OCEANS AND ATMOSPHERE
www.csiro.au

Modelling and monitoring marine litter movement, transport and accumulation

Britta Denise Hardesty, Chris Wilcox and Laurent Lebreton

30 October 2016

For United Nations Environment Programme

Oceans and Atmosphere Flagship

Citation

Hardesty BD, Wilcox C, Lebreton L (2016) Modelling and monitoring marine litter movement, transport and accumulation

Important disclaimer

Contents

Figures

Tables

Acknowledgements

This work has been supported by the United Nations Environment Programme and CSIRO's Oceans and Atmosphere Flagship. We acknowledge the invaluable and thoughtful contributions of participants in webinars and workshops, particularly Erik van Sebille, Nikolai Maximenko, Joseph Harari, Dick Vethaak, Luis Valdes, Jim Potemra, Alexander Turra, Atsuhiko Isobe, Alexander Turra, Kara Lavender Law, Peter Kershaw, Sunwook Hong, Won Joon Shim and Heidi Savelli. We thank James Dixon for editorial assistance.

Executive summary

Marine pollution, particularly plastic pollution, is an issue of international concern. Resolving the biodiversity, environmental, economic, transport, navigation and biological invasion hazards associated with anthropogenic litter in the marine environment requires a substantial, sustained contribution from individuals, industry, governments and international organisations. A recent estimate of 6–12 million tonnes of plastic entering the ocean each year points to the need to tackle the problem at multiple scales.

The goal of this work was to compile a state-of-knowledge report on global marine litter monitoring and modelling. We reviewed the published literature on marine litter movement, transport and accumulation at regional and global scales. We aimed to identify knowledge gaps and priority work areas for further research, while acknowledging the persisting uncertainties about the movement, transportation and accumulation of the litter.

We also carried out new modelling that used existing data to model floating marine litter at global and regional scales and applied forecasting and hindcasting to understand and predict key source points and end points for the debris. Those analyses took into account model predictions of litter losses based on human population density in coastal areas, as well as available broadscale information available on waste management within regions. We applied these models to facilitate the monitoring and quantification of marine litter and to identify key sources of marine plastic debris and microplastics at the global and regional levels.

With a better understanding of what is currently known about litter movement, transportation and accumulation, we can identify areas in need of further research and, we hope, identify potential management actions that could result in reduced losses into the marine environment.

We note that marine litter is understudied in geographical regions such as the Caribbean, the South Pacific, the Mediterranean, Eastern Africa, the Arctic and the Antarctic. The lack of information in those and other areas reduces opportunities to manage coastal and marine litter most effectively, particularly where there is little waste infrastructure and management resources are limited.

By garnering the information needed to identify sources and hotspots of debris, increasing our understanding about the uncertainties that persist in our modelling efforts and identifying those critical areas where filling data gaps can result in the best outcomes, we can better develop effective solutions to tackle global marine litter. Working together, scientists, industry partners, coastal managers and citizen scientists can make significant strides to reduce marine litter inputs and impacts in coastal areas and the oceans.

1 The problem

This report reviews the state of knowledge on anthropogenic marine litter and on systems for monitoring the litter and modelling its flows. The aim is to better understanding of the state of knowledge and to identify gaps and potential priority actions at the global and regional levels.

Adequate quantitative and qualitative knowledge of the sources of marine litter is extremely important because it is the main basis for managerial decisions on actions to prevent, reduce and control problems caused by the litter.

Recognising the severity of the marine litter problem, UNEP initiated activities related to marine litter in 2003 through the work of the Regional Seas Programme and the Global Programme of Action for the Protection of the Marine Environment from Land-Based Activities (GPA). Through the GPA, UNEP also established the Global Partnership for Marine Litter (GPML) in 2012. The GPML is a voluntary, open-ended partnership for international agencies, governments, businesses, academia, local authorities, nongovernment organisations and individuals. Through the GPML and in collaboration with relevant partners, UNEP is supporting modelling and monitoring efforts to increase our knowledge and understanding of marine litter movement, transport and accumulation.

1.1 Marine litter and its sources

Marine debris or marine litter is defined as any persistent, manufactured or processed solid material discarded, disposed of or abandoned in the marine and coastal environment (UNEP 2009).

Marine litter is found in all the oceans of the world, not only in densely populated regions but also in remote areas far from obvious sources and human contact, such as the Arctic (Bergmann & Klages 2012) and the Antarctic (Barnes 2002). It poses environmental, economic, human health and aesthetic problems. It is also a complex and multidimensional challenge with significant implications for marine and coastal environments and human activities all over the world. Those impacts are both cultural and multisectoral, rooted primarily in poor practices of solid waste management, a lack of infrastructure, various human activities, an inadequate understanding on the part of the public of the potential consequences of their actions, a lack of adequate legal and enforcement systems and a lack of financial resources. Because the total degradation time for marine plastics is estimated to be in the range of hundreds of years, this is also a multigenerational problem that goes beyond the lifespan of current ocean users, coastal dwellers and others.

Marine litter causes significant economic losses to various sectors and authorities. Among the most affected are coastal communities and the tourism, shipping, fishing, aquaculture and coastal agriculture on which they rely. In addition to suffering economic losses from marine litter, those sectors also contribute to the problem. The major land-based sources of the litter include wastes from dumpsites located on the coast or banks of rivers; rivers and floodwaters; industrial outfalls; discharge from stormwater drains; untreated municipal sewerage; littering of beaches and coastal picnic and recreation areas; tourism and recreational use of the coasts; fishing industry activities;

ship-breaking yards; and natural storm-related events. The major sea-based sources include shipping and fishing activities; offshore mining and extraction; legal and illegal dumping at sea; abandoned, lost or otherwise discarded fishing gear; and natural disasters.

1.2 Definitions

In this work, we define plastics debris or litter as all debris items, regardless of size. When we use the term *microplastics*, we are referring to items that are <5 mm in size, consistent with the definition provided by the United States National Oceanic and Atmospheric Administration (NOAA).

Furthermore, although microplastics have been identified as a priority for further modelling knowledge and research, they are not dealt with in isolation or necessarily separated out from plastics based on the standard definition. However, oceanographic modelling that considers empirical data typically focuses on small or microplastic litter by default (van Sebille *et al.* 2015). While models can consider the broad spectrum of marine plastic debris (from mega, macro and meso to micro and nano, following NOAA definitions), particles may move differently depending on their size, composition and properties. Studies of litter surveyed at sea find that most litter falls within the micro (or smaller) categories (Law *et al.* 2014; Erikssen *et al.* 2014; Reisser *et al.* 2013, 2014; Cozar *et al.* 2014), although larger (macro) debris is also encountered.

Oceanic litter is presumed to be composed of plastics that break down from large to smaller and smaller pieces, and plastics of different sizes and types have different physical and chemical effects on a wide variety of organisms. Furthermore, pathways and fates may differ, depending on the size and properties of the plastics.

1.3 Why use modelling?

UNEP, through the GPA and the GPML, has participated in and supported a variety of awareness-raising initiatives, workshops, conferences and projects related to the global marine litter problem. However, we still lack some of the fundamental knowledge that is needed to tackle the problem effectively. By summarising the current state of knowledge, we can inform and outline key areas in need of further research. By identifying the state of knowledge across the globe, we can better discern gaps in knowledge, such as perceived gaps in regions such as the Caribbean, the South Pacific and East Africa.

At the first United Nations Environment Assembly in June 2014, marine plastic debris and microplastics featured prominently as an environmental issue of international importance. A marine plastic debris and microplastics resolution was adopted by more than 150 countries, demonstrating an increased awareness of the issue as well as an international commitment to continue work to minimise the sources and impacts of marine plastic debris worldwide. The resolution noted the serious impact that marine litter, including plastics, stemming from land and sea-based sources can have on the marine environment, ecosystem services, natural resources, fisheries, tourism and the economy, as well as potential risks to human health. The resolution called for the strengthening of information exchange mechanisms, and tasked UNEP to present scientific assessments of microplastics by undertaking a study on marine plastics and microplastics for consideration by the next session of the assembly in May 2016.

This project provides information to help us understand and predict key source points and end points for anthropogenic debris. The analyses take into account model predictions of litter losses based on human population density in coastal areas, as well as any available broadscale information on waste management in particular regions.

In addition to variability in plastic sources, sinks, pathways and movements on different temporal scales, there is also tremendous spatial variability. On the global scale, surface plastic accumulates in subtropical gyres (van Sebille 2015), demonstrating the heterogeneity in accumulations of microplastics. Small-scale processes such as wave interactions, Langmuir circulation and (sub)mesoscale eddies create a heterogeneous, patchy debris field on the surface of the ocean. Concentrations of floating plastic might therefore vary considerably on length scales of less than 100 metres. Relatively little is known the patchiness of the debris at such fine-scale resolution, even though patchiness is an important concept when interpreting surface trawl microplastics data. It is entirely conceivable that hitting or missing a high-concentration patch with a trawl might affect the results of an observational study (van Sebille *et al.* 2015).

