Chapter 10
Microplastics in the Marine Environment: Distribution, Interactions and Effects

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Abstract Microplastics are an emerging marine pollutant. It is important to understand their distribution in the marine environment and their implications on marine habitats and marine biota. Microplastics have been found in almost every marine habitat around the world, with plastic composition and environmental conditions significantly affecting their distribution. Marine biota interact with microplastics including birds, fish, turtles, mammals and invertebrates. The biological repercussions depend on to the size of microplastics encountered, with smaller sizes having greater effects on organisms at the cellular level. In the micrometre range plastics are readily ingested and egested, whereas nanometre-sized plastics can pass through cell membranes. Despite concerns raised by ingestion, the effects of microplastic ingestion in natural populations and the implications for food webs are not understood. Without knowledge of retention and egestion rates of field populations, it is difficult to deduce ecological consequences. There is evidence to suggest that microplastics enter food chains and there is trophic transfer between predators and prey. What is clear is that further research on a variety of marine organisms is required to understand the environmental implications of microplastics in more detail and to establish effects in natural populations.

Keywords Distribution · Ingestion · Trophic transfer · Habitat alterations · Biomagnification · Bioaccumulation

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10.1 Introduction

With the increasing reliance on plastics as an everyday item, and rapid increase in their production and subsequent disposal, the environmental implications of plastics are a growing concern. The benefits of plastics, including their durability and resistance to degradation, inversely result in negative environmental impacts. As user-plastics are primarily "*single use*" items they are generally disposed of within one year of production, and whilst some plastic waste is recycled, the majority ends up in land-fill. Concerns arise when plastics enter the marine environment through indiscriminate disposal and it has been estimated that up to 10 % of plastic debris produced will enter the sea (Thompson 2006). Interactions between litter and the marine environment are complex. The impacts of larger plastic debris are discussed by Kühn et al. (2015) and consequences include aesthetic, social and economic issues (Newman et al. 2015), and numerous environmental impacts on marine biota (Derraik 2002; Barnes et al. 2009). However, with an ever increasing reliance on plastic products, and as plastic production, use and disposal continue, microplastics are of increasing concern (Sutherland et al. 2010). Microplastics enter the sea from a variety of sources (Browne 2015) and distributed by oceans currents; these ubiquitous contaminants are widespread (Cózar et al. 2014). The amount of microplastics in the sea will continue to rise, leading to gradual but significant accumulation in coastal and marine environments (Andrady and Neal 2009).

Increasing evidence of microplastics in the sea has led to a need to understand its environmental impacts as a form of marine pollution. A recent review of marine debris research found only 10 % of publications to focus on microplastics, the majority of which were from the last decade (CBD 2012). Even though plastic is the primary constituent of marine debris, microplastics are considered under-researched due to difficulties in assessing their distribution and abundance (Doyle et al. 2011). It has only been in recent years that international, national and regional efforts were made to quantify microplastics in the sea. The Marine Strategy Framework Directive (MSFD, 2008/56/EC) has highlighted concerns for environmental implications of marine litter and one of the key attributes of the MSFD is to determine the ecological harm caused by microplastics and their associated chemicals (Zarfl et al. 2011).

Microplastics were first described as microscopic particles in the region of 20 μm diameter (Thompson et al. 2004). For the purpose of this study, microplastic refers to items <5 mm in size using the criteria developed by US National Oceanic and Atmospheric Administration (NOAA) (Arthur et al. 2009). The small size of microplastics makes them available for interaction with marine biota in different trophic levels. By inhabiting different marine habitats, a range of organisms are vulnerable to exposure (Wright et al. 2013a). At the millimetre and micrometre scale, sorption of microplastics is dominated by bulk portioning, with effects including blockages when fibres or fragments form aggregates.

Whereas at smaller size ranges, specifically the nanometre scale, there is a potential for microplastics to cause harm to organisms (Galloway 2015; Koelmans et al. 2015). Additionally, the consequences of exposure to chemicals associated with plastics are being investigated (Rochman 2015). A widely cited hypothesis explores how the large surface area to volume ratio of microplastics leaves them prone to adsorbing waterborne organic pollutants and the potential for toxic plasticisers to leach from polymer matrices into organisms tissues (Teuten et al. 2007). It was further hypothesized that if subsequently ingested, microplastics may act as a route for toxin introduction to the food chain (Teuten et al. 2009). Whether microplastics act as vectors depends on the gradient between microplastics and biota lipids (Koelmans 2015).

It is important to understand the transport and distribution of microplastics before understanding their fate, including the physical and chemical effects they could have on marine organisms. The objectives of this chapter are to assess the environmental impact of microplastic in the sea by: (1) summarising the distribution of marine microplastics, including the use of models to understand the distribution; (2) determine the interaction of microplastics with marine organisms.

10.2 The Global Distribution of Microplastics in the Sea

From strandlines on beaches to the deep seafloor and throughout the water column, microplastic research is dominated by studies monitoring microplastic distribution and abundance in the marine environment (Ivar du Sol and Costa 2014). A recent estimate suggested there could be between 7000 and 35,000 tons of plastic floating in the open ocean (Cózar et al. 2014). Another study estimated that more than five trillion pieces of plastic and >250,000 t are currently floating in the oceans (Eriksen et al. 2014). Once in the sea microplastics are transported around the globe by ocean currents where they persist and accumulate. Microplastics are suspended in the water column (e.g. Lattin et al. 2004), surface waters (e.g. Cózar et al. 2014), coastal waters (e.g. Ng and Obbard 2006), estuaries (e.g. Browne et al. 2010), rivers (Sadri and Thompson 2014), beaches (e.g. Browne et al. 2011) and deep-sea sediments (Van Cauwenberghe et al. 2013b; Woodall et al. 2014; Fischer et al. 2015). Suspended in the water column, microplastics can become trapped by ocean currents and accumulate in central ocean regions (e.g. Law et al. 2010). Ocean gyres and convergent zones are noteworthy areas of debris accumulation, as the rotational pattern of currents cause high concentrations of plastics to be captured and moved towards the centre of the region (Karl 1999). As gyres are present in all of the world's oceans, microplastic accumulation can occur at a global scale and has been documented during the past four decades. Distribution is further influenced by wind mixing, affecting the vertical movement of plastics

(Kukulka et al. 2012). Physical characteristics of plastic polymers, including their density, can influence their distribution in the water column and benthic habitats (Murray and Cowie 2011). Buoyant plastics float at the surface, whereas more dense microplastics or those fouled by biota sink to the sea floor. It has recently been estimated that 50 % of the plastics from municipal waste have a higher density than seawater such that it will readily sink to the seafloor (Engler 2012). It is currently not economically feasible nor is it desirable to remove microplastics from the ocean.

A number of concerns have been raised regarding the assessment of microplastic distribution. There are multiple pathways for the introduction of microplastics into the marine environment which do not have accurate timescales for the rate of degradation (Ryan et al. 2009). Quantification is complicated by the size of the oceans in relation to the size of plastics being assessed (Cole et al. 2011), which are further confounded by ocean currents and seasonal patterns introducing spatial and temporal variability (Doyle et al. 2011). As a result, there are various techniques applied to the sampling of microplastics in the marine environment (Löder and Gerdts 2015). Results of studies have been reported in different dimensions, e.g. the number of microplastics in a known water volume (particles m^{-3}) or area measurements (particles km^{-2}). This discrepancy presents a problem when comparing between studies, as it is not possible to compare results directly. For the purpose of this review, which aims to carry out a critical assessment of the global knowledge of microplastic distribution, a conversion was made to enable comparisons between the different dimensions of measurement. It is reasonable to assume that surface samples are collected in the top 0.20 m of water and therefore by making a simple calculation to add a third dimension (firstly converting particle km^{-2} to m^{-2}, then multiplying by 0.20 m to convert to a volume measurement, m^{-3}) we are able to compare different sampling methods in a variety of geographical locations. However, because of current directions in relation to boats, and approximate vessel speeds, it is difficult to calculate the amount of water passing through a net. As nets can ride out of the water, the exact volume of water passing through is unknown: the calculations have to be considered, at best, estimations.

It is important to understand the distribution of microplastics in the sea to grasp their potential impacts. This section will present a number of studies documenting microplastics in geographical regions including the Pacific, Atlantic, European Seas and the Mediterranean Sea, Indian Ocean and polar regions. It will introduce modelling strategies that have been utilised to understand microplastic distribution and accumulation around the globe.

10.2.1 Microplastics in the Pacific Ocean

Numerous studies on microplastics have been undertaken in the Pacific Ocean, the world's largest water basin (Table 10.1). One area which has received considerable attention is the North Pacific Central Gyre (NPCG) located off the west coast of

Table 10.1 Mean abundance (±SD, unless stated otherwise) of microplastic debris in the surface waters of the Pacific Ocean

Location	Equipment used	Amount (±SD)	Particles (m^{-3})	Source
North Pacific				
Bering Sea	Ring net	[a]80 (±190) km^{-2}	0.000016	Day and Shaw (1987)
Bering Sea	Ring/neuston net	1.0 (± 4.2) km^{-2}	0.0000002	Day et al. (1990)
Bering Sea	Sameota sampler/ manta net	Range: 0.004–0.19 m^{-3}	0.004–0.19	Doyle et al. (2011)
Subarctic N.P.	Ring net	[a]3,370 (±2,380) km^{-2}	0.00067	Day and Shaw (1987)
Subarctic N.P.	Ring/neuston net	61.4 (±225.5) km^{-2}	0.000012	Day et al. (1990)
Eastern North Pacific				
Vancouver Island, Canada	Underway sampling	279 (±178) m^{-3}	279	Desforges et al. (2014)
Eastern North Pacific	Plankton net	Estimated 21,290 t afloat	/	Law et al. (2014)
N.P. transitional water	Ring/neuston net	291.6 (±714.4) km^{-2}	0.00012	Day et al. (1990)
N.P. central gyre	Manta net	334,271 km^{-2}	*2.23	Moore et al. (2001)
N.P. central gyre	Manta net	85,184 km^{-2}	0.017	Carson et al. (2013)
N.P. subtropical gyre 1999–2010	Plankton net/manta net/neuston net	Median: 0.116 m^{-3}	0.12	Goldstein et al. (2012)
South Californian current system	Manta net	Median: 0.011–0.033 m^{-3}	0.011–0.033	Gilfillan et al. (2009)
Santa Monica Bay, California, USA	Manta net	3.92 m^{-3}	3.92	Lattin et al. (2004)
Santa Monica Bay, California, USA	Manta net	7.25 m^{-3}	7.25	Moore et al. (2002)
N.P. subtropical gyre	Manta net	Median: 0.02–0.45 m^{-2}	0.0042–0.089	Goldstein et al. (2013)
South Equatorial current	Neuston net	137 km^{-2}	0.000027	Spear et al. (1995)
Equatorial counter current		24 km^{-2}	0.0000048	
Western North Pacific				
Subtropical N.P.	Ring net	[a]96,100 (±780,000) km^{-2}	0.019	Day and Shaw (1987)
Subtropical N.P.	Ring/neuston net	535.1 (±726.1) km^{-2}	0.00011	Day et al. (1990)
Near-shore waters, Japan	Ring/neuston net	128.2 (±172.2) km^{-2}	0.000026	Day et al. (1990)
Kuroshio current system	Neuston net	174,000 (±467,000) km^{-2}	0.034	Yamashita and Tanimura (2007)

(continued)

Table 10.1 (continued)

Location	Equipment used	Amount (±SD)	Particles (m^{-3})	Source
Yangtze estuary system, East China Sea	Neuston net	4,137.3 (±8.2 × 10^4) m^{-3}	4137.3	Zhao et al. (2014)
Geoje Island, South Korea	Bulk sampling, hand-net, manta net	16,000 (±14 × 10^3) m^{-3}	16,000	Song et al. (2014)
South Pacific				
South Pacific subtropical gyre	Manta net	26,898 (±60,818) km^{-2}	0.0054	Eriksen et al. (2013)
Australian coast	Neston net	[b]4,256.3 (±757.8) km^{-2}	0.00085	Reisser et al. (2013)
	Manta net			

If particles in m^{-3} were not reported, the values have been converted as follows: (1) km^{-2} to m^{-2}: by division by 1,000,000 followed by multiplication by 0.2 m; (2) m^{-2} to m^{-3} carried out by 0.2 multiplication
[a]Mean ±95 % confidence intervals
[b]Mean ± standard error

California, USA. The gyre contains possibly the most well publicised area of plastic accumulation, known as the "*Great Pacific Garbage Patch*" (Kaiser 2010). Microplastic concentrations in the NPCG have increased by two orders of magnitude in the last four decades (Goldstein et al. 2012). In comparison, microplastic abundance in the North Pacific subtropical gyre (NPSG) is widespread and spatially variable, but values are two orders of magnitude lower than in the NPCG (Goldstein et al. 2013). Microplastic studies in the south Pacific are limited to the subtropical gyre where an increasing trend of microplastics was found towards the centre of the gyre (5.38 particles m^{-3} [1] Eriksen et al. 2013). In a similar way to macroplastic debris, oceanographic features strongly affect the distribution of microplastics in open oceans and areas of upwelling create oceanographic convergence zones for marine debris.

Coastal ecosystems of the Pacific appear to be impacted by microplastics in areas of nutrient upwelling (Doyle et al. 2011) and influenced by local weather systems (Moore et al. 2002; Lattin et al. 2004). Microplastic load increased further inshore, reflecting the inputs from terrestrial runoff and particles re-suspended from sediments following storms (Lattin et al. 2004). Microplastics are in turn transported by ocean currents from populated coastal areas (Reisser et al. 2013). This is also reflected in offshore subsurface waters which had 4–27 times less plastics than coastal sites in the northeast Pacific (Desforges et al. 2014).

Pre-production plastic resin pellets and fragments wash up on coastlines worldwide and have been recovered from several Pacific beaches (Table 10.2). Plastic pellets, typically 3–5 mm in size, are made predominantly from the polymers polyethylene and polypropylene (Endo et al. 2005; Ogata et al. 2009). The average

[1] Calculated from km^{-2}.

