Chapter 8

Dietary Exposure to Additives and Sorbed Contaminants from Ingested Microplastic Particles Through the Consumption of Fisheries and Aquaculture Products

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Abstract Microplastics and nanoplastics may be found in the gastrointestinal tract of some aquatic animals and could potentially be ingested by humans if consumed whole. Information on the toxicity of plastic particles, as well as co-contaminants such as plastic additives, remains scarce. This represents a serious challenge to perform realistic risk assessments. An exposure assessment of selected plastic additives and co-contaminants of known toxicity associated with microplastics was carried out for shellfish in this study, which builds on an exposure assessment of microplastic additives and a limited number of associated contaminants in mussels conducted by the FAO in 2017. This study evaluates possible impacts to food safety by examining a diverse additives and associated sorbed contaminants. The results suggest that the levels of certain microplastic additives and sorbed co-contaminants in target animals (shrimp, prawns, clams, oysters, and mussels) do not pose a food safety threat to consumers. To get to further conclusions, an exposure assessment from the whole diet should be carried out and the toxicity of some of the most common polymers and plastic additives, as well as their mixtures, needs to be carefully evaluated.

8.1 Introduction

Plastic production has been increasing exponentially since the 1950s and was estimated to be 8300 million metric tons to date (Geyer et al. 2017). Since its first development in the 1800s, the production of plastic materials has changed to meet the needs of a variety of sectors and consumers and has enabled technological improvements and solutions. Due to their functional properties (“cheap and

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durable”), plastics have displaced many non-plastic materials, becoming the most utilized materials worldwide.

Plastics consist of a range of synthetic or semi-synthetic chemicals that are made of fossil resources and organic by-products. They are commonly divided into three categories: thermoplastics (polymers that can be re-melted), elastomers (elastic polymers that return to their original shape after being deformed), and thermosets (polymers that remain in a permanent solid state once hardened).

Depending on their specific use, polymers with different physical and chemical properties can be mixed and additives such as plasticizers, colourants, UV-stabilizers, flame retardants, and antioxidants can be added to improve the performance of the final product.

Recycling the complex mixtures of chemicals used for plastic production can be challenging, as well as the evaluation of their impact on the environment and human health. The extensive production of plastic requires efficient waste management systems, but most countries do not have the capacity to develop them.

Microplastic particles have been found in a variety of human food items, such as salt, beer, honey, and aquatic products, with seafood being the best-studied source of dietary intake of microplastics. Exposure to microplastic particles, their additives, and their sorbed co-contaminants depends on several factors, such as particle size, shape, chemical changes that occurred during processing and/or cooking of fisheries and aquaculture products, and consumption patterns.

A previous study on exposure to microplastics and associated co-contaminants suggests that exposure to this contaminant burden is typically less than 0.1% (FAO 2017). Microplastic contribution to the total dietary intake of additives and sorbed co-contaminants was estimated as very low, with maximum increases in BPA, PAHs, and PCBs load of less than 2%, 0.004%, and 0.006% respectively, after the ingestion of a portion of mussels (EFSA 2016). However, a recent study by Barboza et al. (2020) observed a clear correlation between microplastics intake in three species of wild-caught commercial fish and the levels of bisphenols in the muscle and liver. Higher microplastic concentrations in fish were correlated with higher levels of these compounds, whose concentration in the edible tissue exceeded the established limits for human safety set by the EFSA. Furthermore, a relation between the concentration of plastic-associated chemicals and ingested microplastics in marine organisms has already been hypothesized (Granby et al. 2018; Rochman et al. 2013; Teuten et al. 2009). These findings suggest that more investigations should be conducted on this subject to better identify the role of microplastics in the transfer of pollutants and which factors could influence the process. This chapter aims to provide an overview of the dietary exposure of microplastic particles, additives, and common microplastic co-contaminants through aquatic products using consumption data from the FAO/WHO database, while information on contamination levels of plastic pellets and microplastic ingestion by seafood are updated with the current literature. Moreover, four different groups of seafood were considered, to extend the exposure evaluation also to crustaceans and other bivalves. The final estimations are compared with the no observed effect levels (NOELs) and no observed adverse

effect levels (NOAELs) and can be useful to provide a better understanding of the potential impacts on food safety.

8.2 Sorption of Environmental Contaminants by Microplastics

A potential threat to human health deriving from the exposure to microplastics is that these materials can scavenge and thus concentrate pollutants already present in production waters. The ingestion of contaminated plastic could lead to higher exposure to toxic chemicals, with possible endocrine disruption and carcinogenicity. The main process leading to the interaction between microplastics and hydrophobic organic chemicals (HOCs) in the water column does not involve the formation of covalent bonds; thus, its reversible nature preserves the likelihood of chemical desorption from the matrix (Endo and Koelmans 2016). Besides, plastic polymers are also recognized as possible vectors of heavy metals in the marine environment (Holmes et al. 2012, 2014), being experimentally able to accumulate concentrations even 800-fold higher than in seawater (Brennecke et al. 2016). Many field studies such as the International Pellet Watch have reported the concentration levels of persistent organic pollutants (POPs) and metals sorbed on beached and marine pellets, in addition to plastic additives (Tables 8.1 and 8.2).

Sorption processes can be classified into adsorption and absorption, depending on the mechanism of interaction between the polymer and the chemicals. Absorption mainly occurs when the molecules of pollutants diffuse into the bulk matrix of the polymer and interact with it through weak van der Waals forces or hydrophobic interactions. This process is mainly driven by the preferential partitioning of the chemical on plastic compared to water, which is usually linearly and positively correlated to its octanol-water partition coefficient (K_{ow}), a parameter that measures the level of hydrophobicity (Lee et al. 2014; Li et al. 2018c; O'Connor et al. 2016). Adsorption refers to a process that results in the sorption of molecules that are confined to the surface of microplastics (Endo and Koelmans 2016).

Absorption mainly occurs onto rubbery polymers (i.e. PE and PP), where external molecules pass through and associate within their matrix (Hüffer and Hofmann 2016; Müller et al. 2018; Teuten et al. 2009; Wang et al. 2018a). These polymers are generally recognized as the ones concentrating the highest amounts of HOCs and are then possibly more dangerous for marine life (Endo et al. 2005; Fisner et al. 2017; Hirai et al. 2011; O'Connor et al. 2016; Wang and Wang 2018; Wang et al. 2018b) and possibly human health.

Some polymers present several functional groups on their surface, conferring a certain degree of polarity. These are mostly PS and PVC, whose glassy nature also yields the formation of nanovoids and pores on the surface, which are the sites of sorption. In this case, the process is mainly led by adsorption, a mechanism through which the chemicals more efficiently bind to the plastic polymer through ionic,

Table 8.1 Maximum concentrations of persistent organic pollutants (POPs) and metals sorbed on plastic pellets from field studies

Location	Sampling	Extraction	PCB (ng/g)	PAH (ng/g)	DDT (ng/g)	DDE (ng/g)	PBDE (ng/g)	HCH (ng/g)	Cd (ng/g)	Pb (µg/g)	References
Portuguese Coast	Metal mesh sieving	n-Hexane, hexane-acetone	223	44,800	41		n.a.	n.a.	n.a.	n.a.	Antunes et al. (2013)
Greek beaches	N.S.	Hexane	290	500	42	15	n.a.	3.5	n.a.	n.a.	Karapanagioti et al. (2011)
South Devon, England	Plastic tweezers	Dilute aqua regia digestion	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	10	1.08	Ashton et al. (2010)
South West England	Plastic tweezers	Dilute aqua regia digestion	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	76.7	1.64	Holmes et al. (2012)
Portuguese coast	N.S.	Hexane	310	24,000	49	n.a.	n.a.	3.3	n.a.	n.a.	Mizukawa et al. (2013)
Japanese coast	Solvent rinsed stainless tweezers or by hand with PE gloves	Hexane	117	n.a.	n.a.	3.1	n.a.	n.a.	n.a.	n.a.	Mato et al. (2001)
Japanese beaches	Solvent rinsed stainless tweezers	n-Hexane	2300	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	Endo et al. (2005)
Santos Bay, Brazil	Tweezers	DCM: n-hexane	n.a.	39,763	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	Fisner et al. (2017)
Portuguese coast	N.S.	Hexane, hexane: acetone	54	319.6	4.54	0.81	n.a.	n.a.	n.a.	n.a.	Frias et al. (2010)
Belgian coast	N.S.	Hexane: DCM	236	3007	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	Gauquie et al. (2015)
Central Pacific Gyre, Pacific Ocean	Neuston net and solvent rinsed stainless steel tweezers	DCM	78	868	4.8	2	9909	n.a.	n.a.	n.a.	Hirai et al. (2011)

Japanese beaches	Neuston net and solvent rinsed stainless steel tweezers	DCM	436	9297	198	198	230	n.a.	n.a.	n.a.	Hirai et al. (2011)
Tonkin Bay, Vietnam	Neuston net and solvent rinsed stainless steel tweezers	DCM	102	2024	108	42	412	n.a.	n.a.	n.a.	Hirai et al. (2011)
Seal Beach, California	Neuston net and solvent rinsed stainless steel tweezers	DCM	399	656	8	7.6	41	n.a.	n.a.	n.a.	Hirai et al. (2011)
Marbella Beach, Costa Rica	Neuston net and solvent rinsed stainless steel tweezers	DCM	61	284	124	10	180	n.a.	n.a.	n.a.	Hirai et al. (2011)
Hawaii	Squeeze, scoops, or directly in glass jars	DCM	980	500	22	n.a.	n.a.	n.a.	n.a.	n.a.	Rios et al. (2007)
Mexico	Squeeze, scoops, or directly in glass jars	DCM	n.a.	640	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	Rios et al. (2007)
California	Squeeze, scoops, or directly in glass jars	DCM	730	6200	1100	n.a.	n.a.	n.a.	n.a.	n.a.	Rios et al. (2007)
9 Japanese beaches	Solvent rinsed stainless tweezers	DCM	892	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	Takada et al. (2006)
Japanese coast	Neuston net	N.S.	254	9370	n.a.	276	2.1	n.a.	n.a.	n.a.	Teuten et al. (2009)
Central Gyre	Neuston net	N.S.	23	959	n.a.	4.7	57	n.a.	n.a.	n.a.	Teuten et al. (2009)
Japan and South East Asia	Soap rinsed fingers or stainless steel tweezers	Hexane	453	n.a.	163	11.5	n.a.	1.23	n.a.	n.a.	Ogata et al. (2009)