On slightly larger scales (such as hundreds of kilometres), concentrations of floating plastic are also heterogeneous. Local patches of downwelling create accumulation zones of a few tens of kilometres or less in size. Importantly, there are large knowledge gaps about where these mesoscale accumulation regions are located. While the model results from Maximenko *et al.* (2012), Lebreton *et al.* (2012) and van Sebille *et al.* (2015) agree roughly on the location of the large-scale open-ocean accumulation zones in the centres of the gyres, the three models place the meso-scale accumulation zones at very different locations. The meso-scale accumulation zones might hold a significant amount of floating plastic and, because they are often located much closer to shorelines and biologically productive regions, might have a disproportionately large impact on marine life (Wilcox *et al.* 2016).

2 Oceanic compartments

The stock of plastics in the ocean can be divided roughly into five non-overlapping compartments. Plastic can be on or near the ocean surface (including the mixed layer), on the sea floor, on shorelines, in the water column, and in biota. The physical and chemical processes acting on the microplastics in each of these reservoirs are different, and the risks and opportunities for mitigation might also be very different. Separating the total stock of plastic into these five compartments will therefore help us better understand the location of hotspots and the processes that lead to their formation.

With the exception of perhaps the ocean surface, there is a severe paucity in data on the amount of plastic in each of the compartments, and even is less known about the fluxes of plastic between the compartments. Closing off the global plastic budget—that is, accounting for all plastic produced, destroyed and lost into the environment—will require large-scale, targeted sampling of all of the compartments. However, it may be possible to prioritise these investigations according to ease of sampling and likely importance based on our current understanding of their relative contribution to the total volume of plastic in the environment.

2.1 The ocean's surface

Of all the compartments, the surface ocean is probably the best sampled. Extensive plastic trawling data gathered over decades (Law *et al.* 2010, 2014, Cozar *et al.* 2014, Erikssen *et al.* 2013, 2014) have recently been combined in a global dataset of more than 13,000 trawls (van Sebille *et al.* 2015). While the coverage of the dataset is still strongly biased towards some regions, such as the North Pacific and North Atlantic, it reveals clear patterns of (micro)plastic abundance. These studies, while different in their approaches, all come to an estimate of plastic abundance of anywhere between 5 trillion and 50 trillion particles, with a mass of 32,000 to 236,000 tonnes.

Approximately half of the floating plastic detected from surface sampling is in a few relatively confined hotspots—mainly the garbage patches in the centres of the subtropical gyres, where abundances can be a million times higher than in other regions, such as the tropical Pacific and Southern Ocean. Physical oceanographic understanding, including Ekman theory, can explain these patterns, in which the plastic accumulates in areas where wind causes convergence of the surface flow (van Sebille 2015).

2.2 The sea floor

Microplastics have been reported in marine sediments worldwide (Claessens *et al.* 2011; Van Cauwenberghe *et al.* 2013a, 2015; Woodall *et al.* 2015) but the first report on its occurrence in subtidal sediments date back to 2004 (Thompson *et al.* 2004). Deep-sea sediments were demonstrated more recently to also accumulate microplastics (Van Cauwenberghe *et al.* 2013a, 2015; Woodall *et al.* 2015a) with a composition that appears different from surface waters, as fibres were found to be up to four orders of magnitude more abundant in deep-sea sediments from the Atlantic Ocean, Mediterranean Sea and Indian Ocean than in contaminated sea-surface waters (Woodall *et al.* 2015b).

Sediments are suggested to be a long-term sink for microplastics (Cozar *et al.* 2014; Erikssen *et al.* 2014; Woodall *et al.* 2015a). Logically, plastics with a density that exceeds that of seawater (>1.02 g/cm^3) will sink and accumulate in the sediment, while low-density particles tend to float on the sea surface or in the water column.

However, it has been suggested that even low-density plastics can reach the sea floor. Biomass accumulation due to biofouling can lead to an increase in density, resulting in the sinking of the microplastic (Andrady 2011; Zettler *et al.* 2013). Indeed, analysis of polyethylene bags submerged in seawater showed a significant increase in biofilm formation over time, accompanied by corresponding changes in physicochemical properties of the plastic, such as a decrease in buoyancy (Moret-Ferguson *et al.* 2010; Lobelle & Cunliffe 2011). These studies suggest that biofouling can contribute to the settling and eventual burial in sediments of previously buoyant plastic, and biomass accumulation on plastic may even partly explain why the open-ocean surface estimates are two orders of magnitude lower than expected from estimates of plastic releases in the marine environment (Cozar *et al.* 2014; Erikssen *et al.* 2014). However, the situation is probably more complex, as one may argue that, after sinking, biofilms and fouled organisms may not survive and thus disappear; in addition, grazing of the biofilm may enable vertical movement back to the surface layers (Song & Andrady 1991). Alternatively, aggregation with organic matter (marine snow) has also been considered as a main route of transport for microplastics to deep-sea sediments (Van Cauwenberghe *et al.* 2013a).

A number of oceanographic processes could aid in the transfer of microplastics to depth. As stated in Woodall *et al.* (2015), those processes include the cascading of dense shelf water, severe coastal storms, offshore convection and saline subduction. All these induce vertical and horizontal transfers of large volumes of particle-loaded waters, including grains of various sizes and nature as well as litter and contaminants, from shallow ocean layers and coastal regions to deeper ones, with submarine canyons acting as preferential conduits, as for larger debris (Galgani *et al.* 1996; Pham *et al.* 2014).

Mechanisms influencing microplastic distribution on the sea floor are not so well understood. Microplastic fragments are also more likely than larger items to be influenced by advection and, more generally, circulation patterns at all ocean levels because of their small size (Woodall *et al.* 2015). Ocean dynamics could then explain the accumulation of plastics in the deep sea or shallower waters.

In the lagoon of Venice, Vianello *et al.* (2013) detected the lowest microplastic concentrations where water currents were higher (outer lagoon, >1 m/s) when the inner lagoon, which is characterised by lower hydrodynamics, had a higher fine-particle (<63 mm) fraction in the sediment (Table 1). On the deep-sea floor, circulation is not well explained and pathways are different from surface circulation. Thus, the prediction of distribution patterns will require a better understanding of circulation patterns to locate the most probable areas of accumulation, if any. Submarine topographical features may also favour sedimentation and increase the retention of microplastics at particular locations, such as canyons and deeps or smaller scale structures (holes, rocks, geological barriers etc.). As for larger debris, human activities may also affect composition and repartition, as shown by the high densities of microplastics found in harbour sediments (Claessens *et al.* 2011), reaching up to 391 plastic particles per kilogram of dry sediment. Similarly, In Slovenia (Bajt *et al.* 2015), concentrations of between 3 and 87 particles per 100 g were found, and coastal areas were generally more affected.

The Arctic region has low human population density and low local litter inputs to the ocean. Nonetheless, significant densities of debris have been found on the deep-sea floor (Galgani & Lecornu 2004), and microplastics have been detected in polar ice (Obbard *et al.* 2014).

Finally, our understanding of the dynamics of transport, accumulation and associated spatial distribution has been extremely limited. Robust temporal and spatial distribution must be considered in order to estimate globally the quantities that are present and, as for the surface, to predict sea-floor plastic accumulation.

Table 1. Location, location specification and abundance of microplastics in subtidal sediments

Continent	Location	Location specification	Depth	Particle size	Measured abundance	Reference
America	US	Maine subtidal		0.250 mm–4 mm	105 items/L	Graham & Thompson (2009)
	US	Florida subtidal		0.250 mm – 4 mm	116–215 items/L	Graham & Thompson (2009)
	Brazil	Tidal plain		1 mm – 10 cm	6.36–15.89 items/m^2	Costa *et al.* (2011)
Asia	India	Ship-breaking yard		1.6 mm – 5 mm	81.4 mg/kg	Reddy *et al.* (2006)
	Singapore	Mangroves		1.6 mm – 5 mm	36.8 items/kg dry	Nor & Obbard (2014)
Europe	UK	Estuary			2.4–5,6 fibres/50 mL	Thompson *et al.* (2004)
	Sweden	Subtidal		2 mm – 5 mm	2–332 items/100 mL	Noren (2007)
	Belgium	Harbour		0.38 mm – 1 mm	166.7 items/kg dry	Claessens *et al.* (2011)
		Continental shelf		0–200 m	97.2 items/kg dry	
	Italy	Subtidal		0.7 –1.0 mm	672–2,175 items/kg dry	Vianello *et al.* (2013)
	Slovenia	Shelf	Infra-littoral (<50 m)		30–800 items/kg dry	Bajt *et al.* (2015)
Oceanic sediments	Polar ocean, Mediterranean, North Atlantic, Gulf of Guinea	Deep sea	1,176–4,848 m	1 mm – 5 mm	0.5 items/cm^2	Van Cauwenberghe *et al.* (2013a)
	North-west Pacific	Deep sea trench	4869–5,766 m	0.3–5 mm	60–2,020 items/m^2	Fisher *et al.* (2015)
	Subpolar /North Atlantic	Deep sea mount Slope	1,000–2,000 m	0.032–5 mm	10–15 pieces/50 mL	Woodall *et al.* (2015a)

Continent	Location	Location specification	Depth	Particle size	Measured abundance	Reference
	North-east Atlantic	Canyons / slope	1,400–2,200	0.032–5 mm	6–40 pieces/50 mL	Woodall *et al.* (2015a)
	Mediterranean	Canyons / slope / basin	300–3,500	0.032–5 mm	10–35 pieces/50 mL	Woodall *et al.* (2015a)
	South-west Indian	Seamount	500–1,000	0.032–5 mm	Up to 4 pieces/50 mL	Woodall *et al.* (2015a)

Source: Modified after van Cauwenberghe *et al.* (2015).