Table 10.2 Mean microplastic abundance (±SD, unless otherwise stated) in sediments from the Pacific

Location	Types	Amount (±SD)	Source
North Pacific			
Pacific beaches	Fragments 10 mm	/	Hirai et al. (2011)
9 beaches, Hawaiian islands	Fragments 1–15 mm	[a]37.8 kg^{-1}	[a]McDermid and McMullen (2004)
	Pellets 1–15 mm	[a]4.9 kg^{-1}	
Hawaiian islands	Pellets and fragments	/	Rios et al. (2007)
Kauai, Hawaiian islands	Fragments and pellets 0.8–6.5 mm	/	Corcoran et al. (2009)
Kauai, Hawaiian islands	Fragments <1 cm	/	Cooper and Corcoran (2010)
Kamillo Beach, Hawaii	Pellets and fragments	Total: 248	Carson et al. (2011)
Northeast Pacific			
Los Angeles, California, USA	Pellets and fragments	/	Rios et al. (2007)
San Diego, California, USA	Pellets and fragments <5 mm	/	Van et al. (2012)
Beaches, western USA	Pellets	/	Ogata et al. (2009)
Guadalupe Island, Mexico	Pellets and fragments	/	Rios et al. (2007)
Northwest Pacific			
Coastal beaches, Russia	Fragments and pellets	[b]29 m^{-2}	Kusui and Noda (2003)
Tokyo, Japan	Pellets	>1,000 m^{-2}	Kuriyama et al. (2002)
Coastal beaches, Japan	Pellets	/	Mato et al. (2001)
Coastal beaches, Japan	Pellets	[b]0.52 m^{-2}	Kusui and Noda (2003)
	Fragments	[b]1.1 m^{-2}	
Coastal beaches, Japan	Pellets <5 mm	>100 per beach	Endo et al. (2005)
Korean Strait			
Heugnam Beach, South Korea	PS spheres	874 (±377) m^{-2}	Heo et al. (2013)
	Fragments	25 (±10) m^{-2}	
	Pellets	41 (±19) m^{-2}	
South China Sea			
Ming Chau Island, Vietnam	Pellets	/	Ogata et al. (2009)
Hong Kong, China	Pellets	/	Ogata et al. (2009)
South Pacific			
Coastal beaches, New Zealand	Pellets <5 mm	>1,000 m^{-1}	Gregory (1978)
Coastal beaches, Chile	Fragments and pellets 1–10 mm	30 m^{-2}	Hidalgo-Riz and Thiel (2013)
Easter Island, Chile	Fragments and pellets 1–10 mm	805 m^{-2}	Hidalgo-Riz and Thiel (2013)

[a]Calculated from total plastic collected from an overall total of 440 L of beach sediment
[b]Calculated from total plastics found over total survey area

abundance of plastic fragments on beaches in the southeast Pacific was greater in isolated areas (Easter Island: >800 items m^{-2}) than on beaches from continental Chile (30 items m^{-2}) (Hidalgo-Ruz and Thiel 2013). This trend has been seen in the Hawaiian archipelago, where the remotest beaches on Midway Atoll and Moloka'I contained the highest quantity of plastic particles (McDermid and McMullen 2004; Corcoran et al. 2009; Cooper and Corcoran 2010).

10.2.2 Microplastics in the Atlantic Ocean

Research on microplastic distribution in the Atlantic is less extensive than in the Pacific (Table 10.3), but includes a number of long-term studies. A time-series conducted in the north Atlantic and Caribbean Sea identified microplastics in 62 % of the trawls conducted with densities reaching 580,000 particles km^{-2} (Law et al. 2010). Distinct patterns emerged with the highest concentration (83 % of plastics) in subtropical latitudes, 22°N and 88°N, of the north Atlantic gyre marking the presence of a large-scale convergence zone (Law et al. 2010; Morét-Ferguson et al. 2010) similar to the south Pacific (Eriksen et al. 2013). Converging surface currents driven by winds are assumed to be the driving force of this accumulation. To assess long-term trends in abundance, a time-series data set of continuous plankton recorder (CPR) samples from north Atlantic shipping routes were re-examined and microplastics were identified from the 1960s with a significant increase over time (Thompson et al. 2004). Regular sampling schemes have begun to monitor the spatial and temporal trends of microplastics in the northeast Atlantic and found microplastics to be widespread and abundant (Lusher et al. 2014).

Microplastics accumulate in the coastal pelagic zones of the Atlantic (Table 10.3). Water samples from the Portuguese coast identified microplastics in 61 % of the samples with higher concentrations found in Costa Vicentina and Lisbon (0.036 and 0.033 particles m^{-3}, respectively) than in the Algarve and Aveiro (0.014 and 0.002 particles m^{-3}, respectively). These results are probably related to the proximity to urban areas and river runoff (Frias et al. 2014), which is similar to the trend seen in the Pacific. Following a MARMAP cruise in the south Atlantic, microplastic beads were present in 14.6–34.2 % of tows conducted (van Dolah et al. 1980). Pelagic subsurface plankton samples from a geographically isolated archipelago, Saint Peter and Saint Paul, were not free of microplastic fragments. Modelling studies suggested that oceanographic mechanisms promote the topographic trapping of zooplankton and therefore microplastics might be retained by small-scale circulation patterns (Ivar do Sul et al. 2013). Additionally, research in the Firth of Clyde (U.K.) indicated that intense environmental sampling regimes are necessary to encompass the small-scale and temporal variation in coastal microplastic abundance (Welden, *pers. comm.*).

Microplastic granules and pellets have been identified on Atlantic beaches since the 1980s (Table 10.4). It was hypothesised that pre-production pellets are

Table 10.3 Mean abundance (±SD, unless stated otherwise) of microplastic debris in the surface waters of the Atlantic Ocean

Location	Equipment used	Amount (± SD)	Particles (m^{-3})	Source
North Atlantic				
North Atlantic gyre (29–31°N)	Plankton net	20,328 (±2,324) km^{-2}	0.0041	Law et al. (2010)
North Atlantic	Continuous plankton recorder (CPR)	1960–1980: 0.01 m^{-3}	0.01	Thompson et al. (2004)
		1980–2000: 0.04 m^{-3}	0.04	
Northwest Atlantic				
Northwest Atlantic	Neuston net	[a]490 km^{-2}	0.00098	Wilber (1987)
Block Island Sound, USA	Plankton net	Range: 14–543 m^{-3}	14–543	Austin and Stoops-Glass (1977)
Gulf of Maine	Plankton net	1534 (±200) km^{-2}	0.00031	Law et al. (2010)
New England, USA	Plankton net	Mean ranges: 0.00–2.58 m^{-3}	0.00–2.58	Carpenter et al. (1972)
Continental shelf, west coast USA	Neuston net	2,773 km^{-2}	0.00056	Colton et al. (1974)
Western Sargasso Sea	Neuston net	3,537 km^{-2}	0.00071	Carpenter and Smith (1972)
Caribbean Sea				
Caribbean	Neuston net	60.6–180 km^{-2}	0.000012–0.000036	Colton et al. (1974)
Caribbean	Plankton net	1,414 (±112) km^{-2}	0.00028	Law et al. (2010)
Northeast Atlantic				
Offshore, Ireland	Underway sampling	2.46 m^{-3}	2.46	Lusher et al. (2014)
English Channel, U.K.	Plankton net	0.27 m^{-3}	0.27	Cole et al. (2014a)
Bristol Channel, U.K.	Lowestoft plankton sampler	Range: 0–100 m^{-3}	0–>100	Morris and Hamilton (1974)
Severn Estuary, U.K.				Kartar et al. (1973, 1976)
Portuguese coast	Neuston net/ CPR	0.02–0.036 m^{-3}	0.02–0.036	Frias et al. (2014)
Equatorial Atlantic				
St. Peter and St. Paul Archipelago, Brazil	Plankton net	0.01 m^{-3}	0.01	Ivar do Sul et al. (2013)

(continued)

Table 10.3 (continued)

Location	Equipment used	Amount (± SD)	Particles (m^{-3})	Source
South Atlantic				
South Atlantic Bight	Neuston net	Mean weight: 0.03–0.08 mg m^{-2}		van Dolah et al. (1980)
Cape Basin, South Atlantic	Neuston sledge	1,874.3 km^{-2}	0.00037	Morris (1980)
Cape Province, South Africa	Neuston net	3,640 km^{-2}	0.00073	Ryan (1988)
Fernando de Noronha, Abrolhos and Trindade, Brazil	Manta net	0.03 m^{-3}	0.03	Ivar do Sul et al. (2014)
Gioana estuary, Brazil	Conical plankton net	26.04–100 m^{-3}	0.26	Lima et al. (2014)

If particles in m^{-3} were not reported, the values have been converted as follows: (1) km^{-2} to m^{-2}: by division by 1,000 000, followed by multiplication by 0.2 m; (2) m^{-2} to m^{-3} carried out multiplication by 0.2

[a]This value is for pellets only, although fragments >5 mm were also reported

Table 10.4 Mean microplastic abundance (±SD, unless stated otherwise) in sediments from the Atlantic

Location	Types	Amount	Source
North Atlantic			
Nova Scotia, Canada	Pellets	Max: <10 m^{-1}	Gregory (1983)
Nova Scotia, Canada	Fibres	200–800 fibres kg^{-1}	Mathalon and Hill (2014)
Beaches, eastern USA	Pellets		Ogata et al. (2009)
Factory beaches, New York, USA	Spheres		Hays and Cormons (1974)
*Maine, USA	Pellets and fragments	105 kg^{-1}	Graham and Thompson (2009)
*Florida, USA	Pellets and fragments	214 kg^{-1}	Graham and Thompson (2009)
Florida Keys, USA	Pellets and fragments	100–1,000 m^{-2}	Wilber (1987)
Cape Cod, USA	Pellets and fragments	100–1,000 m^{-2}	Wilber (1987)
North Carolina, USA	Fragments <5 cm	60 % of debris in size class	Viehman et al. (2011)
Bermuda	Pellets	>5,000 m^{-1}	Gregory (1983)
Bermuda	Pellets and fragments	2,000–10,000 m^{-2}	Wilber (1987)
Bahamas	Pellets and fragments	Windward: 500–1,000 m^{-2}	Wilber (1987)
		Leeward: 200–500 m^{-2}	
Lesser Antilles	Pellets and fragments	Windward: 100–5,000 m^{-2}	Wilber (1987)
		Leeward: 50–100 m^{-2}	

(continued)

Table 10.4 (continued)

Location	Types	Amount	Source
Le Havre, France	Pellets		Endo et al. (2013)
Costa Nova, Portugal	Pellets		Ogata et al. (2009)
Lisbon, Portugal	Fibres and pellets		Frias et al. (2010)
Portuguese coast	Pellets and fragments	185.1 m^{-2}	Martins and Sobral (2011)
Portuguese coast	Pellets 3–6 mm	1,289 m^{-2}	Antunes et al. (2013)
*Porcupine abyssal plain	Fragments	[a]40 item m^{-2}	Van Cauwenberghe et al. (2013b)
Canary Islands, Spain	Pellets and fragments <5 mm	<1 g kg^{-1}–>40 g kg^{-1}	Baztan et al. (2014)
English Channel			
Estuarine sediment, U.K.	Fragments and fibres	Maximum: 31 kg^{-1}	Thompson et al. (2004)
*Subtidal sediments, U.K.	Fragments and fibres	Maximum: 86 kg^{-1}	Thompson et al. (2004)
Plymouth, U.K.	Pellets		Ogata et al. (2009)
South Devon, U.K.	Pellets	~100	Ashton et al. (2010)
Tamar estuary, U.K.	Fragments <1 mm	65 % of total debris	Browne et al. (2010)
Southwest England, U.K.	Pellets	~100 at each location	Holmes et al. (2012)
South Atlantic			
Fernando de Noronha, Brazil	Pellets 23 %	[b]3.5 kg^{-1}	Ivar do Sul et al. (2009)
	Fragments 65 %	[b]9.63 kg^{-1}	
	Nylon monofilament 5 %	[b]0.73 kg^{-1}	
Recife, Brazil	Fragments 96.7 %	[c]300,000 m^{-3}	Costa et al. (2010)
	Pellets 3.3 %		
Northeast Brazil	Fragments 1–10 mm	59 items m^{-3}	Costa et al. (2011)
*Southern Atlantic	Fragments	[a]40 items m^{-2}	Van Cauwenberghe et al. (2013b)
Santos Bay, Brazil	Pellets	0–2,500 m^{-3}	Turra et al. (2014)

All sediments are beach sediments unless annotated with *, which refers to benthic or subtidal sediment. d.w. is dry weight of sediment. When originally reported in l, values were converted to kg
[a]Estimated from 1 item 25 cm^{-2}
bCalculated from total weight of sand (13,708 g)
[c]Calculated from 0.3 items cm^{-3}

transported by trans-oceanic currents before being washed ashore in areas such as the mid-Atlantic Archipelago, Fernando de Noronha (Ivar do Sul et al. 2009). Fragments make up a considerable proportion of marine debris on saltmarsh beaches in North Carolina (Viehman et al. 2011), the Canary Islands (Baztan et al. 2014) and beaches and intertidal plains in Brazil (Costa et al. 2010, 2011). Whereas, fibres were primarily identified in sediment samples from an intertidal ecosystem in Nova Scotia, Canada (Mathalon and Hill 2014).

Table 10.5 Mean microplastic abundance in surface waters of the Mediterranean and European seas

Location	Equipment used	Amount	Particles (m^{-3})	Source
West coast, Sweden	Manta net (80 µm)	Range: 150–2,400 m^{-3}	150–2400	Norén (2007)
	Manta net (450 µm)	Range: 0.01–0.14 m^{-3}	0.01–0.14	
Skagerrak, Sweden	Submersible in situ pump	Maximum: 102,000 m^{-3}	102,000	Norén and Naustvoll (2011)
Northwest Mediterranean	Manta net	1.33 m^{-2}	0.27	Collignon et al. (2012)
Bay of Calvi, Corsica, France	wp2 net	0.062 m^{-2}	0.012	Collignon et al. (2014)
Gulf of Oristano, Sardinia, Italy	Manta net	0.15 m^{-3}	0.15	de Lucia et al. (2014)
North Sea, Finland	Manta net	Range: 0–0.74 m^{-3}	0–0.74	Magnusson (2014)

If particles in m^{-3} were not reported, the values have been converted as follows: (1) km^{-2} to m^{-2}: by division by 1,000,000 followed by multiplication by 0.2 m; (2) m^{-2} to m^{-3} carried out multiplication by 0.2

10.2.3 Microplastics in European Seas and the Mediterranean Sea

Marine litter including microplastic is a serious concern in the Mediterranean, with plastics accounting for 70–80 % of litter identified (Fossi et al. 2014). This enclosed water basin is not free of microplastic contamination (Table 10.5). Levels of microplastics in surface waters of the northwest Mediterranean were similar to those reported for the NPCG, (0.27 particles m^{-3} [2] Collignon et al. 2012), and areas far away from point sources of pollution have high microplastic abundance (0.15 particles m^{-3}; de Lucia et al. 2014). Interestingly, fewer particles were recorded from surface waters from coastal Corsica (0.012 particles m^{-3} [3] ; Collignon et al. 2014). Microplastic distribution is strongly influenced by wind stress, which may redistribute particles in the upper layers of the water column and preclude sampling by surface tows (Collignon et al. 2012). Oceanographic influences may affect the distribution of microplastics in the Mediterranean. Further research will help to clarify if the new hypothesis by de Lucia et al. (2014) holds, which suggests that upwelling dilutes the amount of plastic in the surface waters.