(continued)

Table 8.1 (continued)

Location	Sampling	Extraction	PCB (ng/g)	PAH (ng/g)	DDT (ng/g)	DDE (ng/g)	PBDE (ng/g)	HCH (ng/g)	Cd (ng/g)	Pb (µg/g)	References
US. Coast	Soap rinsed fingers or stainless steel tweezers	Hexane	605	n.a.	267	128	n.a.	0.94	n.a.	n.a.	Ogata et al. (2009)
Africa	Soap rinsed fingers or stainless steel tweezers	Hexane	41	n.a.	4.49	0.91	n.a.	37.1	n.a.	n.a.	Ogata et al. (2009)
India	Soap rinsed fingers or stainless steel tweezers	Hexane	141	n.a.	29.8	15.8	n.a.	3.24	n.a.	n.a.	Ogata et al. (2009)
Australia	Soap rinsed fingers or stainless steel tweezers	Hexane	16	n.a.	6.69	0.57	n.a.	0.19	n.a.	n.a.	Ogata et al. (2009)
West Europe	Soap rinsed fingers or stainless steel tweezers	Hexane	169	n.a.	27.6	15.9	n.a.	1.12	n.a.	n.a.	Ogata et al. (2009)
South Atlantic Ocean	Manta trawl	DCM	590	n.a.	n.a.	n.a.	4.6	n.a.	n.a.	n.a.	Rochman et al. (2014)
Saint Helena British territory	Soap rinsed fingers or stainless steel tweezers	Hexane	7	n.a.	3.4	0.5	n.a.	19	n.a.	n.a.	Heskett et al. (2012)
Cocos Islands, Australia	Soap rinsed fingers or stainless steel tweezers	Hexane	6.5	n.a.	3.4	0.8	n.a.	1.7	n.a.	n.a.	Heskett et al. (2012)
Hawaii, USA	Soap rinsed fingers or stainless steel tweezers	Hexane	9.9	n.a.	3.4	0.5	n.a.	0.6	n.a.	n.a.	Heskett et al. (2012)
Island of Oahu, USA	Soap rinsed fingers or stainless steel tweezers	Hexane	1.5	n.a.	0.8	0.6	n.a.	<LOQ	n.a.	n.a.	Heskett et al. (2012)
Barbados, USA	Soap rinsed fingers or stainless steel tweezers	Hexane	1.7	n.a.	3.1	0.3	n.a.	<LOQ	n.a.	n.a.	Heskett et al. (2012)
Canary Islands, Spain	Soap-rinsed fingers or stainless steel tweezers	Hexane	9.0	n.a.	4.1	2.5	n.a.	0.6	n.a.	n.a.	Heskett et al. (2012)
North Pacific Central Gyre	Hand net, manta trawl	DCM	2856	14,459	336	126	n.a.	n.a.	n.a.	n.a.	Rios et al. (2010)

Table 8.2 Maximum concentrations of selected plastic additives in plastic pellets from field studies

Location	Sampling	Extraction	UV stabilizers (µg/g)	BPA (ng/g)	Antioxidants (µg/g)	NP (ng/g)	OP (ng/g)	References
Japanese coasts	Solvent rinsed stainless tweezers or by hand with PE gloves	Hexane	n.a.	n.a.	n.a.	16,000	n.a.	Mato et al. (2001)
Central Pacific Gyre, Pacific Ocean	Neuston net and solvent rinsed stainless steel tweezers	DCM	n.a.	283	n.a.	997	4	Hirai et al. (2011)
Japanese beaches	Neuston net and solvent rinsed stainless steel tweezers	DCM	n.a.	3.4	n.a.	1244	49	Hirai et al. (2011)
Tonkin Bay, Vietnam	Neuston net and solvent rinsed stainless steel tweezers	DCM	n.a.	263	n.a.	551	154	Hirai et al. (2011)
Seal Beach, California	Neuston net and solvent rinsed stainless steel tweezers	DCM	n.a.	26	n.a.	130	17	Hirai et al. (2011)
Marbella Beach, Costa Rica	Neuston net and solvent rinsed stainless steel tweezers	DCM	n.a.	730	n.a.	3936	14	Hirai et al. (2011)
9 Japanese beaches	Solvent rinsed stainless tweezers	DCM	n.a.	n.a.	n.a.	17,000	n.a.	Takada et al. (2006)
Japanese coast	Neuston net	N.S.	n.a.	<25	n.a.	n.s.	n.a.	Teuten et al. (2009)
Central Gyre	Neuston net	N.S.	n.a.	284	n.a.	2,660,000	n.a.	Teuten et al. (2009)
Island of Kauai, Hawaii, USA	Stainless steel tweezers	n-Hexane	1133	n.a.	n.a.	n.a.	n.a.	Tanaka et al. (2020)
Geoje, South Korea	N.S.	DCM	82	n.a.	1620	n.a.	n.a.	Rani et al. (2017)
South Atlantic Ocean	Manta trawl	DCM	n.a.	4.9	n.a.	n.s.	n.s.	Rochman et al. (2014)

steric, non-covalent, or covalent bonds (i.e. π - π interactions) (Brennecke et al. 2016; Hüffer and Hofmann 2016; Velzeboer et al. 2014; Wang et al. 2019).

Sorption is also largely influenced by the surface area to volume ratio of a plastic particle, which increases as the size decreases (Brennecke et al. 2016; Li et al. 2019; Ma et al. 2016; Teuten et al. 2009; Zhan et al. 2016; Zhang et al. 2018). When exposed to weathering, the plastic surface can be subjected to embrittlement and fragmentation, steps that increase the surface/volume and provide more space, a larger contact area, and new sorption sites for external molecules (Napper et al. 2015; Wang et al. 2018b). UV rays-induced weathering, or photo-oxidation, can also lead to chemical alterations and loss of hydrophobicity through the creation of new oxygen-rich functional groups (i.e. carbonyl moieties). Salinity and pH can also play a role in sorption mechanisms. When pH is above the point of zero charge (PZC) of the plastic polymer, it assumes a negative charge that could result in electrostatic repulsion between its surface and other anionic chemicals (Holmes et al. 2014; Li et al. 2019; Wang et al. 2015). Salinity, on the other hand, can either increase the partitioning of nonpolar compounds (salting out) or decrease that of polar molecules due to competition in the adsorption sites (Karapanagioti and Klontza 2008; Llorca et al. 2018; Wang et al. 2015; Zhan et al. 2016; Zuo et al. 2019). This can result in a difference in sorption capacities between freshwater and seawater environments. Exposure time, chain length, and temperature have also been observed as influential factors in hydrophobic partitioning (Engler 2012; Llorca et al. 2018; Mato et al. 2001; Takada et al. 2006; Zhan et al. 2016; Zhang et al. 2018).

Finally, it must be noted that new kinds of biodegradable plastic polymers are being designed and are expected to be more easily and fully degraded over a short time, thus reducing their potential harm. Despite this assumption, Zuo et al. (2019) recently indicated that those highly rubbery MP could become even stronger vectors of organic chemicals. Evaluation of toxicity of the alternative materials and experimental studies are needed to clarify the possible harm.

8.2.1 Polycyclic Aromatic Hydrocarbons (PAHs)

The presence of polycyclic aromatic hydrocarbons (PAHs) in the environment can be the result of three different processes: the incomplete combustion of organic material (pyrolytic origin), spillage discharge of crude oil (petrogenic origin), or the post-depositional transformation of biogenic precursors (diagenetic origin). The contribution of petrogenic over pyrogenic sources (and vice versa) can be manifested through the calculation of the ratio between lighter (2–3 rings) and heavier congeners (4–6 rings), with higher values of this parameter indicating a major contribution of fossil sources (low molecular weight congeners). High molecular weight PAHs are generally the ones detected at higher concentrations on plastic pellets in

the environment, implying combustion processes to be the main source of contamination (Gauquie et al. 2015; Rios et al. 2007). In a recent study, low molecular weight PAHs were mostly found on clearer materials, while high molecular weight PAHs were mostly detected on darker materials (Fisner et al. 2017). The colour of microplastics, along with their size and smell, is an important factor since some organisms may selectively feed on those pellets which resemble their prey (Chagnon et al. 2018; Hipfner et al. 2018; Ory et al. 2017; Savoca et al. 2016).

8.2.2 Polychlorinated Biphenyls (PCBs)

Polychlorinated biphenyls (PCBs) are a group of 209 lipophilic chemicals of a completely anthropogenic origin, which have been widely used from the 1930s to 1970s until they were banned because of their harmful nature.

The concentration of these pollutants in plastic pellets nowadays is mostly related to the presence of legacy PCBs in those industrialized countries that formerly used high amounts of them, which are still present in the environment due to their persistent nature (Ogata et al. 2009). When lower chlorinated congeners are detected inside an organism, their presence can be generally linked to the ingestion of contaminated MPs, since they would be more easily depleted along the trophic chain (Teuten et al. 2009). For this, only those highly substituted congeners would be more prone to be transferred and biomagnify in the trophic chain, and their exposure would then be mainly caused by prey ingestion (Yamashita et al. 2011). However, due to the high K_{ow} of highly chlorinated PCBs, they are more efficiently bound to the plastic and then less likely released from polymers (Colabuono et al. 2010). Their concentration is typically expressed as ICES-7 (Σ PCBs 28, 52, 101, 118, 138, 153, 180), which corresponds to the sum of seven indicator non-dioxin-like (NDL) congeners presenting the highest concentrations in technical mixtures and the environment (Webster et al. 2013). They are generally calculated as the sum of different congeners. The ones typically showing the highest concentrations are CB 118, 138, 153, 170, and 180 (Antunes et al. 2013; Colabuono et al. 2010; Gauquie et al. 2015; Mato et al. 2001; Yamashita et al. 2011). Congeners 138, 153, and 180 are the ones mostly detected in human serum and tissues (JECFA 2016).

