no effect	Hart 1991; Graham and Thompson 2009; Kaposi *et al.*, 2014; Nobre *et al.*, 2015; Della Torre *et al.*, 2014
Crustacea	5.25 x 10^5 – 9.1 x 10^{11} particles/ml 40–1 000 particles/ml 0.3–120 mg/g 108–1 000 mg/kg	15 min to 60 d	ingestion, ventilation	Thompson *et al.*, 2004; Murray and Cowie 2011; Ugolini *et al.*, 2013; Chua *et al.*, 2014; Hämer *et al.*, 2014; Watts *et al.*, 2014, 2015; Brennecke *et al.*, 2015 and others
Amphibians	1 mm: 10, 10^3, 10^5 particles/ml 10 mm 0.1, 10 and 10^3 particles/ml	1–48 h	ingestion, egestion	Hu *et al.*, 2016
Fish	10% of diet 3 000 particles/ml 0.216 mg/L	3 min to 2 months	Ingestion, decreased predatory performance, increase AChE activity, weight loss,changes in feeding behaviour and metabolic performance, endocrine disruption and liver toxicity	dos Santos and Jobling 1991; Cedervall *et al.*, 2012; Oliveira *et al.*, 2013; Rochman *et al.*, 2013;2014; Mazurais *et al.*, 2015; De Sa *et al.*, 2015; Avio *et al.*, 2015b; Luís *et al.*, 2015 and others

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Glossary

GLOSSARY OF WORKING DEFINITIONS

Antioxidants

Antioxidants are added to synthetic polymers in order to prevent oxidation reactions. Oxidation, if not prevented, can affect the polymer by adverse colour change, loss of flexibility, loss of tensile strength and elongation, reduced impact resistance, cracking, and other surface deterioration. Oxidation can also affect the molecular weight of a polymer, which in turn affects its physical properties.

Absorption

In chemistry absorption is a physical or chemical process by which atoms, molecules or ions are incorporated into a bulk phase (gas, liquid or solid material).

Adsorption

In chemistry adsorption is a physical or chemical process by which atoms, molecules or ions adhere to a surface.

Bioconcentration

Uptake of chemical into or onto an organism from the surrounding environmental media (typically water) to levels exceeding its ambient concentration; bioconcentration by definition excludes uptake from food.

Bioaccumulation

Accumulation of substances, e.g. chemicals in an organism, from direct uptake from the surrounding environment and from ingestion of food.

Biomagnification

Increased concentration of a substance, e.g., a chemical, in the tissues of organisms at successively higher levels up the food chain.

Colourants

Pigments added to plastic materials to achieve a desired colour of the end product.

Flame retardants

Flame retardants are chemicals (e.g. organohalogens, such as brominated flame retardants) that are added to different materials (e.g. plastics and textiles) to reduce their flammability in order to meet safety standards. Flame retardants may be mixed with the material (additive flame retardants) or chemically bonded to it (reactive flame retardants).

Lubricants

Substances that are used to improve the processability of plastics by increasing their flowability. Internal lubricants improve the melt flow of material by lowering the viscosity and heat dissipation.

Plasticizer
A substance (e.g. Phthalates) added to a synthetic polymer (e.g. Polyvynil chloride) to enhance flexibility and softness, and reduce brittleness.

Sorption/Desorption
In chemistry sorption is a physical or chemical process by which one substance becomes attached to another. Absorption and adsorption are specific cases of sorption. Desorption is the opposite of sorption.

Ultra Violet (UV) stabilizers
Chemical compounds (for example, Benzophenone) added to plastic materials to inhibit reactions which cause undesirable chemical degradation from exposure to UV light.

This report looks into the issue of microplastics from the fisheries and aquaculture perspective. Based on existing scientific literature, a group of experts assessed the potential impact of microplastics and related contaminants on fish consumers' health and the ecological implications for aquatic organisms. A workshop was organized with invited experts (Rome, 5-8 December 2016) who complemented the published information and carried out a risk profiling of microplastics in aquaculture and fishery products. Despite the large amount of scientific data available, there are still significant knowledge gaps, in particular regarding impacts at fish population and community level, detailed data for a proper risk assessment and implications of nanoplastics presence in the marine environment. Nonetheless, measures should be taken at international, governmental and consumer levels to undertake cost-effective ecological and seafood safety risk assessments on micro- and nanoplastics and associated polymers, to reduce plastic use and encourage the use of alternative materials, recycling and the adoption of sustainable practices in using plastics and managing plastic pollution.

ISBN 978-92-5-109882-0 ISSN 2070-7010

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I7677EN/1/08.17