Microplastics, including beads and pellets, have been widely reported for sedimentary habitats and beaches in European Seas and the Mediterranean Sea (Table 10.6). Microplastics have been extracted from sediments from Norderney, in the North Sea (Dekiff et al. 2014; Fries et al. 2013) and samples taken at the East Frisian

[2] Calculated from 1.334 particles m^{-2}.

[3] Calculated from 0.062 particles m^{-2}.

Table 10.6 Mean microplastic abundance (±SD, unless stated otherwise) in sediments from the Mediterranean and European seas

Location	Types	Amount	Source
North Sea			
Harbor sediment, Sweden	Fragments	[a]20 and 50 kg^{-1}	Norén (2007)
Industrial harbor sediment, Sweden	Pellets	[a]3320 kg^{-1}	Norén (2007)
Industrial coastal sediment, Sweden	Pellets	[a]340 kg^{-1}	Norén (2007)
Spiekeroog, Germany	Fibres and granules	[b]3,800 kg^{-1} d.w.	Liebezeit and Dubaish (2012)
Jade System, Germany	Fibres	88 (±82) kg^{-1}	Dubaish and Liebezeit (2013)
	Granules	64 (±194) kg^{-1}	
Norderney, Germany	Fragments	/	Fries et al. (2013)
Norderney, Germany	Fragments	1.3, 1.7, 2.3 kg^{-1} d.w.	Dekiff et al. (2014)
Zandervoord, Netherlands	Pellets	/	Ogata et al. (2009)
*Harbor, Belgium	Fibres, granules, films, spheres	116.7 (±92.1) kg^{-1} d.w.	Claessens et al. (2011)
*Continental shelf, Belgium	Fibres, granules, films	97.2 (±18.6) kg^{-1} d.w.	Claessens et al. (2011)
Beach, Belgium	Fibres, granules, films	92.8 (±37.2) kg^{-1} d.w.	Claessens et al. (2011)
Beach, Belgium	Pellets and fragments	17 (±11) kg^{-1}	Van Cauwenberghe et al. (2013a)
Forth estuary, U.K.	Pellets	/	Ogata et al. (2009)
Mediterranean Sea			
8 beaches, Malta	Pellets	0.7–167 m^{-2}	Turner and Holmes (2011)
Sicily, Italy	Pellets	/	Ogata et al. (2009)
Venice lagoon, Italy	Fragments and fibres	672–2,175 kg^{-1} d.w.	Vianello et al. (2013)
*Nile deep sea fan, Mediterranean	Fragments	[c]40 items m^{-2}	Van Cauwenberghe et al. (2013b)
Lesvos, Greece	Pellets	/	Karapangioti and Klontza (2007)
Kato Achaia, Greece	Pellets	/	Ogata et al. (2009)
Beaches, Greece	Pellets	/	Karapanagioti et al. (2011)
Kea Island, Greece	Pellets	10, 43, 218, 575 m^{-2}	Kaberi et al. (2013)
Tripoli-Tyre, Lebanon	Pellets and fragments	/	Shiber (1979)
Costa del Sol, Spain	Pellets	/	Shiber (1982)
18 beaches, western Spain	Pellets	/	Shiber (1987)
Izmir, Turkey	Pellets	/	Ogata et al. (2009)

All sediments are beach sediments unless annotated with *, which refers to benthic or subtidal sediment. d.w. is dry weight of sediment. When originally reported in l, values were converted to kg
[a]Calculated from 100 ml sediment
[b]Calculated from 10 g sediment
[c]Estimated from 1 item 25 cm^{-2}

Islands, where tidal flats were more contaminated than sandy beaches (Liebezeit and Dubaish 2012). Areas of low hydrodynamics appear to have high microplastic abundance, such as the Venice lagoon (Vianello et al. 2013). Reduced water movement could also be attributed to the difference between concentrations of microplastics in Belgium: higher concentrations of microplastics were identified in sediments from Belgium harbors (Claessens et al. 2011) than in beach samples (Van Cauwenberghe et al. 2013a). Lastly, microplastics were recorded in deep offshore sediments (Van Cauwenberghe et al. 2013b; Fischer et al. 2015), which shows that microplastics sink to the deep seafloor. In fact, the deep seafloor may be considered a major sink for microplastic debris (Woodall et al. 2014) and explain the current mismatch between estimated global inputs of plastic debris to the oceans (Jambeck et al. 2015) and field data (Cózar et al. 2014; Eriksen et al. 2014), which refer largely to floating litter.

10.2.4 Microplastics in the Indian Ocean and Marginal Seas

To date there are few large-scale reports on microplastics from the Indian Ocean. Reddy et al. (2006) reported microplastic fragments from a ship-breaking yard in the Arabian Sea, and microplastics accounted for 20 % of the plastics recorded on sandy beaches in Mumbai (Jayasiri et al. 2013). Pellets were also recorded on

Table 10.7 Mean microplastic abundance (±SD, unless stated otherwise) in sediments from the Indian Ocean and marginal seas

Location	Types	Amount	Source
Arabian Sea			
Ship-breaking yard, Alang-Sosiya, India	Fragments	81 mg kg^{-1}	Reddy et al. (2006)
Mumbai, Chennai and Sunderbans, India	Pellets	/	Ogata et al. (2009)
Mumbai, India	Fragments	41.85 % of total plastics	Jayasiri et al. (2013)
East Asian Marginal Seas			
Coastline, Singapore	Fragments	/	Ng and Obbard (2006)
Coastline, Singapore	Fibres, grains, fragments	36.8 ± 23.6 kg^{-1}	Mohamed Nor and Obbard (2014)
Selangor, Malaysia	Pellets	<18 m^{-2}	Ismail et al. (2009)
Lang Kawi, Penang and Borneo, Malaysia	Pellets	/	Ogata et al. (2009)
Rayong, Thailand	Pellets	/	Ogata et al. (2009)
Jakarta Bay, Indonesia	Pellets	/	Ogata et al. (2009)
Southern Indian Ocean	Pellets	/	Ogata et al. (2009)
Mozambique	Pellets	/	Ogata et al. (2009)
Gulf of Oman	Pellets	>50–200 m^{-2}	Khordagui and Abu-Hilal (1994)
Arabian Gulf	Pellets	>50–80,000 m^{-2}	

All sediments are beach sediments

Malaysian beaches (Ismail et al. 2009). Most of the studies shown in Table 10.7 are part of the "*International Pellet Watch*" (Takada 2006; Ogata et al. 2009). Shoreline surveys conducted in surface waters and sediments on Singapore's coasts identified microplastics >2 μm (Ng and Obbard 2006). This highlights an area that requires further investigation to obtain a wider picture of microplastic distribution around the globe.

10.2.5 Microplastics in Polar Regions

Prior to 2014, there had been no direct studies of microplastics in either the Arctic or Antarctica; the plastic flux into the Arctic Ocean has been calculated to range between 62,000 and 105,000 tons per year, with variation due to spatial heterogeneity, temporal variability and different sampling methods (Zarfl and Matthies 2010). With the estimated value four to six orders of magnitude below the atmospheric transport and ocean current fluxes, the study concluded that plastic transport levels to the Arctic are negligible and that plastics are not a likely vector for organic pollutants to the Arctic. However, Obbard et al. (2014) published results from ice cores collected from remote locations in the Arctic Ocean. The levels of microplastics observed (range: 38–234 particles m^{-3}) were two orders of magnitude greater than previously reported in the Pacific gyre (Goldstein et al. 2012). Macroplastics have been identified floating in surface waters of Antarctica. However, trawls for microplastics did not catch any particles (Barnes et al. 2010). Dietary studies of birds from the Canadian Arctic have reported ingested plastics (Mallory et al. 2006; Provencher et al. 2009, 2010), and macroplastics were observed on the deep Arctic seafloor (Bergmann and Klages 2012). This indirect evidence suggests that microplastics have already entered polar regions. A modelling study even suggests the presence or formation of a sixth garbage patch in the Barents Sea (van Sebille et al. 2012).

10.2.6 Modelling the Distribution of Microplastics

Studies have highlighted the interaction of oceanographic and environmental variables on the distribution of microplastics (e.g. Eriksen et al. 2013). As polymer densities affect the distribution of plastics in the water column, it is important to understand how microplastics are transported at the surface and at depths. Knowledge of point-source pollution, including riverine input and sewage drainage into marine and coastal environments, can be useful in understanding the extent to which certain ecosystems are affected. Furthermore, knowledge of plastic accumulation on beaches will benefit the study of microplastics. For example, a study of plastic litter washed onto beaches developed a particle tracking model, which indicated that, if levels of plastic outflow remain constant over the coming decade, plastic litter quantity on beaches would continue to increase, and in some cases (3 % of all east Asian beaches) could see a 250-fold increase in plastic litter (Kako et al. 2014). If not removed, these larger items of plastic litter will break down into microplastics over time.

The fate of plastics in the marine environment is affected by poorly understood geophysical processes, including ocean mixing of the sea-surface boundary layer, re-suspension from sediments, and sinking rates plastics denser than seawater. Modelling approaches are required to further understand, and accurately estimate the global distribution, residence time, convergence zones, and ecological consequences of microplastics (Ballent et al. 2013). Models predicting the breakdown, fragmentation, and subsequent mixing and re-suspension of microplastics in sediments and seawater could provide an estimation of microplastic accumulation over short and long time scales; as well as an estimation of the dispersal patterns of microplastics in the marine environment. Generalized linear models have indicated that oceanographic mechanisms may promote topographic trapping of zooplankton and microplastics, which may be retained by small-scale circulation patterns in the Equatorial Atlantic, suggesting there is an outward gradient of microplastics moving offshore (Ivar do Sul et al. 2013). The recovery of plastic from surface seawater is dependent on wind speeds: stronger winds resulted in the capture of fewer plastics because wind-induced mixing of the surface layer vertically distributes plastics (Kulkula et al. 2012). Furthermore, by integrating the effect of vertical wind mixing on the concentrations of plastics in Australian waters, researchers estimated depth-integrated plastic concentrations, with high concentrations expected at low wind speeds. Thus, with the inverse relationship between wind force and plastic concentration, net tow concentrations of microplastics increased by a factor of 2.8 (Reisser et al. 2013).

Ballent et al. (2013) used the MOHID modelling system to predict the dispersal of non-buoyant pellets in Portugal using their density, settling velocity and re-suspension characteristics. Researchers simulated the transport of microplastic pellets over time using oceanographic processes, scales and systems. Model predictions suggest that the bottom topography restricts pellet movement at the head of the Nazaré Canyon with a potential area of accumulation of plastics pellets on the seafloor, implying long-term exposure of benthic ecosystems to microplastics. Tidal forces, as well as large-scale oceanographic circulation patterns are likely to transport microplastics up and down the Nazaré Canyon, which may be greatly increased during mass transport of waters linked to storms (Ballent et al. 2013) or deep-water cascading events (Durrieu de Madron et al. 2013).

With residence times from decades to centuries predicted for microplastics in the benthic environment (Ballent et al. 2013), future studies should assess the degradation of microplastics on the seafloor to be able to estimate residence times in those potential sink environments. Coupled with observations of microplastics in surface waters, the total oceanic plastic concentrations might be underestimated because of limited but growing knowledge of the geophysical and oceanographic processes in the surface waters. Furthermore, as microplastics degrade towards a nanometre scale, transport properties may be affected, and as a result, long-term transport models will need to be corrected. Modelling should be adapted to bring in ecological consequences of microplastics in benthic environments and the water column.

Research should focus on critical areas such as biodiversity hotspots and socio-economic hotspots that could affect vulnerable marine biota and coastal communities.

10.2.7 Summary

Microplastics have been documented in almost every habitat of the open oceans and enclosed seas, including beaches, surface waters, water column and the deep seafloor. Although most water bodies have been investigated, there is a lack of published work from polar regions and the Indian Ocean. Further research is required to accurately estimate the amount of different types of microplastics in benthic environments around the globe. Distribution of microplastics depends on environmental conditions including ocean currents, horizontal and vertical mixing, wind mixing and biofilm formation, as well as the properties of individual plastic polymers. A number of modelling approaches have been considered in the recent literature, which highlighted the effect of wind on the distribution of microplastics in the ocean. Oceanographic modelling of floating debris has shown accumulation in ocean gyres, and the distribution of microplastics within the water column appears to be dependent on the composition, density and shape of plastic polymers affecting their buoyancy. Further modelling studies may help to identify and predict regions with ecological communities and fisheries more vulnerable to the potential consequences of plastic contamination. The distribution of microplastic plays a significant role in terms of which organisms and habitats are affected. Widespread accumulation and distribution of microplastics raises concerns regarding the interaction and potential effects on marine organisms.

10.3 Interactions of Microplastics with Marine Organisms

Recently, Wright et al. (2013a) discussed the biological factors, which could enhance microplastic bioavailability to marine organisms: the varying density of microplastics allows them to occupy different areas of the water column and benthic sediments. As microplastics interact with plankton and sediment particles, both suspension and deposit feeders may be at risk of accidentally or selectively ingesting marine debris. However, the relative impacts are likely to vary across the size spectrum of microplastic in relation to the organisms affected, which is dependent on the size of the microplastic particles encountered. Microplastics in the upper end of the size spectrum (1–5 mm) may compromise feeding and digestion. For example, Codina-García et al. (2013) isolated such pellets and fragments from the stomachs of seabirds. Particles <20 μm are actively ingested by small invertebrates (e.g. Thompson et al. 2004) but they are also egested (e.g. Lee et al. 2013). Studies have shown that nanoparticles can translocate (e.g. Wegner et al. 2012) and model simulations have indicated that nano-sized polystyrene (PS) particles may permeate into the lipid membranes of organisms, altering the membrane structure, membrane protein activity, and therefore cellular function (Rossi et al. 2013). The following section deals with incidences of ingestion, trophic transfer and provision of new habitat by the presence of microplastics in the marine environment. Although the sections contain examples, comprehensive lists of microplastics ingestion are included in the corresponding tables.