2.3 The shoreline and coastal margin

Current estimates of the distribution of debris along coastlines (for example, Hardesty *et al.* 2015), particularly for microplastics, may substantially underestimate the total standing stock. A recent estimate for microplastics suggests that the surface may contain as little as 10% of the total stock in coastal sediments (Turra *et al.* 2014). Even in remote areas, shorelines may contain substantial amounts of debris. Samples from isolated beaches in the outer Hawaiian Islands contained over 23 grams of plastic per 20 litres of sediment, on average (McDermid & McMullen 2004). This is potentially driven by deposition from high-concentration areas in the nearby marine environment, and is similar to patterns found on Easter Island, which adjoins the high-concentration area in the southern Pacific (Hidalgo-Ruz & Thiel 2013).

The most recent estimate suggests that approximately 8.4 million tons of plastic enters the ocean each year, around 80% of it from land-based sources. Models using this flux to estimate the standing stock in the ocean predict approximately two orders of magnitude more plastic than is currently found in the ocean (Cozar *et al.* 2014, Erikssen *et al.* 2014, van Sebille *et al.* 2015). Given this mismatch, one clear possibility is that much of this material is deposited along the coastline near its sources. It has long been reported anecdotally that coastal debris increases near urban centres, suggesting local deposition of debris transported by the marine system from nearby urban sources (Hardesty *et al.* 2014; Browne *et al.* 2011; Claessens 2011).

Recent research using a robust sampling approach and a statistical model to correct for bias and the confounding effect of direct inputs from land to coastal environments supports the assertion that coastal sediments are a reservoir for debris from the marine system (Hardesty *et al.* in press). This suggests that local marine transport of debris from land-based sources onto the coastline may account for a substantial portion of the missing debris, at an estimated rate of 5.2 items per metre of coastline in Australia (Hardesty *et al.* in press). This inference is supported by preliminary modelling results, which suggest that, when onshore transport mechanisms are considered, in the order of 90% of marine debris generated in a coastal region may be deposited on the coastline in the region (C. Wilcox *et al.*, unpublished data).

Coastal debris surveys often report an increase in beach deposition of litter following storms or large rain events in which litter is washed landward (Frost & Cullen 1997; Gabrielides *et al.* 1991; Vauk & Shrey 1987), further supporting the importance of local contributions to marine litter.

Many authors have noted the presence of higher levels of litter in coastal regions near river outflows, particularly those that pass through urbanised areas. There is long-term evidence of increased transport of land-based pollutants due to extreme events such as hurricanes or typhoons (Kuo *et al.* 2014; Osborn *et al.* 2008; Nixon & Barnea 2010). Research in Korea estimated that deposition rates of marine debris in coastal regions were 14 times higher than normal after the passage of a typhoon (Park & Yoon 2007). Work by NOAA researchers after a hurricane in the Gulf of Mexico found that most of the larger debris transported offshore was deposited within a few kilometres of the coastline in areas close to heavy infrastructure damage on land (Nixon & Barnea 2010). Studies of solid waste losses into the ocean have also noted the effect of smaller events, such as the substantial effect of the first rainfall of the year (Armitage & Rooseboom 2000; Allen *et al.* 2015; Guneroglu 2010; Marais *et al.* 2004).

While the connection between land-based littering and debris in marine systems has long been appreciated (for example, Armitage & Rooseboom 2000), there is less appreciation of links to illegal dumping, particularly in developed countries where waste management is well regulated. Coastal debris has been linked to illegal dumping in large-scale empirical analyses (Hardesty *et al.* in press). Similarly, waste management in coastal regions, and unregulated dumping in particular, have been assumed to be a major driver of solid waste input into marine systems (Jambeck *et al.* 2015). In developed countries with strong waste management regulation, illegal dumping appears to be a major problem in waste management (Baird *et. al* 2014). For example, there were an estimated 5,000 illegal dump sites of commercial scale in southern Italy in 2010. Illegal disposal on public lands, which frequently include waterways, wetlands and other locations in the coastal zone is a common problem globally across the range of development levels (Vieira *et al.* 2013; Glanville & Chang 2015; Njue *et al.* 2012; Hettiarachchi *et al.* 2011).

Approximately 20% of the plastic load in the ocean is assumed to come from marine sources, including shipping, fishing, and oil and gas platforms (Watkins *et al*. 2015). A growing number of studies link debris, including plastic, on the sea floor to vessel-based activities and areas of high vessel density. This suggests that marine sources of plastic can be quite significant in some areas, in particular on fishing grounds and in navigation channels (NOAA Technical Report 2011). This variation is reflected in debris washing up in coastal areas: regions of high fishing activity, low coastal populations, or both frequently show a preponderance of fishing- and vessel-related waste. In some areas, such as the North Sea, this source has been estimated to contribute as much as 90% of the waste stranded along the coastline (van Franeker 2010).

It is very difficult to get data on the waste disposed of by vessels. A recent study of port reception facilities in Europe provided an estimate of 150,000 to 220,000 tons of waste disposed in port facilities across the European Union per year between 2004 and 2010 (Øhlenschlæger *et al.* 2013). The study authors also noted significant variation in the waste from year to year, but could not find definitive explanations for it. They listed a number of potential drivers for the variation, including the development of private disposal arrangements, issues related to reporting structures, and the potential for illegal disposal, driven by cost in time and money, the level of supervision and the provision of facilities (Øhlenschlæger *et al.* 2013).

A recent study of solid waste disposal behaviour on large fishing vessels operating in the Western Pacific was able to estimate dumping rates of shipboard waste, oil discharges and leaks, and losses of fishing gear in 8,000 incidents between 2004 and 2013 (Richardson *et al.* 2015). Discharges of solid waste accounted for 69% of the incidents observed, while losses of fishing gear accounted for only 13%. Of the solid waste discharges, the largest fraction (36%) were plastic. Nearly all of these discharge events are prohibited under the MARPOL Convention, the international treaty regulating waste disposal at sea (Richardson *et al.* 2015). While Richardson *et al.* were able to estimate frequencies and spatial distributions of disposals, they did not provide estimates of volume.

Loss of fishing gear at sea is substantial source of plastic materials in marine systems, at least among at-sea sources. As in other cases, estimates of the rates of loss and the volumes are difficult to come by, largely due to limited monitoring efforts and little synthesis at the global scale to date (Gilman 2015). Globally, it is estimated that around 6.4 million tonnes of fishing gear is lost into the sea annually (MacFadyen *et al.* 2009). Loss rates vary by fishery, and the causes include gear design, conflicts between fisheries, overcrowding, weather and other drivers. Studies suggest gear loss rates in the order of 10–20% per year for trap fisheries, spatially correlated with fishing effort and catch (Maselko *et al.* 2013). Loss rates exceeding 70% have been reported in some contexts, particularly in cases where gear is in contact with the bottom (Ayaz 2010). Evidence from coastal net-removal operations suggests that loss rates are particularly high in some regions. For example, Wilcox *et al.* (2016) report densities of derelict nets as high as 3 tonnes per kilometre of coast in northern Australia, in that case due to a complex of illegal fishing, overcrowding and low awareness or capacity among operators and crew.

Particularly in developing regions, we find that the capacity of the waste management system influences whether waste is in the control of the local authority's management system, dumped into the environment or picked over by the informal sector. The capacity of available collection vehicles, the availability of staff for collection, the condition of waste containers and their ability to hold waste, and the capability of the final disposal site influence the path of waste from the household (Dangi *et al.* 2013). Losses from coastal regions can readily make their way to the shoreline through surface transport, waste mismanagement, stormwater transport, roadways and local users.

2.4 The water column

The vertical distribution of marine debris has been documented in the surface and subsurface portions of the water column (Lattin *et al.* 2004; Lusher *et al.* 2015; Reisser *et al.* 2015), although how much plastic is just below the ocean surface is less well understood. Recent modelling (Kukulka *et al.* 2012) and observations with vertically stacked trawl nets (Reisser *et al.* 2015) have shown that, depending on the sea state, a significant fraction of plastic may be mixed down due to wave breaking and mixing in the upper few metres of the ocean surface. Since most 'standard' trawls skim only the top 10 cm of the ocean surface, they may miss a considerable part of the plastic, especially in rough seas.