Moreover, among PCBs, 12 congeners have a coplanar conformation (dioxin-like PCBs) that enables them to interact with xenobiotic receptors in the cell (i.e. aryl hydrocarbon receptor, AhR) and makes them able to interfere with the endocrine system, causing reproductive disorders among others (JECFA 2016; Pocar et al. 2005).

8.2.3 *Dichlorodiphenyltrichloroethane (DDT)*

Dichlorodiphenyltrichloroethane (DDT) is an organochlorine compound used as an insecticide. Total DDT concentration (tDDT) is usually expressed as the sum of DDT and its metabolites DDD and DDE (Antunes et al. 2013; Hirai et al. 2011; Ogata et al. 2009; Rios et al. 2007). When DDE/DDT ratio is very small, it indicates recent contamination. High levels of DDT in the environment and on plastics can be ascribed to their high production levels, especially by the USA, their intensive use as pesticides in agriculture, and as insecticides (JECFA 1961). While DDT production has been banned in some countries such as the USA since 1972, DDT and its derivatives are still in use as insecticides in some countries to prevent the spread of malaria.

8.2.4 *Polybrominated Diphenyl Ethers (PBDEs)*

Polybrominated diphenyl ethers (PBDEs) are a group of 209 anthropogenic chemicals produced and added to materials in order to improve their resistance to fire. They make up one of the major classes of brominated flame retardants (BFR). They are part of the group of lipophilic, persistent organic contaminants, can be accumulated through the food chain, and converted from one congener to another through metabolization. Their concentration in commercial products and, thus, their occurrence in the environment are not as high as those of PCBs. Industrial products are mostly made up of few congeners, so environmental data are available only for penta- hepta-, octa- and deca-BDEs, among which BDE-47 is the most abundant together with BDE-99, BDE-100, and BDE-209 (Darnerud et al. 2001). Their exposure is generally associated with hypothyroidism, since they can bind to thyroid hormone transporters, act as agonists/antagonists, or displace the bound hormones (Darnerud et al. 2001).

The congeners most commonly analysed to check contamination in feed and food are BDE-28, BDE-47, BDE-99, BDE-100, BDE-153, BDE-154, BDE-183, and BDE-209 (EFSA 2011). The log K_{ow} of these highly hydrophobic congeners ranges from 5.94 to more than 8 (Braekevelt et al. 2003). This class of POPs has among the greatest potential to cause harm to biological systems even at low concentrations (Abdelouahab et al. 2009; Carlson 1980; Fair et al. 2012).

8.2.5 *Hexachlorocyclohexanes (HCHs)*

Hexachlorocyclohexanes (HCHs) are long-range transported organochlorine pesticides that were mostly used either in agriculture or as insecticides. Several isomers can be present in the commercial mixtures, and when performing environmental

analyses, their concentration is expressed as ΣHCH ($\alpha\text{-HCH}$, $\beta\text{-HCH}$, $\gamma\text{-HCH}$, and $\delta\text{-HCH}$). $\gamma\text{-HCH}$, the major component of pesticides, was found as the most abundant isomer in plastic samples from Mozambique and South Africa (Ogata et al. 2009). $\text{Log } K_{ow}$ ($\gamma\text{-HCH} = 3.8$) for HCHs is lower than that for PCBs and DDTs, and they are thus supposed to partition the plastic pellets less than the other more hydrophobic chemicals (Mizukawa et al. 2013).

8.3 Desorption of Environmental Pollutants from Microplastics

New evidence implies a possible transfer of persistent, bioaccumulative, and toxic (PBT) pollutants into organisms from contaminated plastics (Avio et al. 2015; Browne et al. 2013; Chua et al. 2014; Engler 2012; Ryan et al. 1988; Tanaka et al. 2013; Wardrop et al. 2016; Yamashita et al. 2011). This would lead to an increased exposure to xenobiotic molecules once the plastic item is ingested.

Considering that the bond between a co-contaminant and the plastic substrate can be reversible, sorbed POPs will tend to desorb from the polymer into the water until equilibrium is finally attained (Andrady 2011). The desorption behaviour can be enhanced by the presence of some chemical surfactants, which are molecules able to increase the solubility of hydrophobic substances. These molecules possess an amphipathic nature, containing both hydrophilic and hydrophobic moieties, enabling them to interact with the two phases. Their mechanism of action consists of the lowering of aqueous phase polarity and the formation of micelles to include and contain the nonpolar chemicals. Among them, humic acids and linear alkyl benzene sulphonate, which can be found in the environment, have been found to significantly affect PCDD/DFs and PCBs leaching in some shredder residues and municipal solid waste (Sakai et al. 2000). In the gut, some digestive detergents are present and, through their surfactant properties, could induce the release of toxic substances from the ingested MPs, in a process that is negatively correlated with lipophilicity (Ahrens et al. 2001; Heinrich and Braunbeck 2019). This could suggest a higher degree of bioavailability.

8.3.1 Leaching of Additives from Microplastics

Many of the plastic additives (plasticizers, PBDEs flame retardants, antioxidants, and stabilizers) are not chemically bound to the polymer and can thus more easily migrate from the material. Only some reactive organic additives, such as some flame retardants, are polymerized with the plastic molecules becoming part of the polymer chain. All these chemicals are intentionally added during plastic manufacture in order to give plastics some specific features and improve their functional

characteristics (e.g. UV and thermal resistance, flexibility). These compound are mixed within the bulk matrix, and their leaching behaviour can be influenced and enhanced by external factors, such as changes in temperature or pH, plastic ageing, and the presence of surfactants (Suhrhooff and Scholz-Böttcher 2016; US EPA 1992; Wei et al. 2019).

8.4 Microplastics and Nanoplastics Occurrence in Foods

Microplastic pollution has become a potential food safety threat that is especially relevant for fishery products as well as other seafood products including table salt. Although microplastics have been reported in products such as honey and sugar (Liebezeit and Liebezeit 2013) or beer (Liebezeit and Liebezeit 2014), aquatic products and water seem to be the best-studied source of dietary intake of microplastics.

8.4.1 *Microplastics and Nanoplastics Occurrence in Fisheries and Aquaculture Products*

Ingestion of microplastics by aquatic organisms have been reported in numerous studies; indeed, microplastics have been found in 12 out of the 25 most important species and genera that contribute to global marine fisheries (FAO 2017).

Small concentrations of microplastics of around one to two particles per fish have been observed in many important commercial species such as sardine, mackerel, anchovy, herring, and sprat from the Pacific, Atlantic, and Indian oceans, as well as the Mediterranean Sea (GESAMP 2016; Lusher et al. 2016). Other aquatic species of local and regional relevance from marine and freshwater environments have been found to contain microplastic particles too (FAO 2017).

Most microplastics have been observed in the gastrointestinal tract of fish, where most particles seem to concentrate after ingestion. However, several studies in marine organisms have shown that smaller microplastics and nanoplastics could be translocated in other organs such as the liver, although the translocation pathways are not well understood yet (Collard et al. 2017).

Bivalve molluscs have also been found to contain microplastic particles. One of the best-studied are mussels, where their occurrence has been reported in Europe, North America, Brazil, and China. The lowest level of microplastic concentration in mussels (less than 0.5 particle/g) was observed in Europe and the highest concentration of microplastics in mussels was reported in China, amounting to 4 particles/g (EFSA 2016).

Microplastics have also been observed in crustaceans, such as shrimps and in 83% of Norway lobsters in coastal waters of the North Sea and the Irish Sea, with

average concentrations ranging from 0.03 to 1.92 particles/g in shrimp, mostly found in the digestive tract (Devriese et al. 2015; Murray and Cowie 2011).

An emerging threat to the consumption of fisheries and aquaculture products derives from the ability of microplastics to sorb environmental contaminants as persistent organic pollutants (POPs), metals, and pathogens, where its concentration can be several folds higher than in the water column. Humans can be exposed to these contaminants through the consumption of fisheries products especially bivalves and crustaceans, where the gastrointestinal tract is not removed. Another route of exposure might be the consumption of farmed species fed with contaminated fish or fishmeal (GESAMP 2016). Information on microplastics presence in different marine organisms (including commercial seafood species), sampling method/instrument, polymer type, and particle size are summarized in Table 8.3.

8.4.2 Microplastics and Nanoplastics Occurrence in Salt

Microplastics may contaminate table salt, as other sea products, and their occurrence has been reported in different countries such as Italy, Croatia, and Spain. The average microplastic content in Italy fluctuated from 1.57 (high-quality table salt) to 8.23 (low-quality table salt) particles/g, with a size range from 4 μm to 2100 μm . The results in Croatia fluctuated from and 27.13 (high-quality table salt) to 31.68 (low-quality table salt) particles/g with an average size range from 15 to 4628 μm (Renzi and Blašković 2018).

In Spain, microplastics were found in commercial table salt. The content went from 50 to 280 particles/kg, being PET the most frequently reported polymer, followed by PP and PE (Iñiguez et al. 2017).

8.4.3 Microplastics and Nanoplastics Occurrence in Water

The presence of microplastics has been reported in raw and treated drinking water. Depending on the water treatment, microplastic particles are found in different concentrations in water samples. Their average abundance was described from 1473 ± 34 to 3605 ± 497 particles/L in raw water and from 338 ± 76 to 628 ± 28 particles/L in treated water, depending on the treatment (Pivokonsky et al. 2018). A relevant finding is that some microplastic particles were reported to be down to the size of 1 μm and around 12 different microplastics compounds were identified, being PET, PP, and PE the most prominent polymers.