10.3.1 Ingestion

Ingestion is the most likely interaction between marine organisms and microplastics. Microplastics' small size gives them the potential to be ingested by a wide range of biota in benthic and pelagic ecosystems. In some cases, organisms feeding mechanisms do not allow for discrimination between prey and anthropogenic items (Moore et al. 2001). Secondly, organisms might feed directly on microplastics, mistaking them for prey or selectively feed on microplastics in place of food (Moore 2008). If there is a predominance of microplastic particles associated with planktonic prey items, organisms could be unable to differentiate or prevent ingestion. A number of studies have reported microplastics from the stomachs and intestines of marine organisms, including fish and invertebrates. Watts et al. (2014) showed that shore crabs (*Carcinus maenas*) will not only ingest microplastics along with food (evidence in the foregut) but also draw plastics into the gill cavity because of their ventilation mechanism: this highlights that it is important to consider all sorts of routes of exposure to microplastics. If organisms ingest microplastics they could have adverse effects on individuals by disrupting feeding and digestion (GESAMP 2010). Laboratory (Table 10.8) and field (Table 10.9) studies highlighted that microplastics are mistaken for food by a wide variety of animals including birds, fish, turtles, mammals and invertebrates. Despite concerns raised regarding microplastic ingestion, few studies specifically examined the occurrence of microplastic in natural, in situ, populations as it is methodologically challenging to assess microplastic ingestion in the field (Browne et al. 2008).

10.3.1.1 Planktonic Invertebrates

Microplastics can enter the very base of the marine food web via absorption. Such was observed when charged nano-polystyrene beads were absorbed into the cellulose of a marine alga (*Scenedesmus* spp.), which inhibited photosynthesis and caused oxidative stress (Bhattacharya et al. 2010). Microplastics can also affect the function and health of marine zooplankton (Cole et al. 2013; Lee et al. 2013). Decreased feeding was observed following ingestion of polystyrene beads by zooplankton (Cole et al. 2013). Furthermore, adult females and nauplius larvae of the copepod (*Tigriopus japonicus*) survived acute exposure, but increased

Table 10.8 Laboratory studies exposing organisms to microplastics

Organism	Size of ingested material	Exposure concentration	Effect	Source
Phylum Chlorophyta				
Scenedesmus spp.	20 nm	1.6–40 mg mL^{-1}	Absorption, ROS increased, photosynthesis affected	Bhattacharya et al. (2010)
Phylum Haptophyta				
Isochrysis galbana	2 µm PS	9×10^4 mL^{-1}	Microspheres attached to algae, no negative effect observed	Long et al. (2014)
Phylum Dinophyta				
Heterocapsa triquetra	2 µm PS	9×10^4 mL^{-1}	Microspheres attached to algae, no negative effect observed	Long et al. (2014)
Phylum Cryptophyta				
Rhodomonas salina	2 µm PS	9×10^4 mL^{-1}	Microspheres attached to algae, no negative effect observed	Long et al. (2014)
Phylum Ochrophyta				
Chaetoceros neogracilis	2 µm PS	9×10^4 mL^{-1}	Microspheres attached to algae, no negative effect observed	Long et al. (2014)
Phylum Ciliophora				
Strombidium sulcatum	0.41–10 µm	5–10 % ambient bacteria concentration	Ingestion	Christaki et al. (1998)
Tintinnopsis lobiancoi	10 µm PS	1,000, 2,000, 10,000 mL^{-1}	Ingestion	Setälä et al. (2014)
Phylum Rotifera				
Synchaeta spp.	10 µm PS	2,000 mL^{-1}	Ingestion	Setälä et al. (2014)
Phylum Annelida				
Class Polychaete				
Lugworm (*Arenicola marina*)	20–2000 µm	1.5 g L^{-1}	Ingestion	Thompson et al. (2004)

(continued)

Table 10.8 (continued)

Organism	Size of ingested material	Exposure concentration	Effect	Source
Arenicola marina	130 μm UPVC	0–5 % by weight	Ingestion, reduced feeding, increased phagocytic activity, reduced available energy reserves, lower lipid reserves	Wright et al. (2013b)
Arenicola marina	230 μm PVC	1500 g of sediment mixture	Ingestion, oxidative stress	Browne et al. (2013)
Arenicola marina	400–1300 μm PS	0, 1, 10, 100 g L^{-1}	Ingestion, reduced feeding, weight loss	Besseling et al. (2013)
Fan worm (*Galeolaria caespitosa*)	3–10 μm	5 microspheres μL^{-1}	Ingestion	Bolton and Havenhand (1998)
Galeolaria caespitosa	3 and 10 μm PS	635, 2,240, 3,000 beads mL^{-1}	Ingestion, size selection, egestion	Cole et al. (2013)
Mud worms (*Marenzelleria* spp.)	10 μm PS	2,000 mL^{-1}	Ingestion	Setälä et al. (2014)
Phylum Mollusca				
Class Bivalvia				
Blue mussel (*Mytilus edulis*)	30 nm PS	0, 0.1, 0.2, and 0.3 g L^{-1}	Ingestion, pseudofaeces, reduced filtering	Wegner et al. (2012)
Mytilus edulis	0–80 μm HDPE	2.5 g L^{-1}	Ingestion, retention in digestive tract, transferred to lymph system, immune response	von Moos et al. (2012) Köhler (2010)
Mytilus edulis	0.5 μm PS	50 μL per 400 ml seawater	Ingestion, trophic transfer → *Carcinus maenas*	Farrell and Nelson (2013)
Mytilus edulis	3, 9.6 μm	0.51 g L^{-1}	Ingestion, retention in digestive tract, transferred to lymph system	Browne et al. (2008)
Mytilus edulis	10 μm PS	2×10^4 mL^{-1}	Ingestion, egestion	Ward and Tagart (1989)
		1,000 mL^{-1}		Ward and Kach (2009)
Mytilus edulis	10, 30 μm PS	3.10×10^5 mL^{-1}	Ingestion	Claessens et al. (2013)
		8.65×10^4 mL^{-1}		

(continued)

Table 10.8 (continued)

Organism	Size of ingested material	Exposure concentration	Effect	Source
Bay mussel (*Mytilus trossulus*)	10 µm PS	/	Ingestion	Ward et al. (2003)
Atlantic Sea scallop (*Placopecten magellanicus*)	15, 10, 16, 18, 20 µm PS	1.05 mL^{-1}	Ingestion, retention, egestion	Brilliant and MacDonald (2000, 2002)
Eastern oyster (*Crassostrea virginica*)	10 µm PS	1,000 mL^{-1}	Ingestion, egestion	Ward and Kach (2009)
Pacific oyster (*Crassostrea gigas*)	2, 6 µm PS	1,800 mL^{-1} for the 2 µm size 200 mL^{-1} for the 6 µm size	Increased filtration and assimilation, reduced gamete quality (sperm mobility, oocyte number and size, fecundation yield), slower larval rearing for larvae from MP exposed parents	Sussarellu et al. (2014)
Phylum Echinodermata				
Class Holothuridea				
Giant Californian sea cucumber (*Apostichopus californicus*)	10, 20 µm PS	2.4 μL^{-1}	Ingestion, retention	Hart (1991)
Stripped sea cucumber (*Thyonella gemmata*)	0.25–15 mm PVC shavings, nylon line, resin pellets	10 g PVC shavings, 60 g resin pellets	Selective ingestion	Graham and Thompson (2009)
Grey sea cucumber (*Holothuria* (*Halodeima*) *grisea*)				
Florida sea cucumber (*Holothuria floridana*)		2 g nylon line added to 600 mL of silica sand		
Orange footed sea cucumber (*Cucumaria frondosa*)				
Class Echinoidea				

(continued)

Table 10.8 (continued)

Organism	Size of ingested material	Exposure concentration	Effect	Source
Collector urchin (*Tripneustes gratilla*)	32–35 μm PE	1, 10, 100, 300 mL^{-1}	Ingestion, egestion	Kaposi et al. (2014)
Eccentric sand dollar (*Dendraster excentricus*)	10, 20 μm PS	2.4 μL^{-1}	Ingestion, retention	Hart (1991)
Sea urchin (*Strongylocentrotus* sp.)	10, 20 μm PS	2.4 μL^{-1}	Ingestion, retention	Hart (1991)
Class Ophiuroidea				
Crevice brittlestar (*Ophiopholis aculeata*)	10, 20 μm PS	2.4 μL^{-1}	Ingestion, retention	Hart (1991)
Class Asteriodea				
Leather star (*Dermasterias imbricata*)	10, 20 μm PS	2.4 μL^{-1}	Ingestion, retention	Hart (1991)
Phylum Arthropoda				
Subphylum Crustacea				
Class Maxillopoda				
Barnacle (*Semibalanus balanoides*)	20–2,000 μm	1 g L^{-1}	Ingestion	Thompson et al. (2004)
Subclass Copepoda				
Tigriopus japonicus	0.05 μm PS	9.1×10^{11} mL^{-1}	Ingestion, egestion, mortality, decreased fecundity	Lee et al. (2013)
	0.5 μm PS	9.1×10^{8} mL^{-1}		
	6 μm PS	5.25×10^{5} mL^{-1}		
Acartia (*Acanthacartia*) *tonsa*	10–70 μm	3,000–4,000 beads mL^{-1}	Ingestion, size selection	Wilson (1973)
Acartia spp.	10 μm PS	2,000 mL^{-1}	Ingestion	Setälä et al. (2014)
Eurytemora affinis	10 μm PS	1,000, 2,000, 10,000 mL^{-1}	Ingestion, egestion	Setälä et al. (2014)

(continued)

Table 10.8 (continued)

Organism	Size of ingested material	Exposure concentration	Effect	Source
Limnocalanus macrurus	10 µm PS	1,000, 2,000, 10,000 mL^{-1}	Ingestion	Setälä et al. (2014)
Temora longicornis	20 µm PS	100 mL^{-1}	Ingestion 10.7 ± 2.5 beads per individual	Cole et al. (2014a)
Calanus helgolandicus	20 µm PS	75 mL^{-1}	Egestion, ingestion	Cole et al. (2014b)
Class Malacostraca				
Orchestia gammarellus	20–2000 µm	1 g per individual (n = 150)	Ingestion	Thompson et al. (2004)
Talitrus saltator	10–45 µm PE	10 % weight food (0.06–0.09 g dry fish food)	Ingestion, egestion after 2 h	Ugolini et al. (2013)
Allorchestes compressa	11–700 µm	0.1 g	Ingestion, egestion within 36 h	Chua et al. (2014)
Neomysis integer	10 µm PS	2,000 spheres mL^{-1}	Ingestion	Setälä et al. (2014)
Mysis relicta	10 µm PS	2,000 spheres mL^{-1}	Ingestion, egestion	Setälä et al. (2014)
Shore crab (*Carcinus maenas*)	8–10 µm PS	4.0×10^4 L^{-1} ventilation 1.0×10^6 g^{-1} feeding	Ingestion through gills and gut, retention and excretion, no biological effects measured	Watts et al. (2014)
Norway lobster (Nephrops norvegicus)	5 mm PP fibres	10 fibres per 1 cm^3 fish	Ingestion	Murray and Cowie (2011)
Nephrops norvegicus	500–600 µm PE loaded with 10 µg of PCBs	150 mg microplastics in gelatin food	Ingestion, 100 % egestion. Increase of PCB level in the tissues. Same increase for positive control. No direct effect of microplastics	Devriese et al. (2014)
Class Branchipoda				
Bosmina coregoni	10 µm PS	2,000, 10,000 spheres mL^{-1}	Ingestion	Setälä et al. (2014)

(continued)

Table 10.8 (continued)

Organism	Size of ingested material	Exposure concentration	Effect	Source
Phylum Chordata				
Common goby (*Pomatoschistus microps*)	1–5 µm PE	18.4, 184 µg L^{-1}	Ingestion, modulation bioavailability or biotransformation of pyrene, decreased energy, inhibited AChE activity	Oliveira et al. (2013)
Atlantic cod (*Gadus morhua*)	2, 5 mm	/	Ingestion, egestion, 5 mm held for prolonged periods, emptying of plastics improved by food consumption additional meals	Dos Santos and Jobling (1992)
Japanese medaka (*Oryzias latipes*)	3 mm LDPE	Ground up as 10 % of diet	Liver toxicity, pathology, hepatic stress	Rochman et al. (2013)
Oryzias latipes	PE pellets	Two months chronic exposure	Altered gene expression, decreased choriogenin regulation in males and decreased vitellogenin and choriogenin in females	Rochman et al. (2014)
Seabass larvae (*Dicentrarchus labrax*)	10–45 µm PE	0–105 g^{-1}incorporated with food	Ingestion, no significant increase in growth, effect on survival of larvae. Possible gastric obstruction	Mazurais et al. (2014)

For comparison the size of ingested material increases within species

Table 10.9 Evidence of microplastic ingestion by field studies organisms

Species	Number studied	Percentage with plastic (%)	Mean number of particles per individual (±SD)	Type and size ingested (mm)	Location	Source
Phylum Mollusca						
Humbolt squid (*Dosidicus gigas*)	30	26.7	Max: 11	Nurdles: 3–5 mm	British Columbia, Canada	Braid et al. (2012)
Blue mussel (*Mytilus edulis*)	45	/	3.7 per 10 g mussel	Fibres 300–1,000 μm	Belgium, The Netherlands	De Witte et al. (2014)
Mytilus edulis	36	/	0.36 (±0.07) g^{-1}	5–25 μm	North Sea, Germany	Van Cauwenberghe and Janssen (2014)
Pacific oyster (*Crassostrea gigas*)	11	/	0.47 (±0.16) g^{-1}	5–25 μm	Atlantic Ocean	Van Cauwenberghe and Janssen (2014)
Phylum Crustacea						
Goosneck barnacle (*Lepas spp.*)	385	33.5	1–30	1.41	North Pacific	Goldstein and Goodwin (2013)
Norway lobster (*Nephrops norvegicus*)	120	83	/	/	Clyde, U.K.	Murray and Cowie (2011)
Brown shrimp (*Crangon crangon*)	110	/	11.5 fibres per 10 g shrimp	95 % fibres, 5 % films 300–1000 μm	Belgium	Devriese et al. (2014)
Phylum Chaetognatha						
Arrow worm (*Parasagitta elegans*)	1	100	/	0.1–3 mm PS	New England, USA	Carpenter et al. (1972)
Phylum Chordata						
Class Mammalia						
Harbor seal (*Phoca vitulina*)	100 stomachs, 107 intestines	S:11.2	Max: 8 items	>0.1	The Netherlands	Bravo Rebolledo et al. (2013)
		I: 1	Max: 7 items			