The most extensive sampling of the vertical distribution of plastics in the water column to date examined the abundance and mass of plastic debris with depth in the North Atlantic gyre, an area of high plastic concentration. That study found an exponential decay in abundance and mass with increasing depth to a maximum of 5 metres below the surface. The study also documented relationships with wind and sea conditions, in which higher wind resulted in increased downward mixing. The distribution of smaller plastic items was more strongly affected by these conditions, driven by slower rising rates of these less buoyant particles (Reisser *et al.* 2015).

This pattern of wind-driven mixing complements an analysis of the global microplastics dataset. Analysing the effects of sampling conditions on the densities observed in 13,000 at-sea samples, van Sebille et al. (2015) found a significant negative effect of wind on the densities observed. The effect of wind was a linear first order one, suggesting that increasing wind velocities mix plastic debris down below the surface layer available to the standard plankton tows used for debris sampling.

Both empirical measurements of the depth profile of debris (Reisser *et al.* 2015) and statistical analysis of debris fields (van Sebille *et al.* 2015) have found that existing physical models of downward mixing underestimate the observed mixing (Kukulka *et al.* 2012). More recent developments of these physical models of mixing may improve the characterisation (Kukulka & Brunner 2015). Overall, empirical studies, statistical models and physical models suggest that debris in the water column is confined mainly to regions near the surface and the ocean floor.

2.5 Biota

Plastics accumulate in oceanic and coastal areas and can be pollutant vectors, promoting effects on marine biota. However, the risk of such impacts depends on the type, size and amount of plastic present in the environment, the presence of contaminants in plastic and contact with sensitive biota.

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

Marine organisms from microbes to invertebrates have always attached to natural floating substrates (macroalgae, feathers, wood, pumice and so on), but one important difference between those natural materials and plastic is the greater longevity of plastic. This allows more mature communities to form and persist, facilitating the transport of viable populations further than would have been possible in the pre-plastic era. The distribution of plastic is different from that of natural substrates, and plastic has substantially increased the available substrate in oligotrophic open ocean regions, potentially altering the distributions of marine organisms (Goldstein *et al.* 2012).

Some marine animals are indiscriminate feeders that will ingest anything in the appropriate size range. Others, such as seabirds and turtles, use visual, chemical or electrical cues to find and select food, so the probability plastic being ingested depends not only on size and encounter rates, but also on a number of other cues, including shape, colour, smell and taste. Encounter rates, however, are good predictors of ingestion for some marine taxa (Wilcox *et al.* 2015).

Fish eggs and other biological material can effectively cover and functionally 'hide' plastic from consumers, which may also increase the likelihood of ingestion. The smell and taste of plastic can also be influenced by the microbial biofilm on the surface, and microbes colonise plastic in seawater very quickly; within a week, most of the surface may be covered. The thin layer of living organic matter and by-products makes the plastic smell and presumably taste like nutritious particles. This increases the likelihood of ingestion by animals that use chemoreception to select food particles. Both the likelihood of ingestion and the potential impact on the organism ingesting it will vary depending on the composition of microbial community, including whether it includes potential pathogens.

The microbial community associated with plastic in the ocean also varies regionally and seasonally (Oberbeckman *et al.* 2014), as well as on larger scales, such as between the Atlantic and Pacific oceans (Amaral-Zettler *et al.* 2013; 2015), suggesting that risk management approaches to addressing the issue will require the appropriate resolution of risk factors in both space and time.

3 Fluxes, distribution and hotspots

Closing off the complete plastic budget and understanding litter hotspots requires an understanding not only of the inventory (stock) of plastic in each of the compartments, but also of the movement (flux) between them. Because it is much harder to measure fluxes than stock, this is an area where even less is known. Most analysis to date has been on the physical transport of plastics in the ocean surface and between the land and ocean surface.

Another open question is how plastic leaves the ocean surface. From the ocean's surface compartment, plastic can move to any of the other four compartments: to the water column and sea floor by sinking (most likely through density increases caused by biofouling; Andrady, 2011; Zettler *et al.* 2013), to the shoreline by beaching (which may be event-driven as storms wash up large amounts of plastic), and into biota through ingestion. Finally, plastic might degrade into (almost unobservable) nanoplastic through fragmentation and breakdown.

The distribution of debris, and particularly microdebris, in the ocean is increasingly well known. However, three major areas of uncertainty (detailed in Section 8.2) suggest avenues for further research.

3.1 Physical processes

Plastic on the surface of the ocean can be considered passive when it is carried by currents. However, the depth at which the plastic resides has large impacts on its pathway, as the currents in the upper ocean vary quite significantly over the top 50 metres or so (the Ekman spiral). The buoyancy of the plastic particles and the amount of wind mixing and wave breaking make it very difficult to predict where plastic particles will be. In general, there appears to be an exponential decay of the concentration of plastic with depth (Kukulka *et al.* 2012, 2015; Reisser *et al.* 2015; Brunner *et al.* 2015).

Beyond vertical mixing, waves and wind also affect the horizontal transport of plastic. Stokes drift within waves can be a significant factor in the pathway of plastic, especially in coastal regions. And while direct windage is likely to have more of an impact on macroplastic that has some surface area protruding above the water surface, there is the possibility that microplastic can be moved ballistically in high winds.

Recent analysis of the match between the global dataset of plastic, based on more than 13,000 samples, and oceanographic models treating plastic as passively drifting in the surface layer, produce reasonable predictions of the spatial distribution of plastic at the global scale (van Sebille *et al.* 2015). While these models do not include the influence of Stokes drift or ballistic wind transport, it appears that they provide an adequate first approximation for the physical process of movement in offshore regions at the global scale.

Studies of the size distribution of particles in the water column suggest that larger particles remain closer to the surface because the balance of downward mixing and buoyance shifts with particle size (Reisser *et al.* 2015). The implication of this is that surface-layer transport by Stokes drift, and potentially ballistic transport by wind during extreme winds, will differentially transport larger items to coastlines (Reisser *et al.* 2015).

3.1 Biological processes

Consumption of plastic by large animals, including fish, seabirds, marine turtles and other species, is increasingly a focus of research. Much of this work is centred on estimating the impact on the species, as opposed to estimating the effect on the transport of debris between compartments. However, some taxa, such as seabirds, have been observed to transport large volumes of plastics between compartments, in this case from the ocean surface to the terrestrial environment.

Recent research suggests that fouling may be more rapid for plastics that are thinner, such as films, while those that are relatively compact for their volume are slower to sink due to fouling (P. Ryan, pers. comm.). This pattern of fouling has been found to be similar to the distribution of shapes relative to coastal regions in South Africa, with a higher abundance of film- and sheet-like plastics near shore and a higher relative abundance of more compactly shaped items offshore—potentially indicating the effect of differential fouling and settling rates (P. Ryan, pers. comm.).

3.2 Mechanical processes

The kinetics of fragmentation at sea and the particle size spectrum that results remain unknown for even the most common plastics, although many processes in the marine environment that cause disintegration have been identified. The mechanical energy required for disintegration may come from physical, biological or anthropogenic processes. Wind, sand and wave action at the sea surface, on the sea floor or on beaches abrades or alters weakened plastics. Some animals also reduce objects' size by chewing them, and marks on debris, especially polystyrene, from large fish, including sharks, or birds are also common (Cadée 2011; Carson 2013). The grinding of ingested plastics may also reduce the size of plastic marine debris. Tube-nosed seabirds alone may alter hundreds of tonnes each year (van Franeker 2011; van Franeker *et al.* 2011), and even minor disintegration in fish stomachs could make a non-negligible contribution to particle fragmentation.

Finally, disintegration of plastics in interactions with at-sea vessels may be caused by the mechanical stresses encountered in collisions, in grinding in propellers or from passage through circulation systems. Although expected to be minor compared to other disintegration mechanisms, such anthropogenic processes may be non-negligible, especially for the polystyrene foam that may make up as much as 90% of litter floating in coastal zones (Hinojosa & Thiel 2009) and 18% of microplastics in the Mediterranean Sea (Collignon *et al.* 2012).

3.3 Distribution and hotspots

A number of recent analyses have used a mixture of at-sea sampling and oceanographic models to infer the distribution of plastics at sea (Lebreton *et al.* 2012; Maximenko *et al.* 2012; Erikssen *et al.* 2014; Cozar *et al.* 2014; van Sebille *et al.* 2015). These analyses are all in relative concordance, suggesting that there are concentrations of microdebris in the five major oceanic convergence zones spread across the Pacific, Atlantic and Indian oceans.

The recent review and reanalysis of the all of the available global data by van Sebille *et al.* (2015) compared three previous oceanographic models to identify areas of concordance and evaluate uncertainties caused by differences in the models. Areas of major difference include the coastal regions and some enclosed basins, such as the Mediterranean. The differences at the coastal margin arise largely from differences in the source functions used in the models, some of which include modelled plastic inputs from the coast based on some mixture of changes in plastic production, coastal population and mechanisms driving inputs to the ocean. Differences in enclosed basins, such as the Mediterranean, arise from model structure and fitting approaches.