Table 8.3 Field studies on microplastic occurrence in fisheries and aquaculture products

Species	Particle amount	Sampling	Particle size	Polymers	Location	References
5 mesopelagic and 1 epipelagic fish species	1–83 items/fish (2.1 ± 5.78 items/fish)	Manta trawl	1–2.79 mm		North Pacific Central Gyre	Boerger et al. (2010)
10 species of fish (5 pelagic, 5 demersal)	1–15 items/fish (1.90 ± 0.10 items/fish)	Standard haul trawl	0.13 mm–14.3 mm	PA, LDPE, PS, RY, PEST	English Channel	Lusher et al. (2013)
<i>Mytilus edulis</i> <i>Crassostrea gigas</i>	0.36 ± 0.07 items/g ww 0.47 ± 0.16 items/g ww	Mussel farm, supermarket	5–10 μ m 11–20 μ m		North Sea, Atlantic Ocean	Van Cauwenberghe and Janssen (2014)
<i>Mytilus edulis</i>	Wild: 34 items/ ind 106–126 items/ ind Farmed: 75 items/ind, 178 items/ind	Collection at low tide, grocery store			Newfoundland, Nova Scotia, Canada	Mathalon and Hill (2014)
26 species of commercial fish	0–20 items/fish 1.40 ± 0.66 items/fish or 0.27 ± 0.63 items/fish	Stern trawlers	0.217–4.81 mm	Alkyd resin, PE, PP, RY, PA-6, PEST, acrylic,	Portuguese coast	Neves et al. (2015)
9 species of commercial bivalves	4.3–57.2 items/ ind 2.1–10.5 items/g ww	Shanghai fishery market	5 μ m–5 mm	PE, PET, PA	Coastal waters of China	Li et al. (2015)

Indonesia: 11 fish species USA: 12 fish species and <i>Crassostrea gigas</i>	Indonesia: 0–21 items/fish (1.4 ± 3.7 items/fish) USA: 0–10/fish 0–2/ oyster (0.5 ± 1.4 items/ind)	Fish markets, fishermen	Indonesia: 3.5 ± 1.1 mm USA: 6.3 ± 6.7 mm (fish) 5.5 ± 5.8 mm (oysters)	Sulawesi (Indonesia) and California (USA)	Rochman et al. (2015)
<i>Crangon crangon</i>	0.68 ± 0.55 items/g ww Max 1.92 ± 0.61 items/g ww 1.23 ± 0.99 items/ind	Shrimp trawl and beam trawl net	200–1000 µm	Channel area and southern part of the North Sea (France, Belgium, Netherlands, UK)	Devriese et al. (2015)
<i>Mytilus edulis</i> <i>Arenicola marina</i>	0.2 ± 0.3 items/g ww Max 1.1 items/g ww 1.2 ± 2.8 items/g Max 11.3 items/g ww	Randomly on breakwaters Bait-pump or shovel	20–90 µm 15–100 µm	LDPE, HDPE, PS French, Belgian, and Dutch North Sea coast	Van Cauwenberghe et al. (2015)
5 fish species (3 demersal and 2 pelagic)	0.03 ± 0.18 items/fish 0.19 ± 0.61 items/fish (pelagic)	Bottom/pelagic trawling	180 µm–50 cm	PE, PA, PP, PS, PET, PEST, PU North and Baltic seas	Rummel et al. (2016)

(continued)

Table 8.3 (continued)

Species	Particle amount	Sampling	Particle size	Polymers	Location	References
<i>Mytilus edulis</i>	1.5–7.6 items/ ind (4 items/ind) 0.9–4.6 items/g ww (2.2 items/g ww)	Tweezers at low tide, underwater with fishermen	< 250 µm Fibres: 33 µm–4.7 mm	CP, PET, PEST	Coastline of China	Li et al. (2016)
<i>Perna perna</i>	75% mussels contaminated with MPs	Byssus cut and removal	< 5000 µm		Santos estuary, São Paulo, Brazil	Santana et al. (2016)
<i>Saccostrea cucullata</i>	1.4–7.0 items/ ind 1.5–7.2 items/g ww	Intertidal zone	20–5000 µm (mostly <100 µm)	PET, PP, PE, PS, CP, PVC, PA, EPS	Pearl River estuary, South China	Li et al. (2018a)
<i>Penaeus monodon</i> <i>Metapenaeus</i> <i>monoceros</i>	6.60 ± 2 items/ ind 1.55–4.84 items/g ww (3.40 ± 1.23 items/g ww) 7.80 ± 2 items/ ind 2.17–4.88 items/g ww (3.87 ± 1.05 items/g ww)	Offshore shrimp trawlers and bag net fishing	1–5 mm 250–500 µm	PA-6, RY	Northern Bay of Bengal, Bangladesh	Hossain et al. (2020)

<i>Venerupis philippinarum</i>	0.07–5.47 items/g ww Wild: 8.4 ± 8.5 items/ind Max 12.7 ± 13.0 items/ind 0.9 ± 0.9 items/g Farmed: 11.3 ± 6.6 items/ind Max 15.4 ± 6.3 items/ind 1.7 ± 1.2 items/g Max 2.2 ± 0.8 items/g	0.5 m × 0.5 m quadrat, from farmed sites			Baynes sound, British Columbia	Davidson and Dudas (2016)
<i>Emerita analoga</i>	0.65 ± 1.64 items/ind Max 16 items/ind	Shovel, sand-coring tool		PP, polyacrylate, PEST, PE	Pacific coast of California	Horn et al. (2019)
<i>Lithodes santolla</i>	1–3 items/ind	Trap gears	3–>20 mm		Nassau Bay, Cape Horn, Chile	Andrade and Ovando (2017)

(continued)

Table 8.3 (continued)

Species	Particle amount	Sampling	Particle size	Polymers	Location	References
<i>Saccostrea forskalii</i> <i>Littoraria</i> sp. <i>Balanus amphitrite</i>	0.37 ± 0.03– 0.57 ± 0.22 items/g ww 0.17 ± 0.08– 0.23 ± 0.02 items/g ww 0.23 ± 0.10– 0.43 ± 0.33 items/g ww	Intertidal zone at low tide		PS, PET, PA	Upper gulf of Thailand	Thushari et al. (2017)
<i>Crassostrea gigas</i> <i>Gammarus</i> sp. <i>Mytilus edulis</i> <i>Littorina littorea</i>	87 items/g dw 11 items/g dw 105 items/g dw 20 items/g dw	From littoral zones	1–5000 µm (Mostly 1–300 µm)		Dutch coast	Leslie et al. (2013)
<i>Mytilus edulis</i>	0.04–0.81 items/g ww (0.37 items/g ww)	Department stores, manual sampling at groynes and quayside	200 µm–1500 µm		Belgium, Netherlands	De Witte et al. (2014)
<i>Chelon aurata</i> <i>Rutilus kutum</i>	2.95 ± 1.98 items/fish 1.66 ± 1.23 items/fish		1.94 ± 0.71 mm 1.77 ± 0.53 mm	PA	Southern Caspian Sea	Zakeri et al. (2020)
<i>Platichthys flesus</i> <i>Diplodus vulgaris</i> <i>Dicentrarchus labrax</i>	0.18 ± 0.55 items/fish 3.14 ± 3.25 items/fish 0.30 ± 0.61 items/fish	Beam trawl		PE, PP, PEST, PA-6, RY, PAN	Mondego estuary, Portugal	Bessa et al. (2018)

21 species of marine fish, 6 species of freshwater fish	1.1 ± 0.3–7.2 ± 2.8 items/fish 0.2 ± 0.1–17.2 ± 9.7 items/g	Fishery markets, fishermen	0.04–5 mm	CP, PET, PEST	Yangtze estuary, East China and South China Sea, Taihu lake	Jabeen et al. (2017)
<i>Boops boops</i>	2.47 ± 0.23–4.89 ± 0.45 items/fish	Bottom trawl nets and purse seine	1 mm–5 mm		Balearic Islands	Nadal et al. (2016)
6 marine fish species	1.62 ± 1.58–2.96 ± 5.21 items/fish (0–32 items/fish)	Hook and line	<5 mm		Texas Gulf Coast	Peters et al. (2017)
28 marine fish species	2.36 items/fish (1–35 items/fish)	Trawl net	656 ± 803 µm (range: 9.07–12074.11 µm)	Copolymers, alloys, PA, LDPE, PP, rubber	Mediterranean coast of Turkey	Güven et al. (2017)
26 marine fish species	0–3 items/fish	Tucker nets, fishermen	2.39 ± 0.28 mm (range: 1–3 mm)	PP, PE, PS, PVC, PAN	Saudi Arabian Red Sea coast	Baalkhuyur et al. (2018)
<i>Hoplosternum littorale</i>	3.6 items/fish (1–24 items/fish)	Fishermen	<1 mm–12 mm		Pajerut River, Northeast Brazil	Silva-Cavalcanti et al. (2017)
<i>Scomberomorus cavalla</i> <i>Rhizoprionodon lalandii</i>	2–6 items/fish 1–3 items/fish	Fishermen	2–5 mm 1–3 mm		Salvador, Northeastern Brazil	Miranda and de Carvalho-Souza (2016)
<i>Rastrelliger kanagurta</i> <i>Stolephorus waitei</i> <i>Chelon subviridis</i> <i>Johnius belangerii</i>	0–3 items /fish	Local markets		PP, PE, PS, PET, PA-6	Malaysia	Karami et al. (2017a)

(continued)

Table 8.3 (continued)

Species	Particle amount	Sampling	Particle size	Polymers	Location	References
<i>Alepes djedaba</i>	0.80 ± 0.12 items/g muscle	Fishmongers	<300 µm		Persian Gulf	Akhbarizadeh et al. (2018)
<i>Epinephelus coioides</i>	0.78 ± 0.22 items/g muscle					
<i>Sphyræna jello</i>	0.57 ± 0.17 items/g muscle					
<i>Platycephalus indicus</i>	1.85 ± 0.46 items/g muscle					
<i>Mytilus galloprovincialis</i>	1.7 ± 0.2–2.0 ± 0.2 items/ind	Hand collection, trawling	0.1–0.5 mm	PE, PP, PET, PS, PTFE	Northern Ionian Sea	Digka et al. (2018)
<i>Sardina pilchardus</i>	(1.9 ± 0.2 items/ind)					
<i>Pagellus erythrinus</i>	2.5 ± 0.3–5.3 ± 0.5 items/g ww					
<i>Mullus barbatus</i>	1.8 ± 0.2 items/fish					
	34.9 ± 7.9 items/g ww					
	1.9 ± 0.2 items/fish					
	27.8 ± 24.6 items/g ww					
	1.5 ± 0.3 items/fish					
	11.2 ± 2.8 items/g ww					