(continued)

Table 10.9 (continued)

Species	Number studied	Percentage with plastic (%)	Mean number of particles per individual (±SD)	Type and size ingested (mm)	Location	Source
Fur seal (*Arctocephalus* spp.)	145 scat	100	1–4 per scat	4.1	Macquarie Island, Australia	Eriksen and Burton (2003)
Class Reptilia						
Green turtle (*Chelonia mydas*)	24	/	Total: 11 pellets	<5 mm	Rio Grande do Sul, Brazil	Tourinho et al. 2010
Class Actinoptergii						
Order Atheriniformes						
Atlantic silversides (*Menidia menidia*)	9	33	/	0.1–3 mm PS	New England, USA	Carpenter et al. (1972)
Order Aulopiformes						
Longnosed lancetfish (*Alepisaurus ferox*)	144	24	2.7 (±2.0)	68.3 (±91.1)	North Pacific	Choy and Drazen (2013)
Order Beloniformes						
Cololabis saira	52	*35	3.2 (±3.05)	1–2.79	North Pacific	Boerger et al. (2010)
Order Clupeiformes						
Atlantic herring (*Clupea harengus*)	2	100	1	0.1–3 mm PS	New England, USA	Carpenter et al. (1972)
Clupea harengus	566	2	1–4	0.5–3	North Sea	Foekema et al. (2013)
Anchovy (*Stolephorus commersonnii*)	16	37.5	/	1.14–2.5	Alappuzha, India	Kripa et al. (2014)
Order Gadiformes						
Saithe (*Pollachius virens*)	1	100	1	0.1–3 mm PS	New England, USA	Carpenter et al. (1972)
Five-bearded rockling (*Ciliata mustela*)	113	0–10	/	1 mm PS	Severn Estuary, U.K.	Kartar (1976)

(continued)

Table 10.9 (continued)

Species	Number studied	Percentage with plastic (%)	Mean number of particles per individual (±SD)	Type and size ingested (mm)	Location	Source
Whiting (*Merlangius merlangus*)	105	6	1–3	1.7 (±1.5)	North Sea	Foekema et al. (2013)
Merlangius merlangus	50	32	1.75 (±1.4)	2.2 (±2.3)	English Channel	Lusher et al. (2013)
Haddock (*Melanogrammus aeglefinus*)	97	6	1.0	0.7 (±0.3)	North Sea	Foekema et al. (2013)
Cod (*Gadus morhua*)	80	13	1–2	1.2 (±1.2)	North Sea	Foekema et al. (2013)
Blue whiting (*Micromesistius poutassou*)	27	51.9	2.07 (±0.9)	2.0 (±2.4)	English Channel	Lusher et al. (2013)
Poor cod (*Trisopterus minutus*)	50	40	1.95 (±1.2)	2.2 (±2.2)	English Channel	Lusher et al. (2013)
Order Lampriformes						
Lampris sp. (*big eye*)	115	29	2.3 (±1.6)	49.1 (±71.1)	North Pacific	Choy and Drazen (2013)
Lampris sp. (*small eye*)	24	5	5.8 (±3.9)	48.8 (±34.5)	North Pacific	Choy and Drazen (2013)
Order Myctophiformes						
Hygophum reinhardtii	45	*35	1.3 (±0.71)	1–2.79	North Pacific	Boerger et al. (2010)
Loweina interrupta	28	*35	1.0	1–2.79	North Pacific	Boerger et al. (2010)
Myctophum aurolaternatum	460	*35	6.0 (±8.99)	1–2.79	North Pacific	Boerger et al. (2010)
Symbolophorus californiensis	78	*35	7.2 (±8.39)	1–2.79	North Pacific	Boerger et al. (2010)
Anderson's lanternfish (*Diaphus anderseni*)	13	15.4	1	/	North Pacific	Davison and Asch (2011)
Lanternfish (*Diaphus fulgens*)	7	28.6	1	/	North Pacific	Davison and Asch (2011)

(continued)

Table 10.9 (continued)

Species	Number studied	Percentage with plastic (%)	Mean number of particles per individual (±SD)	Type and size ingested (mm)	Location	Source
Boluin's lanternfish (*Diaphus phillipsi*)	1	100	1	Longest dimension 0.5	North Pacific	Davison and Asch (2011)
Coco's lanternfish (*Lobianchia gemellarii*)	3	33.3	1	/	North Pacific	Davison and Asch (2011)
Pearly lanternfish (*Myctophum nitidulum*)	25	16	1.5	Longest dimension 5.46	North Pacific	Davison and Asch (2011)
Order Perciformes						
White perch (*Morone americana*)	12	33	/	0.1–3 mm PS	New England, USA	Carpenter et al. (1972)
Bergall (*Tautogolabrus adspersus*)	6	<83	/	0.1–3 mm PS	New England, USA	Carpenter et al. (1972)
Goby (*Pomatoschistus minutus*)	200	0–25	/	1 mm PS	Severn estuary, U.K.	Kartar et al. (1976)
Stellifer brasiliensis	330	9.2	0.33–0.83	<1	Goiana estuary, Brazil	Dantas et al. (2012)
Stellifer stellifer	239	6.9	0.33–0.83	<1	Goiana estuary, Brazil	Dantas et al. (2012)
Eugerres brasilianus	240	16.3	1–5	1–5	Goiana estuary, Brazil	Ramos et al. (2012)
Eucinostomus melanopterus	141	9.2	1–5	1–5	Goiana estuary, Brazil	Ramos et al. (2012)
Diapterus rhombeus	45	11.1	1–5	1–5	Goiana estuary, Brazil	Ramos et al. (2012)
Horse mackerel (*Trachurus trachurus*)	100	1	1.0	1.52	North Sea	Foekema et al. (2013)

(continued)

Table 10.9 (continued)

Species	Number studied	Percentage with plastic (%)	Mean number of particles per individual (±SD)	Type and size ingested (mm)	Location	Source
Trachurus trachurus	56	28.6	1.5 (±0.7)	2.2 (±2.2)	English Channel	Lusher et al. (2013)
Yellowtail amberjack (*Seriola lalandi*)	19	10.5	1	0.5–10	North Pacific	Gassel et al. (2013)
Dragonet (*Callionymus lyra*)	50	38	1.79 (±0.9)	2.2 (±2.2)	English Channel	Lusher et al. (2013)
Red band fish (*Cepola macrophthalma*)	62	32.3	2.15 (±2.0)	2.0 (±1.9)	English Channel	Lusher et al. (2013)
Order Pleuronectiformes						
Winter flounder (*Pseudopleuronectes americanus*)	95	2.1	/	0.1–3 mm PS	New England, USA	Carpenter et al. (1972)
Flounder (*Platichthys flesus*)	/	/	/	1 mm PS	Severn estuary, U.K.	Kartar et al. (1973)
Platichthys flesus	1090	0–20.7	/	1 mm PS	Severn estuary, U.K.	Kartar et al. (1976)
Solenette (*Buglossidium luteum*)	50	26	1.23 (±0.4)	1.9 (±1.8)	English Channel	Lusher et al. (2013)
Thickback sole (*Microchirus variegatus*)	51	23.5	1.58 (±0.8)	2.2 (±2.2)	English Channel	Lusher et al. (2013)
Order Scorpaeniformes						
Grubby (*Myoxocephalus aenaeus*)	47	4.2	/	0.1–3 mm PS	New England, USA	Carpenter et al. (1972)
Striped searobin (*Prionotus evolans*)	1	100	1	0.1–3 mm PS	New England, USA	Carpenter et al. (1972)
Sea snail (*Liparis liparis liparis*)	220	0–25	/	1 mm PS	Severn estuary, U.K.	Kartar et al. (1976)

(continued)

Table 10.9 (continued)

Species	Number studied	Percentage with plastic (%)	Mean number of particles per individual (±SD)	Type and size ingested (mm)	Location	Source
Red gurnard (*Chelidonichthys cuculus*)	66	51.5	1.94 (±1.3)	2.1 (±2.1)	English Channel	Lusher et al. (2013)
Order Siluriformes						
Madamago sea catfish (*Cathorops spixii*)	60	18.3	0.47	1–4	Goiana estuary, Brazil	Possatto et al. (2011)
Catfish (*Cathorops* spp.)	60	33.3	0.55	1–4	Goiana estuary, Brazil	Possatto et al. (2011)
Pemecoe catfish (*Sciades herzbergii*)	62	17.7	0.25	1–4	Goiana estuary, Brazil	Possatto et al. (2011)
Order Stomiiformes						
Astronesthes indopacificus	7	*35	1.0	1–2.79	North Pacific	Boerger et al. (2010)
Hatchetfish (*Sternoptyx diaphana*)	4	25	1	Longest dimension 1.58 mm	North Pacific	Davison and Asch (2011)
Highlight hatchetfish (*Sternoptyx pseudobscura*)	6	16.7	1	Longest dimension 4.75 mm	North Pacific	Davison and Asch (2011)
Pacific black dragon (*Idiacanthus antrostomus*)	4	25	1	Longest dimension 0.5 mm	North Pacific	Davison and Asch (2011)
Order Zeiformes						
John Dory (*Zeus faber*)	46	47.6	2.65 (±2.5)	2.2 (±2.2)	English Channel	Lusher et al. (2013)

If mean not available range is reported. Standard deviation is reported where possible. *Represents percentage ingestion by total number of individuals, not separated by species mortality rates were observed following a two-generation chronic toxicity test (12.5 μg mL^{-1}) (Lee et al. 2013). Although a third of gooseneck barnacle (*Lepas* spp.) stomachs examined contained microplastics, no adverse effect was reported for these filter feeders (Goldstein and Goodwin 2013). Interestingly, the stomachs of mass stranded Humboldt squids (*Dosidicus gigas*) contained plastic pellets (Braid et al. 2012). This large predatory cephalopod usually feeds at depth between 200 and 700 m. The route of uptake is unclear; the squid may have fed directly on sunken pellets, or on organisms with pellets in their digestive system.

10.3.1.2 Benthic Invertebrates

A number of benthic invertebrates have been studied under laboratory conditions to investigate the consequences of microplastic ingestion (Table 10.8). Laboratory feeding and retention trials have focused on direct exposure of invertebrates to microplastic particles (as summarised by Cole et al. 2011; Wright et al. 2013a). Exposure studies demonstrated that benthic invertebrates including lugworms (*Arenicola marina*), amphipods (*Orchestia gammarellus*) and blue mussels (*Mytilus edulis*) feed directly on microplastics (Thompson et al. 2004; Wegner et al. 2012), and deposit-feeding sea cucumbers even selectively ingested microplastic particles (Graham and Thompson 2009).

Although microplastic uptake was recorded for a number of species, organisms appear to reject microplastics before digestion and excrete microplastics after digestion. Pseudofaeces production is a form of rejection before digestion but requires additional energetic cost. Furthermore, prolonged pseudofaeces production could lead to starvation (Wegner et al. 2012). On the other hand, polychaete worms, sea cucumbers and sea urchins are able to excrete unwanted materials through their intestinal tract without suffering obvious harm (Thompson et al. 2004; Graham and Thompson 2009; Kaposi et al. 2014). Adverse effects of microplastic ingestion were reported for lugworms: weight loss was positively correlated with concentration of spiked sediments (40–1300 μm polystyrene) (Besseling et al. 2013). Similarly, Wright et al. (2013b) recorded significantly reduced feeding activity and significantly decreased energy reserves in lugworm exposed to 5 % un-plasticised polyvinyl chloride (U-PVC). Supressed feeding reduced energy assimilation, compromising fitness. At the chronic exposure level, either fewer particles were ingested overall or a lack of protein coating on the U-PVC may have weakened particle adhesion to the worm's feeding apparatus.

Several studies have raised concern for microplastic retention and transference between organisms' tissues. For example, microplastics were retained in the digestive tract of mussels, and transferred to the haemolymph system after three days (Browne et al. 2008). However, negative effects on individuals were not detected. Von Moos et al. (2012) tracked particles of high density polyethylene (HDPE) into the lysosomal system of mussels after three hours of exposure; particles were taken up by the gills and transferred to the digestive tract and lysosomal system, again triggering an inflammatory immune response. It should be noted, however, that while these studies succeeded in determining the pathways of microplastics in organisms the exposure concentrations used to achieve this goal exceeded those expected in the field, such that the results have to be treated with care.

Studies of microplastic ingestion by benthic invertebrates in the field are less common than laboratory studies. Murray and Cowie (2011) identified fibres of monofilament plastics that could be sourced to fibres of trawls and fragments of plastic bags in the intestines of the commercially valuable Norway lobster (*Nephrops norvegicus*). These results indicated that normal digestive processes do not eliminate some of the filaments as they cannot pass through the gastric mill system. Norway lobsters have various feeding modes, including scavenging and predation, and are not adapted to cut flexible filamentous materials (Murray and Cowie 2011). The identification of microplastics in organisms that are caught for commercial purposes and subsequently consumed whole (including guts) highlights the potential human health implications. For example, field-caught brown shrimps (*Crangon crangon*) (Pott 2014) and farmed and store-brought bivalves (De Witte et al. 2014; Van Cauwenberghe and Janssen 2014) had microplastics in their digestive system.

Invertebrates could be used as indicator species for environmental contamination. Species such as *Nephrops* are able to integrate seasonal variation in microplastic abundance, providing an accurate measure of environmental contamination (Welden, *pers. comm.*). Additional studies are required to understand the flux of microplastic within benthic sediments and the interaction between different species of benthic infauna feeding in/or manipulating the sediment, such as bivalves and worms. Benthic infauna could ingest and/or excrete microplastics, the individuals or their faecal pellets may in turn be ingested by secondary consumers, thus affecting higher trophic levels.

10.3.1.3 Fish

Some of the earliest studies noting ingestion of microplastics by wild-caught fish include coastal species from the USA (Carpenter et al. 1972) and the U.K. (Kartar et al. 1973, 1976). More recent studies from the NPCG reported microplastic (fibres, fragments and films) ingestion by mesopelagic fish (Boerger et al. 2010; Davison and Asch 2011; Choy and Drazen 2013). Estuarine environments and their inhabitants are also prone to plastic contamination, which is hardly surprising given the riverine input (e.g. Morritt et al. 2014). Estuarine fish affected include catfish, Ariidae, (23 % of individuals examined) and estuarine drums, Scianenidae, (7.9 % of individuals examined), which spend their entire life cycle in estuaries (Possatto et al. 2011; Dantas et al. 2012). Similarly, 13.4 % of bottom-feeding fish (Gerreidae) from a tropical estuary in northeast Brazil contained microplastics in their stomachs (Ramos et al. 2012). The authors suggested that ingestion occurred during suction feeding on biofilms.