Emerging evidence suggests that benthic and coastal regions may also contain significant reservoirs of debris (Galgani *et al.* 1995; Barnes *et al.* 2009; Claessens *et al.* 2011; Keller *et al.* 2010; Hardesty *et al.* 2015). For instance, surveys in coastal California in the nearshore environment in waters off Los Angeles (<30 m depth, within 5 km from shore) noted significant amounts of small-sized plastic on the sea floor, exceeding amounts on the surface (Lattin *et al.* 2004). Reports suggest that something in the order of 50% of plastic is negatively buoyant, so there may be a substantial reservoir of debris in areas near land-based sources (Lattin *et al.* 2004). Reports from more isolated regions generally record lower densities of debris on the sea floor, and the debris appears to be primarily shipping and fisheries waste (Barnes *et al.* 2009; Angillilo *et al.* 2015). However, one of the larger surveys found the opposite pattern (debris increased with depth) and no pattern relative to urban areas (Keller *et al.* 2010). Differences in methodology (such as much larger mesh sizes in nets used for offshore samples) may explain this discrepancy. Recent continental-scale surveys in Australia identified a significant increase in coastal debris in proximity to populated areas—a pattern widely noted elsewhere (Hardesty *et al.* 2015).

Taken together, these results suggest that hotspots for microdebris in the coastal and benthic reservoirs might be in proximity to urban sources.

3.4 Diagnosing areas of uncertainty

Our understanding of the sources, distribution and sinks of plastic debris, and microdebris in particular, is largely based on a growing dataset of at-sea trawls using fairly standardised methods developed originally for sampling plankton (Law *et al.* 2010; Cozar *et al.* 2014; Lebreton *et al.* 2012; Erikssen *et al.* 2014). While the analysis of the global dataset by van Sebille *et al.* (2015) found general agreement among the models, it also raised a number of questions related to uncertainties in our understanding of the distribution and dynamics of debris. There are at least three general sources of uncertainty in our current understanding of the distribution of marine debris that are important to consider.

First, the existing data are not evenly distributed throughout the world (Figure 1). Debris samples are concentrated in the north-eastern Pacific and north-western Atlantic oceans, and sampling elsewhere is substantially more sparse. Areas predicted to have very high land-based inputs, particularly in Africa and Asia, are only very sparsely covered, if at all (van Sebille *et al.* 2015; Jambeck *et al.* 2015). Moreover, sampling is concentrated in the open ocean, with relatively few samples in the coastal region globally. Exceptions include the coastal regions of North America, Australia and a limited portion of Japan (van Sebille *et al.* 2015).

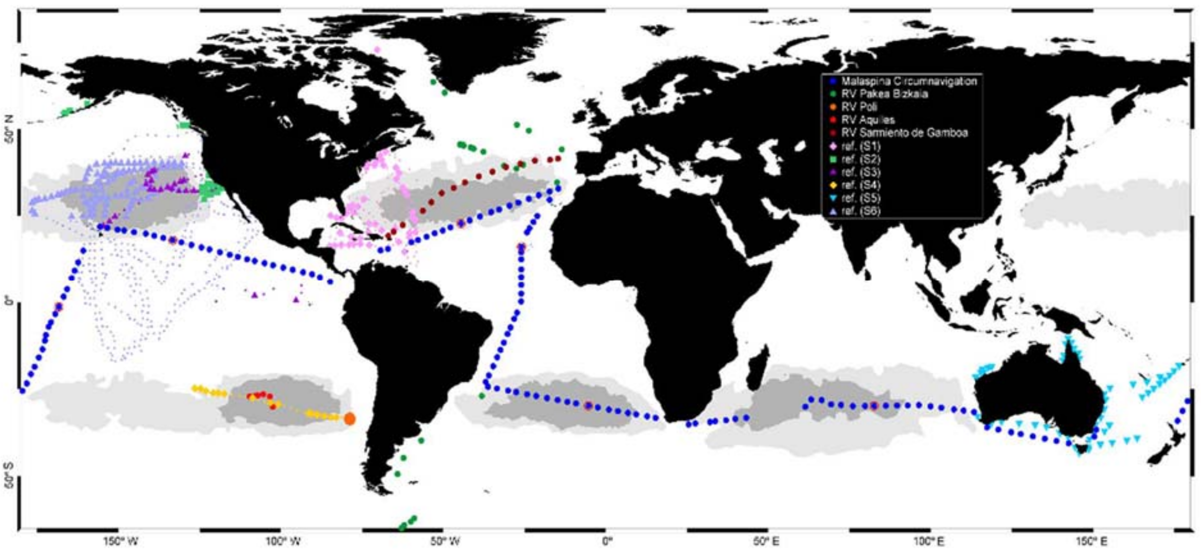


Figure 1. Spatial distribution of data, by source

Note: Figure legend indicates the vessel for the original data and the bibliographic reference for the datasets published. Large solid symbols show wind-corrected data (442 grid cells, 1,127 surface net tows). Small dots show literature data classified as affected by high wind conditions and not included in the wind-corrected dataset (409 grid cells, 1,943 surface net tows). Sampling sites where the size distribution of items that were not plastic was measured are marked with red circles.

Source: Supplementary Information in Cozar *et al.* (2014).

A second source of uncertainty is the variation among samples at a location. Sampling conditions such as wind, time of day and sea state can influence both the availability of debris in the surface component of the ocean to be sampled (Reisser *et al.* 2013) and the accuracy of that sampling. The recent work by van Sebille *et al.* (2015) used a spatial statistical model to account for effects of wind, changes over time in debris and a variety of other sources of variation in creating the corrected global dataset of debris samples. Even in areas with significant sampling, there is still substantial variation among samples. The authors found that their statistical model could account for 71% of the variation in the at-sea data, but nearly 30% of the variation remained unexplained. This variation is likely to be unavoidable, as ocean conditions affect sampling and there is significant variation in densities of debris even at very small scales (Reisser *et al.* 2014).

A third source of uncertainty is driven by the fit of the oceanographic models to the corrected data. The models are estimated from information on ocean currents derived from a mixture of satellite measurements and drifting buoys. Thus, the models themselves depend on measurements and have some level of uncertainty associated with them. The impact of this estimation uncertainty has not been addressed in the scientific literature to date. In addition to the estimation uncertainty, the oceanographic models of debris distribution must also make a number of assumptions about sources, sinks, wind forcing and a variety of other factors that influence their accuracy (van Sebille *et al.* 2015).

4 Translating debris distributions into estimates of ecological risk

This section reviews the use of risk assessments to expand knowledge about marine plastic pollution and investigates the use of the results to prioritise further research and management actions.

4.1 The concept behind risk assessment

A risk framework, as used in risk assessment, separates the risk under consideration into two components: the probability that the event of interest will occur, and the magnitude of the event if it occurs. For example, recent publications in this vein have considered plastic ingestion by seabirds (Wilcox *et al.* 2016). In that case, the event of interest is the presence of plastic in the digestive tract of a seabird. Models of spatial overlap between birds and plastics in the ocean were used to estimate the probability of birds encountering plastic—the first component of the risk framework. The magnitude of the event (ingestion) was then estimated based on a statistical model using data from the literature on plastics found in seabirds. In mathematical terms, this is equivalent to estimating the expected outcome of the process—defined as the product of its probability and its magnitude. This concept of separating the likelihood of occurrence and the magnitude of the event if it occurs is useful in reducing the analysis to components that are easier to understand, estimate or model on their own.

Risk analysis can take a variety of forms, allowing different levels of quantitative analysis and differing emphasis on the two components. For example, a literature review might compile information on whether a particular type of outcome has been observed. In that context, the focus is on the magnitude of the event, with less attention to its probability of occurrence. For instance, a recent review of impacts of marine debris on ecological systems (Rochman *et al.* in press) can be seen in this light, as the fundamental focus is on the outcomes, leaving their likelihood aside. Other analyses might focus more on the frequency of an event as a basis for estimating its probability, with less emphasis on the magnitude of the event.

In addition to the weighting between the two components of the analysis—probability and magnitude—a range of analytical approaches can be taken depending on the availability of information, the complexity of the problem being represented and the specificity of the answer that is sought. For example, a recent analysis focusing on the lethal and sublethal impacts of the most common debris items used a structured questionnaire, administered to a large number of experts, to estimate both the probability and the magnitude of the impact of those items across three taxa—seabirds, marine turtles and marine mammals (Wilcox *et al.* 2016). While this analysis is subject to substantial uncertainty, as it is based on expert judgement, it has the advantage of being able to integrate across a very complex system to produce predictions of impact that are useful for policymaking and advocacy. At the opposing extreme, in cases where there is adequate information, a mixture of mechanistic and statistical models can be used to represent the phenomenon of interest, leading to specific predictions of its magnitude, its probability and even the spatial and temporal distribution of its impacts (Wilcox *et al.* 2013, 2014). In some cases, it may even be possible to link the impacts to standards, such as reductions in population size, that have clear links to other policy instruments, such as species protection legislation or international agreements.

We suggest that the explicit consideration of a risk framework is an appropriate lens through which to examine marine debris issues, regardless of the spatial, temporal or hierarchical level of biological organisation or the taxonomic level of focus.