<i>Engraulis japonicus</i>	2.3 ± 2.5 items/ fish (0–15 items/ fish)	Fishing	783 ± 1020 µm (150–6830 µm)	PE, PP, PS, ethylene/ propylene copolymer, ethylene/propylene/diene terpolymer	Tokyo Bay	Tanaka and Takada (2016)
<i>Mugil cephalus</i>	Wild: 4.3 ± 14.5 items/fish Captive: 0.20 ± 0.48 items/fish	Fish markets, fish ponds	0.1–12 mm	PP, PE, PEST, PET, PA, PTFE	East coast of Hong Kong	Cheung et al. (2018)
<i>Platycephalus indicus</i> , <i>Saurida tumbil</i> , <i>Sillago</i> <i>silhama</i> , <i>Cynoglossus</i> <i>abbreviatus</i> , <i>Penaeus</i> <i>semisulcatus</i>	0.16–1.5 items/g	Trawl net	< 100 µm → 1000 µm		Musa Estuary, Persian Gulf	Abbasi et al. (2018)
<i>Crassostrea gigas</i> <i>Mytilus edulis</i> <i>Tapes philippinarum</i> <i>Patinopecten</i> <i>yessoensis</i>	0.07 ± 0.06 items/g (0–0.19 items/g) 0.12 ± 0.11 items/g (0–0.35 items/g) 0.34 ± 0.31 items/g (0.03–1.08 items/g) 0.08 ± 0.08 items/g (0.01–0.17 items/g) Overall: 0–1.08 items/g ww 0–2.8 items/ind	Fishery markets	100–200 µm	PE, PP, PS PEST, silicon, PA, PEVA, PET, PU, acrylic, PTFE, PVC, PPS, PVA, PBT, polyethyl acrylate styrene, polyepoxides, styrene/acrylonitrile, and polystyrene ethylene butylene styrene	Seoul, Gwangju, Busan	Cho et al. (2019)

(continued)

Table 8.3 (continued)

Species	Particle amount	Sampling	Particle size	Polymers	Location	References
<i>Cerithiidea cingulata</i> <i>Thais mutabilis</i> <i>Aniantis umbonella</i> <i>Aniantis purpuratus</i> <i>Pinctada radiata</i>	Clams and oyster: 0.2–2.2 items/g ww 3.9–6.9 items/ind Snails: 12.8–20.0 items/g ww 3.7–17.7 items/ind	Intertidal zone at low tide	10→5000 µm	PE, PET, PA	Persian Gulf	Naji et al. (2018)
<i>Mytilus edulis</i>	0.23 ± 0.09 items/g	Supermarket, collection from the wild	30–200 µm	PP, PE, PEST, ABS	Region Pays de la Loire, market in Nantes	Phuong et al. (2018)
<i>Mytilus galloprovincialis</i>	8.33 ± 3.58 items/g ww 3.6–12.4 items/ind 4.4–11.4 items/g ww	Local markets, collection from rocky bottom	0.75–6.00 mm (average: 1.15–2.29 mm)		Mediterranean Sea	Renzi et al. (2018)
<i>Modiolus modiolus</i> <i>Mytilus</i> spp.	0.086 ± 0.031 items/g ww 3.5 ± 1.29 items/ind 3.0 ± 0.9 items/g ww 3.2 ± 0.52 items/ind	Scuba diving, collection at low tide	0.2→2 mm	PET, PEST, poly (etherurethane)	Scottish coast	Catarino et al. (2018)

<i>Mytilus edulis</i>	0.7–2.9 items/g ww 1.1–6.4 items/ind	Collection from the coast	8 µm–4.7 mm	PEST, PP, PE, RY	UK	Jiana Li et al. (2018b)
<i>Mytilus galloprovincialis</i> <i>Mytilus edulis</i>	0.04 ± 0.09–0.34 ± 0.33 items/g ww (0.13 ± 0.14 items/g ww)	Hand collection, department stores			Europe	Vandermeersch et al. (2015)
<i>Pyropia</i> spp.	Packaged: 0.9–3.0 items/g dw (1.8 ± 0.7 items/g dw) Processed: 1.0–2.8 items/g dw (1.8 ± 0.6 items/g dw)	Local markets	Packaged: 0.11–4.97 mm Processed: 0.07–4.74 mm	PEST, RY, PP, PA, CP	China, Yellow Sea	Li et al. (2020)
<i>Mytilus edulis</i> <i>Perna viridis</i>	1.52–5.36 items/g ww 0.77–8.22 items/ind	Tweezers at low tide	0.25–1 mm	PET, PE, RY, PVC, PP	Coastal waters of China	Qu et al. (2018)

(continued)

Table 8.3 (continued)

Species	Particle amount	Sampling	Particle size	Polymers	Location	References
<i>Mytilus</i> spp.	0–6.9 items/ind (1.5 ± 2.3 items/ ind) 0–7.9 items/g ww (0.97 ± 2.61 items/g ww)	Hand collection, snorkelling, metal rake with net	70 µm–3870 µm	PET, PP, PE, PA, epoxy resin, PVC, PAN, styrene acrylonitrile, EVA, soloprene plastomer	Norway	Bråte et al. (2018)
<i>Mytilus</i> <i>galloprovincialis</i> <i>Chlamys farreii</i>	1.9–9.6 items/ ind 2.0–12.8 items/g ww 5.2–19.4 items/ ind 3.2–7.1 items/g ww	Fishery markets, coastal sampling	25 µm–5 mm	CP, PP, PTFE	China	DING et al. (2018)

8.5 Risk Profiling of Microplastics in Fisheries and Aquaculture Products

8.5.1 *Microplastics Dietary Intake*

The dietary intake of microplastics in foods depends on consumption habits, especially when dealing with fisheries and aquaculture products that mainly accumulate microplastics in the gastrointestinal tract, which minimizes the direct exposure to these particles when aquatic products are degutted. Previous studies and reviews have provided some estimates of human intake of microplastic particles through the consumption of different food commodities, and their results are illustrated in Table 8.4.

Exposure can be higher through consumption of small aquatic species such as crustaceans, echinoderms, bivalves, and small-sized fish that are commonly eaten whole. Microplastics have been also found in the muscle of canned, dried, and fresh commercial fish species, suggesting that evisceration may not be a completely efficient process, but the number of particles found in muscle is relatively low compared to the number of particles in the gastrointestinal tract (Abbasi et al. 2018; Akhbarizadeh et al. 2018; Karami et al. 2018, 2017a). In crustaceans such as shrimps, peeling practices that remove most of the digestive tract, the head, and the gills will reduce the exposure to plastics, as these parts are estimated to contain 90 percent of the microplastic particles (Devriese et al. 2015).

In Europe, human exposure to microplastics resulting from the consumption of bivalves may account for 1800 to 11,000 particles/year (Van Cauwenberghhe and Janssen 2014). Mussels are among the most consumed bivalve molluscs, and the occurrence of microplastics in these products has been reported in several studies (EFSA 2016).

8.5.2 *Microplastics Uptake and Toxicity*

The physical characteristics of microplastic particles (size, shape), as well as the chemical characteristics, in addition to leaching of additives and pollutants or transport of pathogens, are the main factors for estimating the impact of these particles on food safety, but literature at this respect is indeed limited, and evidence of a possible transfer through the diet are lacking, as are the possible consequences.

Some studies have shown that small plastic particles are able to cross human placenta (Ragusa et al. 2021; Wick et al. 2010), which suggests a possible systemic increase in exposure as their size decreases (De Jong et al. 2008). The most frequent route of uptake would be absorption from the gut epithelium into the lymphatic system (Hussain et al. 2001). Absorption from the intestine is known to be very limited, and it should occur only for those microparticles <150 μm (EFSA 2016; FAO 2017). According to their size, particles are supposed to face different densities

Table 8.4 Previous estimates of exposure to microplastics through the consumption of seafood, water, and salt

Food	Reference intake	Derived MP intake	Country	References
Molluscs	72.1 g/day (top consumers) 11.8 g/day (minor consumers)	11,000 MP/year (top consumers) 1800 MP/year (minor consumers)	Europe	Van Cauwenberghe and Janssen (2014)
Fish muscle	300 g/week (adults) 50 g/week (children)	169–555 MP/week (adults) 28–92 MP/week (children)	Iran	Akhbarizadeh et al. (2018)
Bivalves	3.01 g/day	212 MP/year	South Korea	Cho et al. (2019)
Shellfish	4.03 g/day	283 MP/year	South Korea	Cho et al. (2019)
Mussels	82 g/year	123 MP/year	UK	Catarino et al. (2018)
Mussels	3.08 kg/year	4620 MP/year	Spain/ France/ Belgium	Catarino et al. (2018)
Mussels	225 g	7 µg 0.1 µg/kg bw/day	Globally	EFSA (2016) and FAO (2017)
Fish Crustaceans Molluscs	15.21 kg/year 2.06 kg/year 2.65 kg/year	31–8323 MP/year 206–17,716 MP/year 0–27,825 MP/year	Globally Globally Globally	Danopoulos et al. (2020), Danopoulos et al. (2020) and Danopoulos et al. (2020)
Water	2 l	85 µg/day 1.4 µg/kg bw/day	Globally	WHO (2019)
Water	2.2 l/day (women) 3 l/day (men)	4400 MP/year (women) 5800 MP/year (men)	Globally	Kosuth et al. (2018)
Salt	5 g/day	1000 MP/year	China	Yang et al. (2015)
Salt	3.95 g/day	37 MP/year	Globally	Karami et al. (2017b)
Salt	5 g/day	40.6–1085.2 MP/year	Italy	Renzi and Blašković (2018)
Salt	14.8–18.01 g/day	64–302 MP/year	Turkey	Gündoğdu (2018)
Salt	2.3 g/day	40–680 MP/year	Globally	Kosuth et al. (2018)
Salt	10.06 g/day	0–42,600 MP/year (average 3000)	Globally	Kim et al. (2018)
Salt	5 g/day	510 MP/year	Spain	Iñiguez et al. (2017)
Salt	5 g/day	117 µg/year	India	Seth and Shrivastav (2018)

inside the body, being either phagocytosed by macrophage ($> 0.5 \mu\text{m}$), endocytosed ($<0.5 \mu\text{m}$), cleared through splenic filtration ($>0.2 \mu\text{m}$), or eliminated via kidney filtration ($<10 \text{ nm}$) (Monti et al. 2015; Yoo et al. 2011). Nevertheless, it has to be underlined that microplastics in the lymph will be excreted mostly through the spleen or with faeces through bile clearance in the liver if present in the blood (Yoo et al. 2011).