Lusher et al. (2013) reported microplastic polymers from 10 fish species from the English Channel. Of the 504 fish examined, 37 % had ingested a variety of microplastics, the most common being polyamide and the semi-synthetic material rayon. Similarly, Boerger et al. (2010) recorded microplastics in 35 % planktivorous fish examined from the NPCG (94 % of which were plastic fragments). Fish from the northern North Sea ingested microplastics at significantly lower levels (1.2 %) compared to those from the southern North Sea (5.4 %) (Foekema et al. 2013). All the studies cited suggest direct ingestion as the prime route of exposure, either targeted as food or mistaken for prey items. No adverse effects of ingestion were reported. Consequently, studies are required to follow the route of microplastic ingestion in fish, to assess if microplastics are egested in faecal pellets as seen in invertebrates. Dos Santos and Jobling (1992) showed that microplastic beads (2 mm) were excreted quickly following ingestion, whereas larger beads (5 mm) were held for prolonged periods of time. This implies that larger items of plastic might pose a greater risk following ingestion whereas smaller microplastics are likely to be excreted along with natural faeces.

10.3.1.4 Sea Birds

Numerous studies have dealt with the ingestion of marine debris by sea birds (see Kühn et al. 2015). Microplastics and small plastic items have been isolated from birds targeted deliberately for dietary studies, dead cadavers, regurgitated samples and faeces (Table 10.10). Nearly 50 species of Procellariiformes (fulmars, petrels, shearwaters, albatrosses), known to feed opportunistically at the sea surface had microplastics in their stomachs. Ingested microplastics appeared to comprise primarily of pellets and user-fragments (Ryan 1987; Robards et al. 1995) although there was a decrease in the proportion of pellets ingested by birds from the south Atlantic between the 1980s and 2006 (Ryan 2008). This trend is also true for short-tailed shearwater (*Puffinus tenuirostris*) from the North Sea (Vlietstra and Parga 2002). In this case however, the mass of industrial plastics (pellets) have decreased by half and the mass of plastic fragments has tripled (van Franeker et al. 2011). It is possible that the shift in the type of plastic consumed may be explained by fragmentation of larger user-plastics into smaller microplastics, the accumulation of user-plastic over time and a decreased disposal of industrial plastics (Thompson et al. 2004), or simply by a stronger awareness of the presence of microplastics.

Seabirds appear to be able to remove microplastics from their digestive tracts as regurgitation has been observed in the boluses of glaucous-winged gulls (*Larus glaucescens*) (Lindborg et al. 2012). However, this suggests that parents expose their offspring to plastics during feeding. Juveniles of northern fulmars (*Fulmarus glacialis*) had more plastic in their intestines than adults (Kühn and van Franeker 2012), with higher quantities in areas of higher fishing and shipping traffic (van Franeker et al. 2011). Still, as the majority of birds examined did not die as a direct result of microplastic uptake, it can be concluded that microplastic ingestion does not affect seabirds as severely as macroplastic ingestion. To date, there have been no studies demonstrating nanometre-sized microplastics in sea birds. This could be because it is extremely difficult to control laboratory conditions in terms of contamination.

10.3.1.5 Marine Mammals

Only one study on microplastic ingestion by marine mammals has been published to date. Bravo Rebolledo et al. (2013) recorded microplastics in stomachs (11 %, n = 100) and intestines (1 %, n = 107) of harbour seals (*Phoca vitulina*). Direct microplastic ingestion by other species of marine mammals has not been observed. However, larger plastics items were identified in the stomachs of numerous cetaceans (46 % of all species; Baulch and Perry 2014, see also Kühn et al. 2015). The frequency of microplastic uptake by marine mammals is hitherto unknown, but could occur through filter feeding, inhalation at the water-air interface, or via trophic transfer from prey items. As baleen whales (Mysticetes) strain water between baleen plates, to trap planktonic organisms and small fish (Nemoto 1970), they may incidentally trap microplastics. Thus, their feeding mode may render baleen whales more susceptible to direct microplastic ingestion than toothed (Odotocetes) or beaked whales (Ziphiids) which are active predators of squid and fish (Pauly et al. 1998). It is also likely that marine mammals are exposed to microplastic via trophic transfer from prey species. For example, microplastics were recorded from the scats of fur seals (*Arctocephalus* spp.) believed to originate from lantern fish (*Electrona subaspera*) (Eriksson and Burton 2003).

Cetaceans were suggested as sentinels for microplastic pollution (Fossi et al. 2012a; Galgani et al. 2014). However, it is notoriously difficult to extract and subsequently assess microplastics from cetacean stomachs, the often large size and decomposition rate of stomachs make sampling almost impossible. Furthermore, strandings are infrequent and unpredictable. Although adaption of sampling methods for smaller organisms such as fish and birds have the potential to be implemented, further work is necessary. The assessment of phthalate concentrations in the blubber of stranded fin whales (*Balaenoptera physalus*) (Fossi et al. 2012b, 2014) could serve as an indicator for the uptake of microplastics, but this raises other concerns as it is not possible to distinguish the origin of the phthalates. Exposure routes could be via micro- or macroplastics or simply from direct uptake of chemicals from the surrounding seawater into the blubber. Further work is essential to assess if microplastics significantly affect marine mammals.

10.3.1.6 Sea Turtles

Although all species of marine turtle ingest macroplastics (Derraik 2002; Schuyler et al. 2014; Kühn et al. 2015), only one study reported plastic pellets in the stomachs of the herbivorous green turtles (*Chelonia mydas*) (Tourinho et al. 2010).

Table 10.10 Evidence of microplastic ingestion by seabirds mean (±SD unless * = SE)

Species	Number studied	Percentage with plastic (%)	Mean number of particles per individual	Type and mean size ingested (mm)	Location	Source
Order Procellariiformes						
Family Procellariidae						
Kerguelen petrel (*Aphrodroma brevirostris*)	26	3.8	1	Pellet	North Island, New Zealand	Reid (1981)
Aphrodroma brevirostris	13	8	0.2	Pellets max. mass: 0.0083 g	Gough Island, U.K. South Atlantic	Furness (1985b)
Aphrodroma brevirostris	63	22.2	/	20 % pellet	Breeding grounds, Southern Ocean	Ryan (1987)
Aphrodroma brevirostris	28	7	/	Fragments and pellets 3–6 mm	Antarctica	Ainley et al. (1990)
Cory's shearwater (*Calonectris diomedea*)	7	42.8	/	Pellets 46 %	Breeding grounds, Southern Ocean	Ryan (1987)
Calonectris diomedea	147	24.5	Stomach = 2 Gizzard = 3.1	Beads 63.7 %	North Carolina, USA	Moser and Lee (1992)
Calonectris diomedea	5	100	/	<10	Rio Grande do Sul, Brazil	Colabuono et al. (2009)
Calonectris diomedea	85	83	8 (±7.9)	3.9 (±3.5)	Canary Islands, Spain	Rodríguez et al. (2012)
Calonectris diomedea	49	96	14.6 (±24.0)	2.5 (±6.0[a])	Catalan coast, Mediterranean	Codina-García et al. (2013)
Cape petrel (*Daption capense*)	18	83.3	/	Pellets 48 %	Breeding grounds, Southern Ocean	Ryan (1987)
Daption capense	30	33	1.0	5.0	Ardery Island, Antarctica	van Franeker and Bell (1988)
Daption capense	105	14	/	Fragments and pellets 3–6 mm	Antarctica	Ainley et al. (1990)
Northern fulmar (*Fulmarus glacialis*)	3	100	7.6	Pellets 1–4 mm	California, USA	Baltz and Morejohn (1976)

(continued)

Table 10.10 (continued)

Species	Number studied	Percentage with plastic (%)	Mean number of particles per individual	Type and mean size ingested (mm)	Location	Source
Fulmarus glacialis	79	92	11.9	Pellets 50 %	The Netherlands, Arctic	van Franeker (1985)
Fulmarus glacialis	8	50	3.9	Pellets	St. Kilda, U.K.	Furness (1985a)
Fulmarus glacialis	13	92.3	10.6	Pellets	Foula, U.K.	Furness (1985a)
Fulmarus glacialis	1	100	1	Pellet, 4 mm	Oregon, USA	Bayer and Olson (1988)
Fulmarus glacialis	44	86.4	Stomach = 3 Gizzard = 14	Beads 91.9 %	North Carolina, USA	Moser and Lee (1992)
Fulmarus glacialis	19	84.2	Max: 26	Pellets 36 %	Alaska, USA	Robards et al. (1995)
Fulmarus glacialis	3	100	7.7	Pellets 48 %	Offshore, eastern North Pacific	Blight and Burger (1997)
Fulmarus glacialis	15	36	3.6 (±2.7)	7 (±4.0)	Davis Strait, Canadian Arctic	Mallory et al. (2006)
Fulmarus glacialis	1295	95	14.6 (±2.0*)–33.2 (±3.3*)	>1.0	North Sea	van Franeker et al. (2011)
Fulmarus glacialis	67	92.5	36.8 (±9.8*)	>0.5	Eastern North Pacific	Avery-Gomm et al. (2012)
Fulmarus glacialis	58	79	6.0 (±0.9*)	>1.0	Westfjords, Iceland	Kühn and van Franeker (2012)
Fulmarus glacialis	176	93	26.6 (±37.5)	Fragments and pellets	Nova Scotia, Canada	Bond et al. (2014)
Antarctic fulmar (*Fulmarus glacialoides*)	84	2	/	Fragments and pellets 2–6 mm	Antarctica	Ainley et al. (1990)
Fulmarus glacialoides	9	79	/	<10	Rio Grande do Sul, Brazil	Colabuono et al. (2009)
Blue petrel (*Halobaena caerulea*)	27	100	/	Pellets	New Zealand	Reid (1981)
Halobaena caerulea	74	85.1	/	Pellets 69 %	Southern Ocean	Ryan (1987)

(continued)

Table 10.10 (continued)

Species	Number studied	Percentage with plastic (%)	Mean number of particles per individual	Type and mean size ingested (mm)	Location	Source
Halobaena caerulea	62	56	/	Fragments and pellets 3–6 mm	Antarctica	Ainley et al. (1990)
Prions *Pachyptila* spp.	/	/	/	Pellets	Gough Island, U.K. South Atlantic	Bourne and Imber (1982)
Salvin's prion (*Pachyptila salvini*)	663	20	/	Pellets 2.5–3.5 mm	Wellington, New Zealand	Harper and Fowler (1987)
Pachyptila salvini	31	51.6	/	Pellets 49 %	Breeding grounds, Southern Ocean	Ryan (1987)
Thin-billed prion (*Pachyptila belcheri*)	152	6.6	/	Pellets 2.5–3.5 mm	Wellington, New Zealand	Harper and Fowler (1987)
Pachyptila belcheri	32	68.7	/	Pellets 38 %	Breeding grounds, Southern Ocean	Ryan (1987)
Broad-billed prion (*Pachyptila vittata*)	31	39	0.6	Pellets max mass: 0.066	Gough Island, U.K. South Atlantic	Furness (1985b)
Pachyptila vittata	310	16.5	/	Pellets 2.5–3.5 mm	Wellington, New Zealand	Harper and Fowler (1987)
Pachyptila vittata	137	20.4	/	56 % pellet	Breeding grounds, Southern Ocean	Ryan (1987)
Pachyptila vittata	69	10	/	Fragments and pellets 3–6 mm	Antarctica	Ainley et al. (1990)
Pachyptila vittata	149	/	1987–1989 [b]1.73 ± 3.58	Pellets 43.6 %	Breeding grounds, Southern Ocean	Ryan (2008)
Pachyptila vittata	86	/	1999 [b]2.93 ± 3.80	Pellets 37.3 %	Breeding grounds, Southern Ocean	Ryan (2008)
Pachyptila vittata	95	/	2004 [b]2.66 ± 5.34	Pellets 15.4 %	Breeding grounds, Southern Ocean	Ryan (2008)

(continued)

Table 10.10 (continued)

Species	Number studied	Percentage with plastic (%)	Mean number of particles per individual	Type and mean size ingested (mm)	Location	Source
Antarctic prion (*Pachyptila desolata*)	35	14.3	/	Pellets 2.5–3.5 mm	Wellington, New Zealand	Harper and Fowler (1987)
Pachyptila desolata	88	47.7	/	Pellets 53 %	Breeding grounds, Southern Ocean	Ryan (1987)
Pachyptila desolata	2	100	1.0	6–8.1 mm	Heard Island, Australia	Auman et al. (2004)
Fairy prion (*Pachyptila turtur*)	105	96.2	/	Pellets 2.5–3.5 mm	Wellington, New Zealand	Harper and Fowler (1987)
Snow petrel (*Pagodroma nivea*)	363	1	/	Fragments and pellets 3–6 mm	Antarctica	Ainley et al. (1990)
White-chinned petrel (*Procellaria aequinoctialis*)	193	/	1983-1985 [b]1.66 (±3.04)	Pellets 38.2 %	Breeding grounds, Southern Ocean	Ryan (1987, 2008)
Procellaria aequinoctialis	526	/	2005–2006 [b]1.39 (±3.25)	16.2 % pellets	Breeding grounds, Southern Ocean	Ryan (2008)
Procellaria aequinoctialis	41	/	/	<10	Rio Grande do Sul, Brazil	Colabuono et al. (2009)
Procellaria aequinoctialis	34	44	/	<10	Rio Grande do Sul, Brazil	Colabuono et al. (2010)
Spectacled petrel (*Procellaria conspicillata*)	3	33	/	<10	Rio Grande do Sul, Brazil	Colabuno et al. (2010)
Procellaria conspicillata	9	/	/	<10	Rio Grande do Sul, Brazil	Colabuono et al. (2009)
Tahiti petrel (*Pseudobulweria rostrata*)	121	<1	1	Fragments	Tropical, North Pacific	Spear et al. (1995)
Atlantic petrel (*Pterodroma incerta*)	13	8	0.1	Pellets max mass: 0.0053 g	Gough Island, U.K. South Atlantic	Furness (1985b)

(continued)

Table 10.10 (continued)