4.2 The current state of risk assessments for marine debris impacts

A number of published review papers have documented interactions between plastic debris and marine wildlife, including a recent study documenting 693 species as affected (Gall & Thompson 2015). Rochman and colleagues (2015) recently reviewed the evidence for impacts at a variety of levels of ecological organisation, from cellular to organismal and reaching up to ecosystem level. Using a careful evaluation of experimental design and evidence, they were able to find established evidence of an impact to individuals due to debris in 24 cases out of 362 that they evaluated. However, 20 of the 24 were due to macrodebris and only four were due to microdebris (<1 mm, as defined in the study).

Note that the frequency of demonstrable impacts increases substantially at levels of organisation below the individual, particularly for microdebris (Rochman *et al.* in press). Thus, at cellular or tissue levels, the incidences of demonstrated impacts were 45 and 29 cases, respectively. Many of the suborganismal scale studies were focused on sources of debris aside from marine debris, such as from medical implants. Rochman *et al.* (2015) also found a relationship between the size of the debris and the level of biological organisation at which the impact was demonstrated. Thus, tissue-level impacts were primarily from relatively small debris, ranging from millimetres to centimetres, while demonstrations of debris affecting whole organisms tended to be in much larger size classes, in the order of metres.

Compared to the number of papers dealing with the magnitude of impacts, relatively few studies to date have dealt with the probability of occurrence at the individual, population, species or ecosystem level. Recent work on estimating rates of ingestion of plastic marine debris by turtles (Schuyler *et al.* 2014a, 2015) and seabirds (Wilcox *et al.* 2015) has demonstrated the potential to use global models of debris distribution to estimate ingestion rates for wildlife. More than 50% of turtles (a lower bound estimate of >340,000 animals) and 90% of the world's seabirds (increasing to 99% by 2050) are estimated to ingest debris, which indicates the magnitude of the problem. However, to date, similar large-scale estimates have not been made for ingestion rates in marine mammals, fish or the wide range of marine invertebrates that could be affected by debris.

The remaining task in estimating ingestion risk is to connect the emerging estimates of the frequency or probability of ingestion with experimental or observational results on the relationship between ingestion and demographic impacts. While the review by Rochman *et al.* (in press) suggests that there are established impacts of debris exposure, particularly at the suborganismal level, what is lacking is a known relationship between the dose of marine debris ingested by an organism and the response in terms of survival, reproduction, or both.

In contrast, there are estimates of entanglement risk that predict impacts of marine debris in a mechanistic and statistically robust manner at the population or assemblage level. Gilardi *et al.* (2010) were able to estimate rates of mortality caused by derelict fishing gear for a range of species in Puget Sound, Washington. Similar estimates have been made for commercially harvested species, which can be affected by lost gear, particularly for static gear such as gillnets (for example, see the summary in Pawson 2003). Wilcox *et al.* (2013) extended the work of Gilardi *et al.* (2010), demonstrating the use of oceanographic models for predicting rates of marine turtles' entanglement in drifting debris. They were able to use observed rates of entanglement in derelict gear, together with predictions of drift trajectories, to estimate that approximately 20,000 turtles had been caught by derelict fishing gear and other marine debris arriving on the northern coast of Australia. While they did not translate capture directly into mortality, adding that component would be relatively straightforward, given the results they present.

An alternative to the mechanistic approach to population-level risk assessments described above is to use expert elicitation to estimate the relationship between exposure to debris and impacts on individuals. This involves using a structured questionnaire to obtain estimates from experts on the two risk components (probability and impact). The estimates can then be combined and used to make predictions of the risk of population or higher level impacts. Wilcox *et al.* (2016) recently conducted such an analysis, focusing on the risk from the 20 most common plastic items found in coastal clean-ups around the globe. The analysis estimated the risk to three taxa (turtles, seabirds and marine mammals) from each of the 20 items. Effects of the items were considered across three categories of action (entanglement, ingestion and toxicity). The authors found strong evidence for lethal and sublethal effects of entanglement and ingestion, but much weaker evidence for effects due to toxicity.

There is emerging evidence that plastics in the ocean are having impacts at the individual, population, species and assemblage levels. There are clearly documented anecdotes of both lethal and sublethal effects on individuals. Reviews of the empirical literature document an increasing number of studies investigating debris prevalence in wildlife, continually expanding the number of species known to interact with debris. Similarly, experimental and observational research is increasingly documenting impacts on biological levels of organisation, from DNA up to whole organisms. As modelling efforts have moved forward, studies have gone from estimating impacts to organisms in specific cases to taxon-wide estimates of debris interactions.

The remaining challenge is to connect empirical studies on the effect of debris ingestion to analyses predicting its frequency across populations or species, which will allow estimates of the effect of this environmental threat at the population and species levels.

4.3 Linking likelihood, consequences and responses

In bringing risk analysis on marine debris impacts to bear on decision-making, there is a need to link risk estimates to the values placed on the potential outcome. As the human value of the outcome increases or the certainty of the outcome increases, societies are commonly more willing to take stronger action to compel people to act in consideration of the risk.

It will be difficult to develop quantitative predictions of both the value of an outcome involving marine debris and its probability at all levels of biological organisation. Thus, it will be important to identify levels at which information will make a critical difference.

A number of clear breakpoints, in terms of both social value and regulatory triggers, might be considered in choosing priorities for risk assessment. Human health impacts are a clear breakpoint. Emerging evidence suggests that there are plastics in fish consumed by humans, and that this could have the potential to transfer either plastic itself or leached toxins into the body tissues of consumers (Rochman *et al.* 2015). Based on a review of global studies of fish, plastic is present in roughly one-third of individual fish, including commercially important species (Wilcox *et al.*, unpublished data). Thus, given the evidence for a plausible human health link to marine debris, understanding human exposure through food fish could be a reasonable priority. Because of the link to human health, it is plausible that action could be taken even in the face of substantial uncertainty about the likelihood of occurrence.

Similarly, environmental regulations in many countries target species and/or population protection for species listed as biodiversity conservation priorities. In this case, the level of certainty in analysis is likely to be higher in order to affect regulatory outcomes, as on some level the human value of the outcome is lower. Thus, in contemplating the priority of investigations of population- or species-level impacts, it will be important to establish that adequate information is available to be able to establish a significant, unambiguous impact to accord with the relevant legislation or international agreement.

Risks of impacts at the various levels of biological organisation, from DNA to ecosystems, can in some ways be mapped along with the value of those impacts, as in Figure 1. DNA impacts on humans would clearly fall at the very high end of the value spectrum, while organismal or even population-level impacts on non-commercial marine invertebrates are likely to fall much lower. Thus, in contemplating priorities for risk assessment and, by implication, investments in improving knowledge about the sources, distribution and fate of marine debris, it will be important to keep in mind the values and certainty needed to trigger action by governments, nongovernment organisations and international bodies in order to maximise the value of the investments.

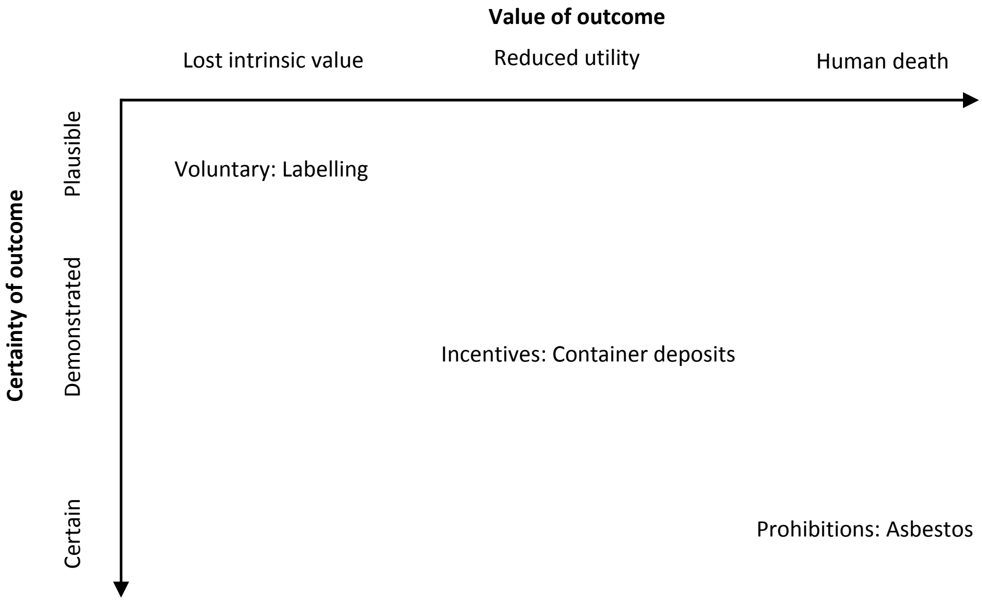


Figure 2. The relationship between certainty and magnitude of effect, in the context of policy responses

5 Monitoring marine litter

The stock of plastics in the ocean is distributed among the five oceanic compartments described in Section 2:

- the ocean's surface
- the sea floor
- the shoreline and coastal margin
- the water column
- biota.