Among the possible adverse effects caused by microplastic exposure, oxidative stress and alteration of the immune function, possibly leading to immune depression, are the most likely to occur (Petit et al. 2002; Schirinzi et al. 2017). This is because some of the particles can be taken up in the lymphatic system by phagocytic cells, such as macrophages, as mentioned before.

Furthermore, it has to be underlined that PVC could cause additional toxicity because of the possible leaching of vinyl chloride, an extremely toxic monomer, classified as carcinogenic, that is used in the production process of this material and makes up 50 to 100% of the polymer by weight (Lithner et al. 2011). The toxicity of plastic is likely due to the unreacted residual monomers that constitute this material, as they may induce genotoxic effects. Lithner et al. (2011) have elaborated a hazard ranking of monomers and additives that are present in plastic polymers, indicating the hazard for human health. PVC is also the type of polymer that requires the highest amount of additives followed by PP, PE, and styrenics.

In addition to possible adverse health consequences of the polymers itself, the combined exposure to plastic additives and associated co-contaminants adsorbed by the surrounding environment must be taken into consideration. Many additives, such as BPA, phthalates, nonylphenols, and PBDE, are known to have an impact in the organism they enter in contact with, mostly through a mechanism of endocrine disruption. Besides, the main harm derives from the fact that these molecules, which give plastic some specific characteristics, are not strongly bound to the polymer matrix and tend to easily leach from it.

On the other hand, persistent organic pollutants (POPs) that interact and become associated with microplastic particles could not only impact and alter endocrine functions in the organism (i.e. PCBs, PBDEs) but also promote carcinogenicity (PAH). These outcomes mainly arise from the interaction with intracellular receptors and gene expression induction (JECFA 2016; Pocar et al. 2005). In consonance with the strength and type of interaction they have with the polymer's matrix, they could be released at different degrees.

The toxicity of dioxins and dioxin-like PCBs can be expressed through their corresponding toxic equivalency factor (TEF), implemented in the 1980s (Barnes 1991; Safe et al. 1985; Safe 1986) and later revised in 2005 (Van den Berg et al. 2006). This parameter expresses the relative toxicity of a compound in respect of a standard compound of known toxicity, the 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD, TEF = 1). The toxic equivalency quotient (TEQ) is a value resulting from the sum of the weighted concentrations of each chemical multiplied by their TEF. Maximum uptake levels have been set for dioxin-like PCBs and dioxins in 2006 (European Commission 2006).

Allowable concentrations are estimated as provisional maximum tolerable daily intake (PTDI), provisional tolerable weekly intake (PTWI), and provisional tolerable monthly intake (PTMI) and can be expressed as WHO-TEQ. Their values are established during the hazard characterization step, by evaluating the no observed adverse effect level (NOAEL), and indicate the amount of a chemical that can be ingested daily, weekly, or monthly over a lifetime without adverse health consequences (JECFA 1995). In 2017, the FAO conducted an exposure exercise that considered mussels as the target species of greatest interest because of their high consumption and because it is consumed with the viscera, where microplastics tend to be located. The exposure assessment took into consideration the highest concentration of microplastics in mussels, which was reported at 4 particles/g in China (EFSA 2016). According to CIFOcOss, the highest reported consumption of mussels corresponded to the Belgian elderly (P95) and was estimated to be 250 g per day per person. Considering that the highest concentration of microplastics reported in mussels was 4 particles/g and the highest consumption was 250 g of mussels, it was estimated that a portion of mussels could contain up to 1000 microplastic particles, i.e. around 9 μg , depending on the volume and density of the particles.

The exposure assessment on microplastics in mussels confirmed that the intake of microplastics per day was 0.15 $\mu\text{g}/\text{kg}$ for a person of 60 kg. These values were selected in order to describe the worst-case scenario and cover all the populations, including those at the highest risk.

Although the exposure assessment for mussels was carried out, the tolerable daily intake (TDI) for microplastic particles and most of its compounds has not yet been established; therefore, it is not possible to determine if this level of exposure is safe. However, the TDI for some plastic additives such as phthalates, BPA, alkylphenols, and brominated flame retardants, as well as associated sorbed contaminants such as PCBs, PAHs, and DDT, has been established, and the exposure assessment estimation shows that the level of the compounds present in microplastics from mussels was significantly lower than the TDI. Therefore, based on these assumptions, the intake of associated chemicals from ingested plastics via seafood consumption is minor compared to the total intake from the diet (FAO 2017).

As a complement, an exposure assessment exercise of microplastics in shellfish has been carried out with updated information from the current literature and also taking into account other contaminants that were not included in the previous estimates (e.g. HCHs). Procedures and outcomes are described in the following paragraph.

8.5.3 Case Study: Exposure Assessment of Microplastic Additives and Associated Sorbed Contaminants via Shellfish Consumption

Since plastic particles are more likely found in the intestinal tract, the consumption of all those seafood organisms whose GI is not removed can be the main route of exposure. Almost all shellfish are eaten whole, and in the present study, the attention is focused on mussels, shrimps, prawns, clams, and oysters. Data on daily seafood consumption were taken from the Chronic Individual Food Consumption Database (CIFOCoSS) of the WHO. In order to perform an exposure assessment, countries reporting the highest consumption levels of shellfish were selected. These countries were China and Finland (mussels, clams, oysters) and the Netherlands (shrimps and prawns). Only the P95 consumers, meaning top consumers of these products from each country, were considered. The adults and elderly category presented the highest daily consumption levels (g/day).

Information about microplastic load in shellfish was taken from scientific papers, only considering the highest detected amount, described in Table 8.3. Plastic materials were assumed to be spherical with a diameter of 25 μm , which was among the most common plastic sizes found in a study by Van Cauwenberghe and Janssen (2014) in mussels. With this information, it was possible to calculate the volume of plastic particles. The density of polymers was taken from scientific papers and reviews, and only maximum values were considered (Andrady 2017; Avio et al. 2017). By knowing the volume and the density, it was then possible to determine the weight of each plastic polymer.

Data on microplastic contamination in bivalve molluscs were taken for clams (*Scapharca subcrenata*), mussels (*Mytilus galloprovincialis*), and oysters (*Saccostrea cucullata*) from three studies on commercial species (Table 8.3). For these species, the highest plastic load was estimated to be 10.5, 12.8, and 7.2 particles/g wet weight, respectively (DING et al. 2018; Li et al. 2015, 2018a). In addition to these, information on shrimp contamination were taken from a recent study, where up to 4.88 particles/g tissue (wet weight) were found in commercial brown shrimp (*Metapenaeus monoceros*) (Hossain et al. 2020).

Indirect exposure to microplastic-bound pollutants was estimated by using the highest reported contamination levels of plastic particles found in field studies (Tables 8.1 and 8.2). These data were used to provide an estimate of exposure to microplastic-bound contaminants through the consumption of shellfish.

The exposure assessment estimation consists of several steps. First, the maximum load of microplastics in shellfish (particles/g) was multiplied by the daily dietary intake of each commodity, or consumer P95 (g/day), thus measuring the total number of particles ingested every day. This result was then multiplied by the weight of each polymeric particle derived before, in order to obtain the estimated consumption of plastic per day (g/day). Then, this value was multiplied by the maximum concentration of each contaminant (ng/g) reported in field studies. With these estimations, it was possible to establish the daily intake of pollutants (ng/day)

which, converted in picograms (pg) and divided by the average body weight of an adult (60 kg), would finally provide information on the daily dietary exposure to environmental contaminants mediated by microplastics (pg/kg bw/day). This information is presented by commodity and polymer type in Table 8.5. The calculations were made based on the assumption that these chemicals were completely released from the microplastic particles.

The highest exposure to environmental contaminants associated with microplastics and plastic additives, as estimated in the present case study, could derive from the consumption of oysters, followed by mussels, clams, and finally shrimps and prawns (Table 8.5). This could cause concern because the three most contaminated shellfish groups are also the ones that are eaten whole, whereas shrimps are peeled, and most microplastics are removed.

Overall, nonylphenols are the group of xenobiotics that presented the highest microplastic-mediated exposure concentrations, which ranged from 0.25 ng/kg bw/day in PP and LDPE-contaminated shrimps to 2.33 ng/kg bw/day in PVC-contaminated mussels. These high levels can most likely be related to the use of this additive in the manufacture of plastics. After NPs, PAHs are the class of environmental pollutants that could bring out the highest harm through shellfish consumption, with a concentration ranging from 4.15 pg/kg bw/day in PP-contaminated shrimps to 39.26 pg/kg bw/day in PVC-contaminated mussels. Contrariwise, according to the present results, HCHs could be classified as the compounds whose sorbed concentration levels on microplastics could pose the least concern. In fact, the daily dietary exposure to HCHs varied from 0.00 pg/kg bw/day to 0.03 pg/kg bw/day, at maximum.

The results of the exercise demonstrate that polypropylene (PP) is the polymer that might raise the least concern when analysing microplastic-mediated exposure to xenobiotics, while PVC might be the most hazardous. Despite this, the dietary exposure to environmental contaminants on ingested plastics, as calculated in the present case study, can be considered negligible, compared to other sources. Also, there is still not enough clarity on tissue transfer dynamics and concentration of chemicals associated with microplastics, which can be influenced by factors as a fugacity gradient, so this can be considered as an estimate. In conclusion, when considering the outcomes of the present exercise, it is important to also keep into account the physical-chemical properties of each polymer type, as they could also influence the sorption/desorption of chemicals and then their concentration on the microplastics.

In addition to the previous analysis, China and Finland seafood consumption values were used to perform an estimate of the total dietary exposure to organic pollutants through shellfish consumption (Table 8.6). These two countries were chosen because they represent the ones with the highest shellfish consumption levels.