Species	Number studied	Percentage with plastic (%)	Mean number of particles per individual	Type and mean size ingested (mm)	Location	Source
Pterodroma incerta	20	5	/	Pellets	Breeding grounds, Southern Ocean	Ryan (1987)
Great-winged petrel (*Pterodroma macroptera*)	13	7.6	/	Pellets	Breeding grounds, Southern Ocean	Ryan (1987)
Soft-plumaged petrel (*Pterodroma mollis*)	29	20.6	/	Pellets 22 %	Breeding grounds, Southern Ocean	Ryan (1987)
Pterodroma mollis	18	6	0.1	Pellets max. mass: 0.014 g	Gough Island, U.K. South Atlantic	Furness (1985b)
Juan Fernández petrel (*Pterodroma externa*)	183	<1	1	Pellets 3–5 mm	Offshore, North Pacific	Spear et al. (1995)
White-necked petrel (*Pterodroma cervicalis*)	12	8.3	5	Fragments 3–4 mm	Offshore, North Pacific	Spear et al. (1995)
Pycroft's petrel (*Pterodroma pycrofti*)	5	40	2.5 (±0.7)	Fragments Pellets 3–5 mm	Offshore, North Pacific	Spear et al. (1995)
White-winged petrel (*Pterodroma leucoptera*)	110	11.8	2.2 (±3.0)	Fragments 2–5 mm	Offshore, North Pacific	Spear et al. (1995)
Collared petrel (*Pterodroma brevipes*)	3	66.7	1	Pellets 2–5 mm	Offshore, North Pacific	Spear et al. (1995)
Black-winged petrel (*Pterodroma nigripenni*)	66	4.5	3.0 (±3.5)	Fragments 3–5 mm	Offshore, North Pacific	Spear et al. (1995)
Stejneger's petrel (*Pterodroma longirostris*)	46	73.9	6.8 (± 8.6)	Fragments and pellets 2–5 mm	Offshore, North Pacific	Spear et al. (1995)
Audubon's shearwater (*Puffinus lherminieri*)	119	5	Stomach = 1 Gizzard = 4.4	Beads 50 %	North Carolina, USA	Moser and Lee (1992)
Little shearwater (*Puffinus assimilis*)	13	8	0.8	Pellets max. mass: 0.12 g	Gough Island, U.K. South Atlantic	Furness (1985b)

(continued)

Table 10.10 (continued)

Species	Number studied	Percentage with plastic (%)	Mean number of particles per individual	Type and mean size ingested (mm)	Location	Source
Buller's shearwater (*Puffinus bulleri*)	3	100	8.5 (±8.6)	Fragments and pellets 2–8 mm	Tropical, North Pacific	Spear et al. (1995)
Pink-footed shearwater (*Puffinus creatopus*)	5	20	2.2	Pellets 1–4 mm	California, USA	Baltz and Morejohn (1976)
Great shearwater (*Puffinus gravis*)	24	100	/	Beads	Briar Island, Nova Scotia	Brown et al. (1981)
Puffinus gravis	13	85	12.2	Pellets max. mass: 1.13 g	Gough Island, U.K. South Atlantic	Furness (1985b)
Puffinus gravis	55	63.6	Stomach = 1 Gizzard = 13.2	Beads 91.2 %	North Carolina, USA	Moser and Lee (1992)
Puffinus gravis	50	66	1983–1985 [b]16.5 (±19.0)	Pellets 64.3 %	Breeding grounds, Southern Ocean	Ryan (1987, 2008)
Puffinus gravis	53	/	2005–2006 [b]11.8 (±18.9)	Pellets 11.3 %	Breeding grounds, Southern Ocean	Ryan (2008)
Puffinus gravis	19	89	/	<10 mm	Rio Grande do Sul, Brazil	Colabuono et al. (2009)
Puffinus gravis	6	100	/	Pellets < 3.2–5.3 mm	Rio Grande do Sul, Brazil	Colabuono et al. (2009)
Puffinus gravis	84	88	11.8 (±16.9)	Fragments and pellets	Nova Scotia, Canada	Bond et al. (2014)
Sooty shearwater (*Puffinus griseus*)	21	43	5.05	Pellets 1–4 mm	California, USA	Baltz and Morejohn (1976)
Puffinus griseus	5	100	/	Beads	Briar Island, Nova Scotia, Canada	Brown et al. (1981)
Puffinus griseus	36	58.3	11.4 (±12.2)	Fragments and pellets 3–20 mm	Tropical, North Pacific	Spear et al. (1995)
Puffinus griseus	218	88.5	/	Pellets 25.4 %	Offshore, North Pacific	Ogi (1990)

(continued)

Table 10.10 (continued)

Species	Number studied	Percentage with plastic (%)	Mean number of particles per individual	Type and mean size ingested (mm)	Location	Source
Puffinus griseus	20	75	3.4	Pellets 38 %	Offshore eastern North Pacific	Blight and Burger (1997)
Puffinus griseus	50	72	2.48 (±2.7)	Fragments and pellets	Nova Scotia, Canada	Bond et al. (2014)
Balearic shearwater (*Puffinus mauretanicus*)	46	70	2.5 (±2.9)	3.5 (±10.5[a])	Catalan coast, Mediterranean	Codina-García et al. (2013)
Christmas shearwater (*Puffinus nativitatis*)	5	40	1	Pellet 3–5 mm	Tropical, North Pacific	Spear et al. (1995)
				Fragment 4 mm		
Wedge-tailed shearwater (*Puffinus pacificus*) dark phase	23	4	2.5 (±2.1)	Fragments	Tropical, North Pacific	Spear et al. (1995)
	62	24.2	3.5 (±2.7)	Fragments and pellets		
Puffinus pacificus	20	60	Max: 11	Pellets 2–4 mm	Hawaii, USA	Fry et al. (1987)
Manx shearwater (*Puffinus puffinus*)	10	30	0.4	Pellets	Rhum, U.K.	Furness (1985a)
Puffinus puffinus	25	60	/	<10 mm	Rio Grande do Sul, Brazil	Colabuono et al. (2009)
Puffinus puffinus	6	17	/	Fragments	Rio Grande do Sul, Brazil	Colabuono et al. (2009)
Short-tailed shearwater (*Puffinus tenuirostris*)	6	100	19.8	Pellets 1–4 mm	California, USA	Baltz and Morejohn (1976)
Puffinus tenuirostris	324	81.8	/	Pellets 67.2 %	Offshore, North Pacific	Ogi (1990)
Puffinus tenuirostris	330	83.9	5.8 (±0.4*)	Pellets 2–5 mm	Bering Sea, North Pacific	Vlietstra and Parga (2002)
Puffinus tenuirostris	5	80	/	Fragments and pellets	Alaska, USA	Robards et al. 1995
Puffinus tenuirostris	99	100	15.1 (±13.2)	>2 mm	Offshore, North Pacific	Yamashita et al. (2011)
Puffinus tenuirostris	129	67	Adults: 4.5	Fragments 0.97–80.8 mm	North Stradbroke Island, Australia	Acampora et al. (2013)
			Juvenile: 7.1			

(continued)

Table 10.10 (continued)

Species	Number studied	Percentage with plastic (%)	Mean number of particles per individual	Type and mean size ingested (mm)	Location	Source
Puffinus tenuirostris	12	100	27	>2 mm	Offshore, North Pacific	Tanaka et al. (2013)
Yelkouan shearwater (*Puffinus yelkouan*)	31	71	4.9 (±7.3)	4.0 (±13.0[a])	Catalan coast, Mediterranean	Codina-García et al. (2013)
Antarctic petrel (*Thalassoica antarctica*)	184	<1	/	Fragments and pellets 3–6 mm	Antarctica	Ainley et al. (1990
Family Hydrobatidae						
White-bellied storm petrel (*Fregetta grallaria*)	13	38	1.2	Pellets max. mass: 0.042 g	Gough Island, U.K. South Atlantic	Furness (1985b)
Fregetta grallaria	296	<1	1	Fragment	Offshore, North Pacific	Spear et al. (1995)
Fregetta grallaria	318	/	1987–89 [b]0.63 ± 1.13	Pellets 33.3 %	Breeding grounds, Southern Ocean	Ryan (2008)
Fregetta grallaria	137	/	1999 [b]0.63 ± 1.37	Pellets 20.9 %	Breeding grounds, Southern Ocean	Ryan (2008)
Fregetta grallaria	95	/	2004 [b]0.72 ± 1.87	Pellets 16.2 %	Breeding grounds, Southern Ocean	Ryan (2008)
Grey-backed storm petrel (*Garrodia nereis*)	11	27	0.3	Pellets max. mass: 0.010 g	Gough Island, U.K. South Atlantic	Furness (1985b)
Garrodia nereis	12	8.3	/	Pellets	Breeding grounds, Southern Ocean	Ryan (1987)
Fork-tailed storm petrel (*Oceanodroma furcata*)	/	/	/	<5 mm	Aleutian Islands, USA	Ohlendorf et al. (1978)
Oceanodroma furcata	21	85.7	Max.: 12	Pellets 22 %	Alaska, USA	Robards et al. (1995)
Oceanodroma furcata	7	100	20.1	Pellets 16 %	Offshore, eastern North Pacific	Blight and Burger (1997)
Leach's storm petrel (*Oceanodroma leucorhoa*)	15	40	1.66 (±1.2)	2–5 mm	Newfoundland, Canada	Rothstein (1973)

(continued)

Table 10.10 (continued)

Species	Number studied	Percentage with plastic (%)	Mean number of particles per individual	Type and mean size ingested (mm)	Location	Source
Oceanodroma leucorhoa	17	58.8	2.9	Pellets	St. Kilda, U.K.	Furness (1985a)
Oceanodroma leucorhoa	354	19.8	3.5 (±2.6)	Fragments and pellets 2–5 mm	Offshore, North Pacific	Spear et al. (1995)
Oceanodroma leucorhoa	64	48.4	Max.: 13	Monofilament line, fragments, pellets	Alaska, USA	Robards et al. (1995)
Wilson's storm petrel (*Oceanites oceanicus*)	20	75	4.4	2.9 mm	Ardery Island, Antarctica	van Franeker and Bell (1988)
Oceanites oceanicus	91	19	/	Fragments and pellets 3–6 mm	Antarctica	Ainley et al. (1990)
Oceanites oceanicus	133	38.3	Stomach = 1.4 Gizzard = 5.4	26 % beads	North Carolina, USA	Moser and Lee (1992)
White-faced storm petrel (*Pelagodroma marina*)	19	84	11.7	Pellets max. mass: 0.34 g	Gough Island, U.K. South Atlantic	Furness (1985b)
Pelagodroma marina	15	73.3	13.2 ± 9.5	Pellets 2–5 mm	Offshore, North Pacific	Spear et al. (1995)
Pelagodroma marina	24	20.8	/	Pellets 41 %	Southern Hemisphere	Ryan (1987)
Pelagodroma marina	253		1987–89 [b]3.98 ± 5.45	Pellets 69.6 %	Breeding grounds, Southern Ocean	Ryan (2008)
Pelagodroma marina	86	/	1999 [b]4.06 ± 5.93	Pellets 37.5 %	Breeding grounds, Southern Ocean	Ryan (2008)
Pelagodroma marina	5	/	2004 [b]2.52 ± 4.43	Pellets 13.5 %	Breeding grounds, Southern Ocean	Ryan (2008)
Family Diomedeidae						
Sooty albatross (*Phoebetria fusca*)	73	42.7	/	Pellets 34 %	Breeding grounds, Southern Ocean	Ryan (1987)
Laysan albatross (*Phoebastria immutabilis*)	/	52	/	Pellets 2–5 mm	Hawaiian Islands, USA	Sileo et al. (1990)

(continued)

Table 10.10 (continued)

Species	Number studied	Percentage with plastic (%)	Mean number of particles per individual	Type and mean size ingested (mm)	Location	Source
Black-footed albatross (*Phoebastria nigripes*)	/	12	/	Pellets 2–5 mm	Hawaiian Islands, USA	Sileo et al. (1990)
Phoebastria nigripes	3	100	5.3	Pellets 50 %	Offshore, eastern North Pacific	Blight and Burger (1997)
Black-browed albatross (*Thalassarche melanophris*)	2	100	3	Pellets 50 %	Rio Grande do Sul, Brazil	Tourinho et al. (2010)
Order Charadriiformes						
Family Laridae						
Audouin's gull (*Larus audouinii*)	15	13	49.3 (±77.7)	2.5 (±5.0*)	Catalan coast, Mediterranean	Codina-García et al. (2013)
Glaucous-winged gull (*Larus glaucescens*)	589 boluses	12.2	/	<10 mm	Protection Island, USA	Lindborg et al. (2012)
Heermann's Gull (*Larus heermanni*)	15	7	1	Pellets 1–4 mm	California, USA	Baltz and Morejohn (1976)
Mediterranean gull (*Larus melanocephalus*)	4	25	3.7 (±7.5)	3.0 (±5.0*)	Catalan coast, Mediterranean	Codina-García et al. (2013)
Yellow-legged gull) (*Larus michahellis*)	12	33	0.9 (±1.5)	2.0 (±8.0*)	Catalan coast, Mediterranean	Codina-García et al. (2013)
Red-legged kittiwake (*Rissa brevirostris*)	15	26.7	/	Pellets 5 % Mean size: 5.87 mm	Alaska, USA	Robards et al. (1995)
Black-legged kittiwake (*Rissa tridactyla*)	8	8	4.0	Pellets 1–4 mm	California, USA	Baltz and Morejohn (1976)
Rissa tridactyla	256	7.8	Max.: 15	Pellets 5 %	Alaska, USA	Robards et al. (1995)
Rissa tridactyla	4	50	1.2 (±1.9)	3.0 (±5.0*)	Catalan coast, Mediterranean	Codina-García et al. (2013)

(continued)

Table 10.10 (continued)

Species	Number studied	Percentage with plastic (%)	Mean number of particles per individual	Type and mean size ingested (mm)	Location	Source
Family Alcidae						
Parakeet auklet (*Aethia psittacula*)	/	/	/	<5 mm	Aleutians Islands, USA	Ohlendorf et al. (1978)
Aethia psittacula	208	93.8	17.1	Pellets > 80 % 4.08 mm	Alaska, USA	Robards et al. (1995)
Tufted puffin (*Fratercula cirrhata*)	489	24.5	Max.: 51	Pellets 90 % 4.10 mm	Alaska, USA	Robards et al. (1995)
Fratercula cirrhata	9	89	3.3	Pellets 43 %	Offshore, North Pacific	Blight and Burger (1997)
Horned puffin (*Fratercula corniculata*)	/	/	/	<5 mm	Aleutian Islands, USA	Ohlendorf et al. (1978)
Fratercula corniculata	120	36.7	Max.:14	Pellets 40 % 5.03 mm	Alaska, USA	Robards et al. (1995)
Fratercula corniculata	2	50	1.5	Pellets	Offshore, North Pacific	Blight and Burger (1997)
Common murre (*Uria aalge*)	1	100	2011–2012 1	6.6 (±2.2)	Newfoundland, Canada	Bond et al. (2013)
Thick-billed murre (*Uria lomvia*)	186	11	0.2 (±0.8)	4.5 (±3.8)	Canadian Arctic	Provencher et al. (2010)
Uria lomvia	3	100	2011–2012 1	6.6 (±2.2)	Newfoundland, Canada	Bond et al. (2013)
Uria lomvia	1249	7.7	1985–1986 0.14 (±0.7*)	10.1 (±7.4)	Newfoundland, Canada	Bond et al. (2013)
Family Stercorariidae						
Brown skua (*Stercorarius antarcticus*)	494	22.7	/	Pellets 67 %	Breeding grounds, Southern Ocean	Ryan (1987)
Tristan skua (*Stercorarius hamiltoni*)	11	9	0.3 Max.: 3	Pellets Max. mass: 0.064 g	Gough Island, U.K. South Atlantic	Furness (1985b)