The physical and chemical processes acting on the microplastics in each of these reservoirs are different, and the risks and opportunities for mitigation might also be very different. Separating the total stock of plastic into the five compartments will therefore help us to better understand the location of litter hotspots and the processes that lead to their formation.

Most of our understanding of marine litter comes from coastal clean-ups and land-based litter surveys. With the possible exception of the ocean surface, there is a severe paucity of data on the amount of plastic in each of the compartments, and even less is known about the fluxes of plastic between them.

Closing the global plastic budget will require large-scale, targeted sampling of all of the compartments. However, it may be possible to prioritise those investigations according to ease of sampling and likely importance, based on our current understanding of their relative contributions to the total volume of plastic in the environment.

5.1 The ocean's surface

Of all the compartments, the surface ocean is the best sampled, and the large volume of trawling data has now been combined is a global dataset (van Sebille *et al.* in press). Approximately half of the floating plastic detected from surface sampling is in the North Pacific and North Atlantic, while the other half is estimated to be in areas that are relatively undersampled, such as the southern portions of the Atlantic and Pacific oceans, the Indian Ocean and the Southern Ocean. There is also a size bias in the data, as most of the surface data are the product of trawls using nets designed for plankton, which typically sample items in the range between 300 µm and 25 cm, due to mesh size and the size of the mouth of the net. However, there are a number of observations that suggest that larger items follow the same general pattern, although with higher concentrations near coastal and oceanic sources (for example, Ryan 2013).

5.1.1 Monitoring methods

Sampling methods are relatively well established for the at-sea measurement of marine plastics. Larger items are often sampled using visual surveys by observers aboard ships of opportunity or research vessels (*sensu* Thiel *et al.* 2003; Eriksen *et al.* 2015; Ryan 2013; Hinojosa & Thiel 2009), and analysis uses distance sampling methods (for example, Ryan 2013). While this approach can provide quantitative estimates of densities, combining estimates from multiple surveys remains difficult because of differences in observability and other factors (Ryan 2013). In addition to the expense and logistics, given the expansiveness of the ocean and humans' capacity for visual detection, shipboard visual surveys are unlikely to be feasible for assessments of floating litter at the global scale.

Smaller debris, such as microplastics, has been sampled using various types of surface nets developed for other purposes, such as plankton sampling (Cozar *et al.* 2014; Erikssen *et al.* 2014; Law *et al.* 2010; Reisser *et al.* 2013, 2014). These methods (Figure 3) were relatively well established and standardised before they were applied to marine debris. As a result, there has been a reasonable measure of success in combining measurements across surveys for analysis (Cozar *et al.* 2014; van Sebille *et al.* 2015). However, given the vastness of the ocean, the cost of operation and the large distances involved, floating plastic litter surveys are often limited to particular regions, leaving significant gaps in the global coverage.

Remote sensing has been explored as an alternative method for estimating densities of plastics in marine systems. Satellite- and drone-based instruments have not yet provided useful data, largely because of the analytical complexity of identifying items automatically and the size distribution of the items (NOAA 2010). There is some possibility of using ship-based instruments, particularly on ships of opportunity. However, some technical feasibility analysis remains to be done (C. Wilcox, unpublished data).

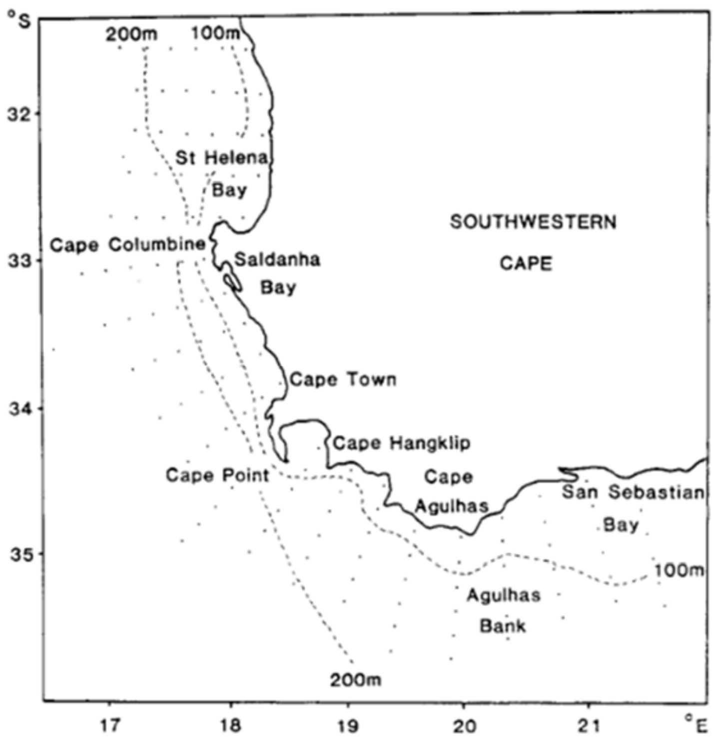


Figure 3. Trawl survey locations and local isobaths for 1,224 neuston trawls carried out in 1977 and 1978

Note the grid layout for surface tows.

Source: Ryan (1988).

5.1.2 Examples of sea-surface monitoring programs

One of the largest floating plastic litter datasets has been collected by the Sea Education Association (SEA).[1] SEA runs a program that combines educational opportunities for students with data collection for surface microplastics (among other activities). The 25+ year dataset of plastic counts for more than 7,500 surface tows is the longest standing monitoring of ocean plastics. The results from these efforts have been fundamental in informing the global discussion on surface plastic debris (Law *et al.* 2010; Law & Thompson 2014; van Franeker & Law 2015). The program continues to collect data based on voyages that take place each year (Figure 4).

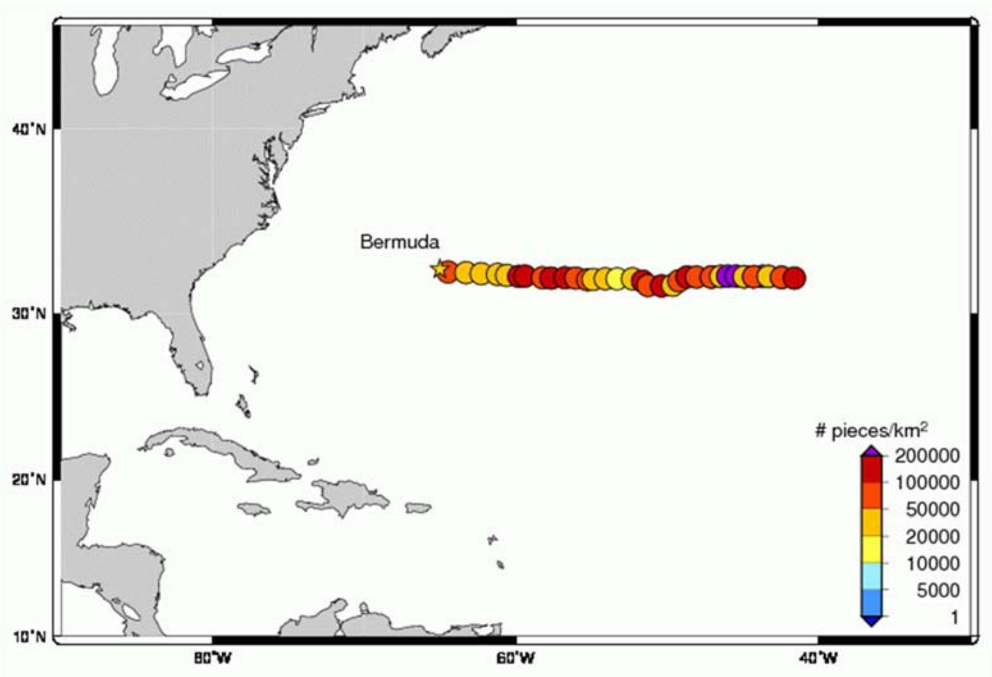


Figure 4. Surface tow samples from near Bermuda in 2010, indicating very high plastic concentrations

Source: SEA website, http://www.sea.edu/plastics2010/science-results6-21.htm.

In 2010, the Spanish Government supported the Malaspina circumnavigation expedition. This interdisciplinary research endeavour set out to assess global change, explore biodiversity and raise interest in marine sciences. During the expedition, researchers collected floating plastic debris from surface trawls. Results from the expedition found the concentration of plastic debris in surface waters of the global ocean to be less than expected, but the researchers pointed out potential sampling bias resulting in the loss of small particles and identified a gap in the size distribution reported for floating plastic debris (Cozar *et al.* 2014). Metadata are available for public and authorised users.[2]

---

[1] SEA; http://www.sea.edu/plastics.

[2] Malaspina Digital, http://scientific.expedicionmalaspina.es/#!/n/malaspina-digital/s209.