When only these two countries are considered in the analysis, the exposure levels to microplastic-associated contaminants seem to be higher. PVC is again the plastic polymer that can apparently cause the most significant exposure to environmental contaminants in organisms after consumption, followed by PET, PA, PS, HDPE, LDPE, and finally PP. Anyway, PVC is generally reported to sorb pollutants to a

Table 8.5 Overall maximum dietary exposure (pg/kg/day) of MP additives and associated sorbed pollutants resulting from the consumption of the four shellfish groups in the Netherlands, Finland, and China (age class: adults and elderly). Results are categorized by polymer type

Polymer	Seafood	MP-PAH (pg/kg bw/day)	MP-PCB (pg/kg bw/day)	MP-DDT (pg/kg bw/day)	MP-PBDE (pg/kg bw/ day)	MP-HCH (pg/kg bw/day)	MP-BPA (pg/kg bw/day)	MP- NP (ng/ kg bw/ day)
HDPE	Shrimps and prawns	4.42	0.28	0.11	0.98	0.00	0.07	0.26
	Mussels	26.26	1.67	0.64	5.81	0.02	0.43	1.56
	Oysters	19.20	1.22	0.47	4.25	0.02	0.31	1.14
	Clams	6.94	0.44	0.17	1.53	0.01	0.11	0.41
LDPE	Shrimps and prawns	4.24	0.27	0.10	0.94	0.00	0.07	0.25
	Mussels	25.18	1.61	0.62	5.57	0.02	0.41	1.50
	Oysters	18.41	1.17	0.45	4.07	0.02	0.30	1.09
	Clams	6.65	0.42	0.16	1.47	0.01	0.11	0.39
PA	Shrimps and prawns	5.24	0.33	0.13	1.16	0.00	0.09	0.31
	Mussels	31.14	1.98	0.76	6.89	0.03	0.51	1.85
	Oysters	22.76	1.45	0.56	5.03	0.02	0.37	1.35
	Clams	8.23	0.52	0.20	1.82	0.01	0.13	0.49
PS	Shrimps and prawns	5.06	0.32	0.12	1.12	0.00	0.08	0.30
	Mussels	30.05	1.92	0.74	6.65	0.02	0.49	1.78
	Oysters	21.97	1.40	0.54	4.86	0.02	0.36	1.30
	Clams	7.94	0.51	0.19	1.76	0.01	0.13	0.47
PP	Shrimps and prawns	4.15	0.26	0.10	0.92	0.00	0.07	0.25
	Mussels	26.64	1.57	0.60	5.45	0.02	0.40	1.46
	Oysters	18.01	1.15	0.44	3.98	0.01	0.29	1.07
	Clams	6.51	0.41	0.16	1.44	0.01	0.11	0.39
PVC	Shrimps and prawns	6.61	0.42	0.16	1.46	0.01	0.11	0.39
	Mussels	39.26	2.50	0.96	8.68	0.03	0.64	2.33
	Oysters	28.70	1.83	0.70	6.35	0.02	0.47	1.70
	Clams	10.37	0.66	0.25	2.29	0.01	0.17	0.62

(continued)

Table 8.5 (continued)

Polymer	Seafood	MP-PAH (pg/kg bw/day)	MP-PCB (pg/kg bw/day)	MP-DDT (pg/kg bw/day)	MP-PBDE (pg/kg bw/ day)	MP-HCH (pg/kg bw/day)	MP-BPA (pg/kg bw/day)	MP- NP (ng/ kg bw/ day)
PET	Shrimps and prawns	6.38	0.41	0.16	1.41	0.01	0.10	0.38
	Mussels	37.90	2.42	0.93	8.38	0.03	0.62	2.25
	Oysters	27.71	1.77	0.68	6.13	0.02	0.45	1.65
	Clams	10.01	0.64	0.25	2.21	0.01	0.16	0.59

Table 8.6 Overall dietary exposure to MP additives and associated sorbed pollutants in Finland and China (age class: adults and elderly) resulting from shellfish consumption. Data on food consumption were taken from WHO CIFOCCOs. Results are presented as overall exposure concentrations resulting from the combination of the four shellfish groups (mussels, oysters, clams, and shrimps and prawns), categorized by polymer type

Polymer	Country	MP-PAH (pg/kg bw/day)	MP-PCB (pg/kg bw/day)	MP-DDT (pg/kg bw/day)	MP-PBDE (pg/kg bw/ day)	MP-HCH (pg/kg bw/day)	MP-BPA (pg/kg bw/day)	MP- NP (ng/ kg bw/ day)
HDPE	China	41.52	2.65	1.02	9.18	0.03	0.68	2.47
	Finland	22.43	1.43	0.55	4.96	0.02	0.37	1.33
LDPE	China	39.81	2.54	0.98	8.80	0.03	0.65	2.36
	Finland	21.51	1.37	0.53	4.76	0.02	0.35	1.28
PA	China	49.22	3.14	1.21	10.89	0.04	0.80	2.92
	Finland	26.60	1.70	0.65	5.88	0.02	0.43	1.58
PS	China	47.51	3.03	1.17	10.51	0.04	0.77	2.82
	Finland	25.67	1.64	0.63	5.68	0.02	0.42	1.52
PP	China	38.95	2.48	0.96	8.62	0.03	0.63	2.31
	Finland	21.05	1.34	0.52	4.65	0.02	0.34	1.25
PVC	China	62.06	3.96	1.52	13.73	0.05	1.01	3.68
	Finland	33.53	2.14	0.82	7.42	0.03	0.55	1.99
PET	China	59.92	3.82	1.47	13.25	0.05	0.98	3.56
	Finland	32.38	2.06	0.79	7.16	0.03	0.53	1.92

lower amount compared to rubbery plastics and especially to PE. Our calculations, in fact, analyse the sorptive capacities of plastic materials by only taking into consideration their densities and not their physico-chemical properties. With respect to this, it should be noted that cooking and food processing can sometimes lead to physicochemical changes in the plastic particles. A recent study has measured a reduction of approximately 14% of microplastics in cooked mussels, with also a

possible size reduction of the particles (Renzi et al. 2018). Moreover, high temperatures could also enhance the release of chemical compounds from microplastics (Bach et al. 2013).

Finally, it is now possible to compare the results of the exposure assessment of associated chemicals from ingested plastics via seafood consumption (Table 8.5; Table 8.6) with the no observed effect levels (NOELs) and no observed adverse effect levels (NOAELs) established by international expert committees such as the Joint FAO/WHO Expert Committee on Food Additives (JECFA) or EFSA (Table 8.7). The purpose of this last step is to check whether the highest load of microplastic-bound pollutants could lead to a significant threat to humans after the consumption of shellfish or not.

The NOEL values are not indicative of the tolerable daily intake (TDI) of contaminants but can provide useful information on the threshold of toxicological concern. In the present case study, the dietary exposures to MP-bound pollutant concentrations were in the order of nanograms (ng/kg bw/day), well below the NOELs set by the literature.

PAHs can be metabolized, resulting in genotoxic effects and carcinogenicity, especially the ones presenting a higher number of aromatic rings (Scientific Committee on Food 2002b). For this reason, no TDI can be estimated. Mean and high daily intake in adults have been estimated to be 4 and 10 ng/kg bw, with children exposure being more than twofolds higher (JECFA 2006a). Benzo(a)pyrene can be used as a marker of PAH contamination in food, but the sum of benzo[a]pyrene and chrysene (PAH2); the sum of benzo[a]pyrene, chrysene, benz[a]anthracene, and benzo[b]fluoranthene (PAH4); and the sum of benzo[a]pyrene, benz[a]anthracene, benzo[b]fluoranthene, benzo[k]fluoranthene, benzo[ghi]perylene, chrysene, dibenz[a,h]anthracene, and indeno[1,2,3-cd]pyrene (PAH8) have been suggested as

Table 8.7 No observed effect levels (NOELs), benchmark dose lower confidence limit (BMDL), no observed adverse effect levels (NOAELs), allowable daily intake (ADI), and tolerable daily intake (TDI) values established by international authorities

Chemical	NOEL /NOAEL/BMDL	ADI/TDI	Source
PCBs	0.04 mg/kg bw/day	20 ng/kg bw	Faroon et al. (2003) and JECFA (1990)
PAHs	0.17 (PAH2), 0.34 (PAH4), 0.49 (PAH8) mg/kg bw	Not established, carcinogen	EFSA (2008)
Benzo(a)pyrene	100 µg/kg bw	Not established, carcinogen	JECFA (2006b)
γ-HCH	0.47 mg/kg bw/day	0–0.005 mg/kg bw	JECFA (2002)
DDT	1 mg/kg bw/day	0.01 mg/kg bw	JECFA (2001)
PBDEs	Not established	Not established	JECFA (2006c)
BPA	5 mg/kg bw/day	4 µg/kg bw	EFSA (2015) and Scientific Committee on Food (2002a)
NP	15 mg/kg/day	5 µg/kg bw	Bontje et al. (2004) and Nielsen et al. (2000)

more suitable alternatives, with the last two parameters being the most appropriate markers for genotoxic and carcinogenic PAHs in food commodities (EFSA 2008).

For DDT, the no observed effect level (NOEL) and provisional tolerable daily intake (PTDI) are 1 mg/kg bw/day and 0–0.01 mg/kg bw/day, respectively, with the highest observed average intake of 0.68 mg/man/day (JECFA 1961).

The tolerable daily intake (TDI) temporarily established for BPA in foodstuff was set as 4 µg/kg bw/day (EFSA 2015), and the daily intake in adults in Europe has been estimated up to 1.5 µg/kg bw (Scientific Committee on Food 2002a). As it concerns intake for nonylphenol, the TDI was estimated at 5 µg/kg bw/day by the Danish Institute of Safety and Toxicology (Nielsen et al. 2000). It can be concluded that the levels of microplastic-bound contaminants found in the selected commodities (shrimp, prawns, clams, oyster, and mussels) are below the ones reported in Table 8.7. Based on these estimation values, it could be assumed that the transfer of environmental pollutants and additives mediated by microplastic particles in shellfish is negligible. In fact, the contribution made by other sources such as the ingestion of contaminated prey items and subsequent transfer of POPs from food to organism supplies most of the contaminant burden (FAO 2017). However, emerging additives from MP should be further explored, and plasticizer additives such as phthalates and organophosphate flame retardants have not yet been investigated.