(continued)

Table 10.10 (continued)

Species	Number studied	Percentage with plastic (%)	Mean number of particles per individual	Type and mean size ingested (mm)	Location	Source
Long-tailed skua (*Stercorarius longicaudus*)	2	50	5	Fragments and pellets	Offshore, eastern North Pacific	Spear et al. (1995)
Arctic skua (*Stercorarius parasiticus*)	2	50	/	Pellets 50 %	Breeding grounds, Southern Ocean	Ryan (1987)
Family Scolopacidae						
Grey phalarope (*Phalaropus fulicarius*)	20	100	Max.: 36	Beads 1.7–4.4 mm	California, USA	Bond (1971)
Phalaropus fulicarius	7	85.7	5.7	Pellets	California, USA	Connors and Smith (1982)
Phalaropus fulicarius	2	50	/	Pellets	Breeding grounds, Southern Ocean	Ryan (1987)
Phalaropus fulicarius	55	69.1	Stomach = 1 Gizzard = 6.7	Beads 16.7 %	North Carolina, USA	Moser and Lee (1992)
Red-necked phalarope (*Phalaropus lobatus*)	36	19.4	Stomach = 0 Gizzard = 3.7	Beads 16.7 %	North Carolina, USA	Moser and Lee (1992)
Family Sternidae						
Sooty tern (*Onychoprion fuscatus*)	64	1.6	2	Pellets 4 mm	Offshore, eastern North Pacific	Spear et al. (1995)
White tern (*Gygis alba*)	8	12.5	5	Fragments 3–4 mm	Offshore, eastern North Pacific	Spear et al. (1995)
Order Suliformes						
Family Phalacrocoracidae						
Macquarie Shag (*Phalacrocorax (atriceps) purpurascens*)	64 boluses	7.8	1 per bolus	Polystyrene spheres	Macquarie Island, Australia	Slip et al. (1990)

[a]Median (±0.8) 95 % confidence intervals. Plastics found in total of 28 % birds
[b]This is total mean abundance of plastics, including pellets and user fragments; sizes of pellets are assumed to be 2–5 mm, according to recent literature

It is highly likely that other species of sea turtle also ingest microplastics incidentally or directly, depending on their feeding habits (Schuyler et al. 2014). Neonatal and oceanic post-hatchlings are generalist feeders (Bjorndal 1997), targeting plankton from surface waters and microplastic uptake may occur. Trophic transfer from prey items could be a pathway to larger individuals; loggerhead (*Caretta caretta*) and Kemp's Ridley (*Lepidochelys kempii*) turtles are carnivores, feeding on crustaceans and bivalves (Bjorndal 1997), which ingest microplastics (e.g. Browne et al. 2008). Flatbacks (*Natator depressa*) are also carnivores but feed on soft bodied invertebrates (Bjorndal 1997), including sea cucumbers, which again, ingest microplastics (Graham and Thompson 2009). Leatherbacks (*Dermochelys coriacea*) feed on gelatinous organisms (Bjorndal 1997) and are thus more likely to ingest macroplastics because of their size and similarity to prey items. If microplastics are ingested they could affect sea turtle growth and development if they are not egested. Additional work is required to understand whether turtles actively ingest microplastics, and if so, the extent of the harm caused.

10.3.2 Trophic Transfer

Absorption and ingestion of microplastics by organisms from the primary trophic level, e.g. phytoplankton and zooplankton, could be a pathway into the food chain (Bhattacharya et al. 2010). Many species of zooplankton undergo a diurnal migration. Migrating zooplankton could be considered a vector of microplastic contamination to greater depths of the water column and its inhabitants, either through predation or the production of faecal pellets sinking to the seafloor (Wright et al. 2013a). Only a few studies deal with the potential for microplastics to be transferred between trophic levels following ingestion. Field observation highlighted the presence of microplastics in the scat of fur seals (*Arctocephalus* spp.) and Eriksson and Burton (2003) suggested that microplastics had initially been ingested by the fur seals' prey, the plankton feeding Mycophiids. In feeding experiments, Farrell and Nelson (2013) identified microplastic in the gut and haemolymph of the shore crab (*Carcinus maenas*), which had previously been ingested by blue mussels (*Mytilus edulis*). There was large variability in the number of microspheres in tissues samples, and the results have to be treated with caution as the number of individuals was low and the exposure levels used exceeded those from the field. Similarly, *Nephrops*-fed fish, which had been seeded with microplastic strands of polypropylene rope were found to ingest but not to excrete the strands (Murray and Cowie 2011), again implying potential trophic transfer. As mentioned above, microplastics were also detected in cod, whiting, haddock, bivalves and brown shrimp, which are consumed by humans and raises concerns about trophic transfer to humans and human exposure (see Galloway 2015). Further studies are required to increase our understanding of trophic transfer.

10.3.3 Microplastic Effect on Habitats

Surfaces of buoyant microplastics provide habitats for rafting organisms. For example, pelagic insects (*Halobates micans* and *H. sericeus*) utilize microplastic pellets for oviposition (Goldstein et al. 2012; Majer et al. 2012). Indeed, Goldstein et al. (2012) attributed an overall increase in *H. sericeus* and egg densities in the NPCG to high concentrations of microplastics. Likewise, plastics serve as a floating habitat for bacterial colonisation (Lobelle and Cunliffe 2011). Microorganisms including Bacillus bacteria (mean: 1664 ± 247 individuals mm^{-2}) and pennate diatoms (mean: 1097 ± 154 individuals mm^{-2}) were identified on plastic items from the North Pacific gyre (Carson et al. 2013). These studies suggest that microplastics affect the distribution and dispersal of marine organisms and may represent vectors to alien invasion. Plastics colonised by pathogenic viruses or bacteria may spread the potential for disease, but there is currently no evidence to support this hypothesis.

Microplastic buried in sediments could have fundamental impacts on marine biota as they increase the permeability of sediment and decrease thermal diffusivity (Carson et al. 2011). This may affect temperature-dependent processes. For example, altered temperatures during incubation can bias the sex ratios of sea turtle eggs. At 30 °C, equal numbers of males and female embryos develop, whereas at temperatures <28 °C all embryos become male (Yntema and Mrosovsky 1982). With microplastics in sediments it will take longer to reach maximum temperatures because of its increased permeability. Therefore, eggs may require a longer incubation period, with more male hatchlings because of the insulating effect. Microplastic concentrations as low as 1.5 can decrease maximum temperatures by 0.75 °C (Carson et al. 2011), which has important implications for sexual bias in sea turtles including loggerhead turtles (*Caretta caretta*) and hawksbill turtles (*Eretmochelys imbricata*) (Yntema and Mrosovsky 1982; Mrosovsky et al. 1992). Changes in the sediment temperatures could also affect infaunal organisms as it may affect enzymatic and other physiological processes, feeding and growth rates, locomotory speeds, reproduction and ultimately population dynamics. However, this remains speculative until further researched.

10.3.4 Summary

Microplastic ingestion has been documented for a range of marine vertebrates and invertebrates (Fig. 10.1). Interactions were recorded primarily during controlled laboratory studies, but results from field sampling of wild populations also indicate microplastic ingestion. In the case of some invertebrates, adverse physiological and biological effects were reported. The biological repercussions depend on to the size of microplastics with smaller sizes having greater effects on organisms at the cellular level. In the micrometre range, plastics are readily ingested and egested whereas

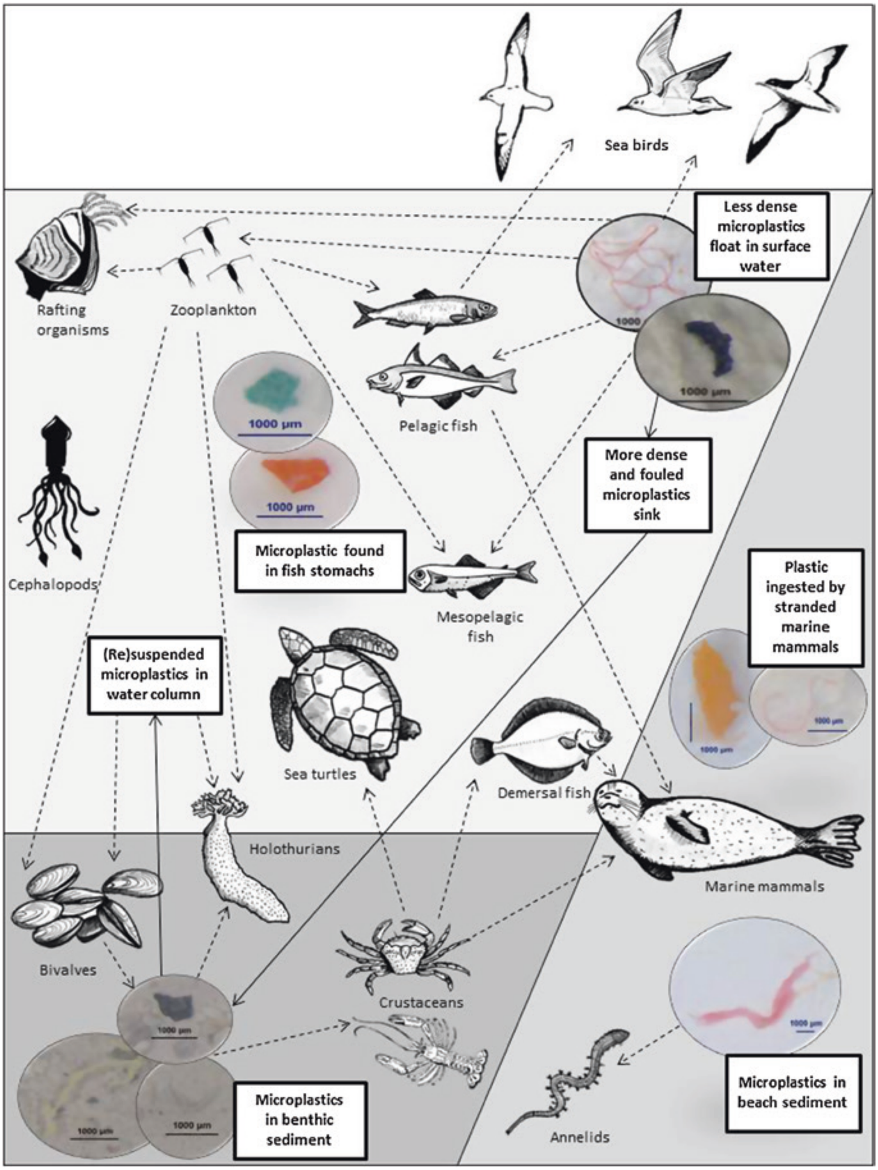

Fig. 10.1 Microplastic interactions in the marine environment including environmental links (*solid arrows*) and biological links (*broken arrows*), which highlights potential trophic transfer (Photos of microplastics: A. Lusher)

nanometre-sized plastics can pass through cell membranes. Acute exposure experiments demonstrated significant biological effects including weight loss, reduced feeding activity, increased phagocytic activity and transference to the lysosomal (storage) system. Larger microplastics (2–5 mm) may take longer to pass from the stomachs of organisms and could be retained in the digestive system, potentially increasing the exposure time to adsorbed toxins (see Rochman 2015).

It is important to determine the ecological effects of microplastic ingestion. Studies are required to assess the contamination of more species of fish, marine mammals and sea turtles, as well as consequences of microplastic uptake and retention. Further research is necessary to determine the limits of microplastic translocation between tissues, and assess the differences between multiple polymer types and shapes. It is likely that additional species of invertebrate ingest microplastics in wild populations, as fibres and fragments found in the field are actively selected in experiments. Although some organisms appear to be able to differentiate between microplastics and prey, and microplastic excretion has been recorded. Without knowledge of retention and egestion rates of field populations, it is difficult to deduce ecological consequences. There is some evidence to suggest that microplastics enter the food chain and transfer of microplastics between trophic levels implies bioaccumulation and biomagnification. Despite concerns raised by ingestion in the marine environment, the effects of microplastic ingestion in natural populations and the implications for food webs are not understood. Such knowledge is crucial in order to be able to develop and implement effective management strategies (Thompson et al. 2009). Additional studies are required to understand the flux of microplastic from benthic sediments to the infauna. Lastly, microplastics provide open ocean habitats for colonisation by invertebrates, bacteria and viruses. As a result, these organisms can be transported over large distances by ocean currents and/or through the water column (Kiessling et al. 2015).

10.4 Conclusion

Microplastics have been found in almost every marine habitat around the world, and plastic density along with ocean currents appears to have a significant effect on their distribution. Modelling studies suggest that floating debris accumulates in ocean gyres but this is dependent on the composition and shape of individual polymers. The widespread distribution and accumulation of microplastics raises concerns regarding the interaction and potential effects of microplastics on marine organisms. As microplastics interact with plankton and sediments, both suspension and deposit feeders may accidentally or selectively ingest microplastics. Despite concerns regarding ingestion, only a limited number of studies examined microplastic ingestion in the field. Knowledge of the retention rates of microplastics would enable estimations of the impacts of microplastic uptake. If rejection occurs before digestion, microplastics might pose less of a threat to organisms than initially assumed. However, there could be energetic costs associated with the production of pseudofaeces. Laboratory studies can be used to determine the end point of microplastic ingestion, and would benefit from using multiple types of microplastics to simulate field conditions. Unfortunately, it is difficult to establish a direct link between microplastics and adverse effects on marine biota experimentally. Furthermore, due to the difficult nature of field studies, it will be harder to understand effects on natural populations.

As microplastic research is still in its infancy, there are many more unanswered questions, the answers to which are required to build on current knowledge to develop a clearer picture of the impact of microplastics in the sea.

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