8.5.4 *Limitations for Food Safety Risk Assessment*

The fate of plastic in the human body and its possible food safety impact are unknown. Although it is thought that the particles below 1.5 µm can penetrate into the capillaries of the organs, while larger particles will be excreted (Yoo et al. 2011), there are many knowledge gaps such as toxicological data of commonly ingested plastics and its compounds.

The best-studied dietary sources of microplastics are fisheries and aquaculture products, which are important food commodities in certain areas. However, the toxicity of most plastic monomers, polymers, and additives present in microplastics has never been evaluated by relevant international expert scientific committees such as the JECFA. International expert committees such as the JECFA are key to evaluate the potential toxicity, considering newly generated scientific data and establishing the basis for the risk analysis exercises (risk assessment, risk management, and risk communication (Fig. 8.1)).

In order to perform a proper exposure and then risk assessment of plastic particles, and plastic as a vector of additives and associated sorbed pollutants, researchers should develop new techniques to better understand toxicity and transfer mechanisms. In addition to that, improvement should be made to detect smaller plastic fragments, especially those in the range of the nanoparticles, which are not much studied. These are, in fact, the ones that could mostly enhance negative consequences on the organism both because of their ability to cross biological barriers and their higher sorption capacity.

Fig. 8.1 FAO/WHO risk analysis framework. (Adapted from FAO 2017)

8.6 Research Gaps

Although most microplastics are found in the gastrointestinal tract of aquatic animals, there is a limited understanding of the presence of microplastics and specially nanoplastics in edible parts of fisheries and aquaculture products. This mainly occurs because there are no standard methodologies to analyse plastic particles in foods. While many researchers are developing analytical methods, there are still research gaps:

- Standard tissue digestion and polymer identification and quantification protocols should be determined.
- Future research should focus more on nanoplastics, which may also be the ones eliciting the highest exposure due to their size.
- As some particles have been found in the edible tissues of fish, future studies should also include muscle analysis.
- More studies should focus their attention on other food commodities that could be contaminated by microplastics, such as table salt or seaweed.
- More investigations on additive leaching and contaminant desorption processes under gut conditions should be carried out, to understand the exposure to xenobiotic chemicals through plastic ingestion.

Data that would allow a food safety risk assessment is limited and the consequences of microplastic exposure through diet are poorly understood. Most studies have investigated the effects of the inhalation of plastic particles, and very few of them analysed the consequences of a dietary uptake. As it concerns the risk assessment of microplastics, the following research gaps were identified:

- Chronic exposure to microplastics and nanoplastics in humans via ingestion, inhalation and, skin contact should be carried out.
- Environmentally relevant concentrations of plastics, their additives, and sorbed pollutants should be used in dose-response assays for a better understanding of their toxicity.
- The exposure to microorganisms present on plastic debris should be evaluated, as some pathogens might also be found on them.
- Consequences of the dietary intake of different concentrations of plastic additives, monomers, and contaminants should be clarified, as many of these chemicals might trigger response mechanisms even at low doses.
- Changes in the structure and chemical composition of microplastics and nanoplastics in foods while processing or cooking, as well as their interaction with the food matrix, should be further studied.
- Once the consequences of microplastic exposure through diets are clarified, mechanisms of control should be put in place.

8.7 Conclusions

Micro- and nanoscale plastic particles are widely distributed in the aquatic environment, and aquatic animals are exposed to them, which results in the presence of microplastics and nanoplastics in fisheries and aquaculture commodities.

Most microplastics and nanoplastics are found in the gastrointestinal tract of aquatic species, and evisceration could lead to a substantial decrease in the exposure to plastics in the final consumer. Nevertheless, there are fisheries and aquaculture commodities such as small fish, crustaceans, and bivalve mollusc that are commonly consumed whole, so the exposure through these products is higher.

There is evidence of a sorptive behaviour of microplastics and nanoplastics, which has been observed both in the laboratory and in the field. Because of their hydrophobicity and high surface/volume, plastic polymers can concentrate organic contaminants by several folds compared to the water column and might also host biofilm-forming microorganisms on their surface.

Microplastic contaminants and additives added during the manufacturing process to add specific features of the final product have been found to have adverse effects on humans and animals, mostly related to carcinogenicity and reproductive toxicity.

Data on human dietary intake of microplastics are very scarce. An exposure assessment was carried out to evaluate human exposure to environmental contaminants and plastic additives through microplastic-contaminated seafood. The worst-case estimate of the exposure to contaminants through seafood indicates the contribution of microplastic contaminants, and additives through fisheries and aquaculture products are negligible compared to other sources. Besides, the exposure scenario assumed the complete release of contaminants and additives once in the

gut, which is unlikely. Moreover, even in this case, the levels of xenobiotics were several orders of magnitude lower than threshold values (NOAEL) indicated by international authorities.

Microplastic dietary intake for other food commodities such as salt or water should be carried out to understand the overall exposure through food and open the possibility for food safety risk assessment.

Literature on the actual effect of microplastics and nanoplastics in humans is extremely poor, and more research is needed to understand the toxicity of the most common polymers and plastic additives, as well as their mixture.

8.8 Glossary

8.8.1 *Microplastics and Nanoplastics Definition*

Recently, many studies have focused their attention on the possible harm caused by micro- and nano-sized plastic particles on human and animal health. The size of the particles seems to be one of the main aspects to possibly pose a food safety threat.

There is an ongoing debate about how to define micro- and nanoplastic particles. One of the most accepted definition describes them as plastic particles consisting of a heterogeneous mixture of different shaped materials in the range from 0.1 μm to 5000 μm in their longest dimensions (EFSA 2016; Lusher et al. 2017), while nanoplastics are defined as plastic particles whose size ranges from 0.001 μm to 0.1 μm (Klaine et al. 2012).

Plastics can also be classified according to the process that generated them. Primary microplastics are intentionally produced (e.g. plastic and/or cosmetics manufacture), while secondary microplastics are the result of fragmentation of larger materials are discharged into the environment (GESAMP 2015).

8.8.2 *Microplastics and Nanoplastics Composition*

Depending on the intended use, polymers with different physical and chemical properties can be mixed. Additionally, additives such as plasticizers, flame retardants, colourants, or antioxidants are normally included in various percentages to improve their performance. When plastics reach the environment, they can also sorb and accumulate many hydrophobic environmental contaminants, being a potential vector for additives and sorbed contaminants to the organisms. Among the sorbed hydrophobic pollutants are polychlorinated biphenyls (PCBs), polycyclic aromatic hydrocarbons (PAHs), and chlorinated pesticides, all of them belonging to the group of persistent organic pollutants (POPs), as they are persistent, bioaccumulative, and

toxic (PBT) substances. Trace metals and microorganisms such as pathogenic bacteria or viruses might also sorb on microplastics (FAO 2017; GESAMP 2016).

8.8.2.1 Monomers and Polymers

Monomers such as ethylene, propylene, and styrene are the building blocks of polymers that lead to the production of a variety of materials. The most common polymers are acrylonitrile butadiene styrene (ABS), acrylic, epoxy resin, expanded polystyrene (EPS), polyethylene high density (HDPE), polyethylene low density (LDPE), polycarbonate, polycaprolactone, polyethylene (PE), polyethylene terephthalate (PET), poly (glycolic) acid, poly(lactide), poly(methyl methacrylate), polypropylene (PP), polystyrene (PS), polyurethane, polyethylene linear low density, polyamide (Nylon) 4, 6, 11, 66 (PA), polyvinyl alcohol, polyvinyl chloride (PVC), styrene-butadiene rubber, and thermoplastic polyurethane (FAO 2017). All these monomers and polymers can be expected to be part of microplastic particles present in the environment and therefore enter different food value chains.

8.8.2.2 Flame Retardants

Today, there are more than 175 chemicals classified as flame retardants (FRs) (Alaee et al. 2003). Some of these compounds are commonly added to polymers to reduce their flammability. Polybrominated diphenyl ethers (PBDEs) and hexabromocyclododecanes (HBCDs) are the most utilized brominated FRs in plastic manufacture and are commonly added to polystyrene, polyesters, polyolefins, polyamides, epoxies, and ABS. Some HBCDs and PBDEs are simply blended with the polymers and therefore are more likely to leach out of the products, while others can be incorporated into the polymers (Hutzinger and Thoma 1987), a consequence that poses an environmental and food safety concern. PBDEs and HBCD are listed by the Stockholm Convention as Persistent Organic Pollutants (POPs) and are associated with hepatotoxicity, kidney toxicity, endocrine-disrupting effects, and teratogenicity (Muirhead et al. 2006; Yogui and Sericano 2009). Nowadays, organophosphorus flame retardants (OPFRs) are extensively used as additives in order to replace the brominated ones. These compounds have been seen to induce toxicity in vitro and in vivo up to a certain degree (e.g. oxidative stress, cytotoxicity, endocrine disruption), but information on their accumulation and biomagnification in the food chain is still scarce (reviewed by Du et al. 2019).

8.8.2.3 Plasticizers

Substances such as phthalates and BP are used to enhance flexibility and softness and to reduce brittleness. These chemicals are usually added to synthetic polymers used in food packaging such as polycarbonate and epoxy resins but also PE, PP, and/

or PVC (EFSA 2007; FAO/WHO 2009). Both phthalates and BPA have been found to act as endocrine disruptors, causing fertility problems, cardiovascular diseases, development disorders, and reproductive cancers, while the toxicity of other BPs either remains unknown or information is not sufficient (Chen et al. 2016; Ma et al. 2019; Rochester 2013).

8.8.2.4 Antioxidants and Stabilizers

Nonylphenols (NPs) are a group of organic compounds belonging to the family of alkylphenols and are extensively used as a stabilizer in food packaging and as antioxidants in polymers such as rubber, vinyl, polyolefins, polystyrenes, and PVC (GESAMP 2016; USEPA 2010). NPs are known to be endocrine disruptors and have been reported to exert synergistic effects following their co-occurrence with other compounds (Soares et al. 2008; Vethaak et al. 2005).

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