The Algalita Research Foundation, founded by Captain Charles Moore, also has a long-term monitoring program that has been focused on plastic pollution, particularly in the ‘Great Pacific Garbage Path’. Expeditions have taken place to survey floating plastic in this accumulation region since the late 1990s. Researchers there are evaluating long-term trends and changes in floating plastic pollution. This monitoring is expected to continue (Eriksen *et al.* 2015).

Many other monitoring efforts and surveys of the ocean’s buoyant plastics have taken place over the past several decades, from surveys in South African waters (Ryan 1988) and the Antarctic (Barnes *et al.* 2010) to the Arctic (Bergmann and Klages 2012) and elsewhere.

Harmonising approaches to data collection, continued sharing of datasets and international collaboration will clearly be an important means of increasing knowledge and identifying target areas for remediation and source reduction.

5.1.3 Recent syntheses

Greater emphasis is now being placed on the value of combining data from multiple surveys and monitoring programs (Cozar *et al.* 2015; Eriksen *et al.* 2015) and on comparing model solutions. Recently, van Sebille and colleagues (2015) compared estimates of microplastic abundance and mass using a rigorous statistical framework and empirical surface trawl data (see Section 6.2.2).

5.2 The sea floor

Plastics with a density that exceeds that of seawater (>1.02 g/cm^3) will sink and accumulate in the sediment, while low-density particles tend to float on the sea surface or in the water column. If 70% of plastics are known to eventually sink (Barnes *et al.* 2009), increased monitoring the ocean floor is clearly essential. It has been suggested, however, that even low-density plastics can reach the sea floor.

Monitoring of macroplastics on the seabed has shown mixed patterns to date. There is some evidence of increased levels of plastic debris in proximity to cities, but conflicting evidence shows offshore areas to have higher densities of debris (Corcoran 2015).

5.2.1 Monitoring methods

Surveying the deep ocean is difficult and expensive, and many monitoring methods have been used. Useful methods have included using bottom trawl nets such as those used in commercial fishing, imaging technologies such as remotely operated vehicles, manned or unmanned submersibles, towed camera systems, or scuba diving enthusiasts.[3]

Sampling methods can be active (such as sediment coring) or passive (such as using sediment traps), but most analysis to date has used active methods in a mix of video, sonar and trawl sampling. All three methods are challenging. Video sampling is very labour intensive, as it

[3] For example, Project Aware, www.projectaware.org.

generally involves substantial manual processing. It usually also covers relatively small areas, making it difficult to use except in areas of very high debris density. Sonar has been used in a number of studies; however, while it overcomes some of the sampling scale issues associated with video, it is limited by the balance between spatial coverage, spatial resolution and signal strength. In contrast to video or sonar, trawl sampling requires relatively little processing and can be done on board vessels as they operate. However, trawl sampling is typically done using fish or invertebrate trawling gear and is often an add-on to the operation, not its primary purpose. It is usually done using mesh sizes greater than 100 mm, is limited to locations with soft sediment substrates and is done only in areas of interest to commercial fisheries, leading to spatial bias and the over-representation of fisheries wastes.

Sediment coring and grab sampling do not suffer from the same bias as trawl sampling. However, they have even smaller coverages than any of the other active methods, implying that they will have very low detection rates except in the most polluted locations. One advantage is the potential for estimating time patterns of deposition, using markers to age sediment cores in areas of sediment deposition.

Passive sampling, for example using sediment traps, could provide an alternative, as it does not necessarily have the same spatial bias as trawls and it is possible to develop relatively low-cost equipment. However, it suffers from bias in how materials are captured if the equipment is not directly relevant for the transport mechanisms. For instance, many sediment traps capture descending materials, but this means that they miss any laterally transported materials, such as those washed along the bottom by tides or currents. Passive sampling equipment also suffers from low coverage, making it difficult to use in a representative way without either very large sampling effort or very high plastic densities.

A comprehensive review of monitoring methods for evaluating debris on the seabed is in Galgani & Andral (1998).

5.2.2 OSPAR benthic monitoring

Seabed litter is the newest of the marine litter indicators that have been developed by the OSPAR Commission. It is used to assess trends in the amount of litter deposited on the sea floor and in analyses of its composition, spatial distribution and, where possible, sources. The monitoring is done through international bottom trawl surveys for fisheries management firms, which have adopted a protocol to monitor 39 commonly found litter items caught in their nets. The 39 types are split into six categories: plastic, metal, rubber, glass/ceramics, natural products/clothes and miscellaneous. The advantage of using fisheries trawls is that all the information about the gear type, the area swept and the trawling speed is already collected, allowing the number of items per square kilometre of seabed to be calculated (Figure 5).

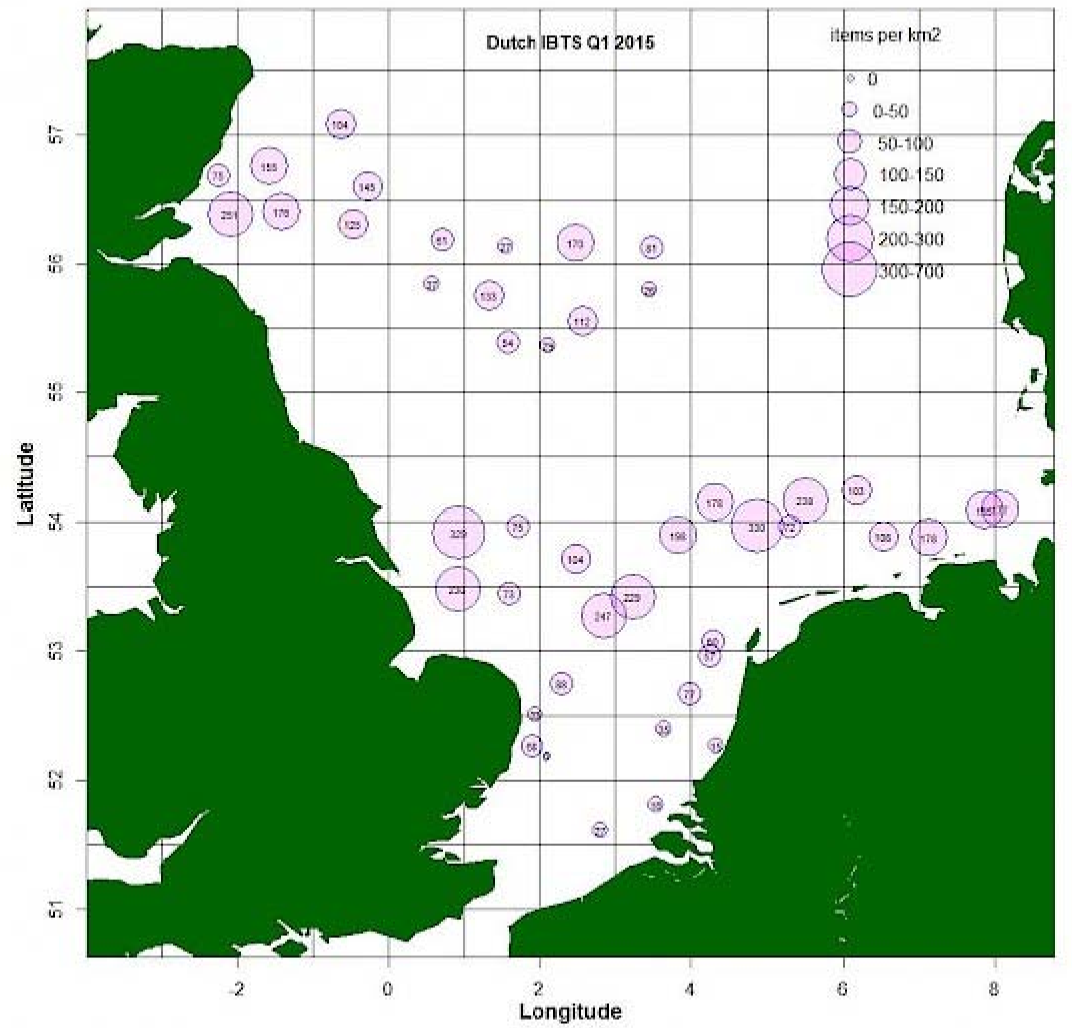


Figure 5. Density of litter items per haul per square kilometre from the 2015 International Bottom Trawl Survey

Note: The numbers in the circles are the numbers of items per square kilometre at sampled areas.

Source: OSPAR website, http://www.ospar.org/work-areas/eiha/marine-litter/marine-litter-indicators.

5.2.3 Other examples of sea-floor and sediment monitoring

Between 1992 and 1998, a research team sampled the continental shelf and slopes in European seas during nearly 30 oceanographic cruises. It found high geographical variation in concentrations and types of litter items, although plastic was the dominant material type (70% of all items found) (Galgani *et al.* 2000). Due to the multi-year monitoring program and geographical expanse of the study, the researchers were able to identify spatial and temporal trends and to detect the influence of local activities, geomorphological factors and riverine inputs.

Stefatos and colleagues (1999) reported on marine debris detected during sea-floor surveys in western Greece. They attributed the high percentage of beverage containers detected during their surveys to ship-based traffic, and attributed packaging items to land-based sources.

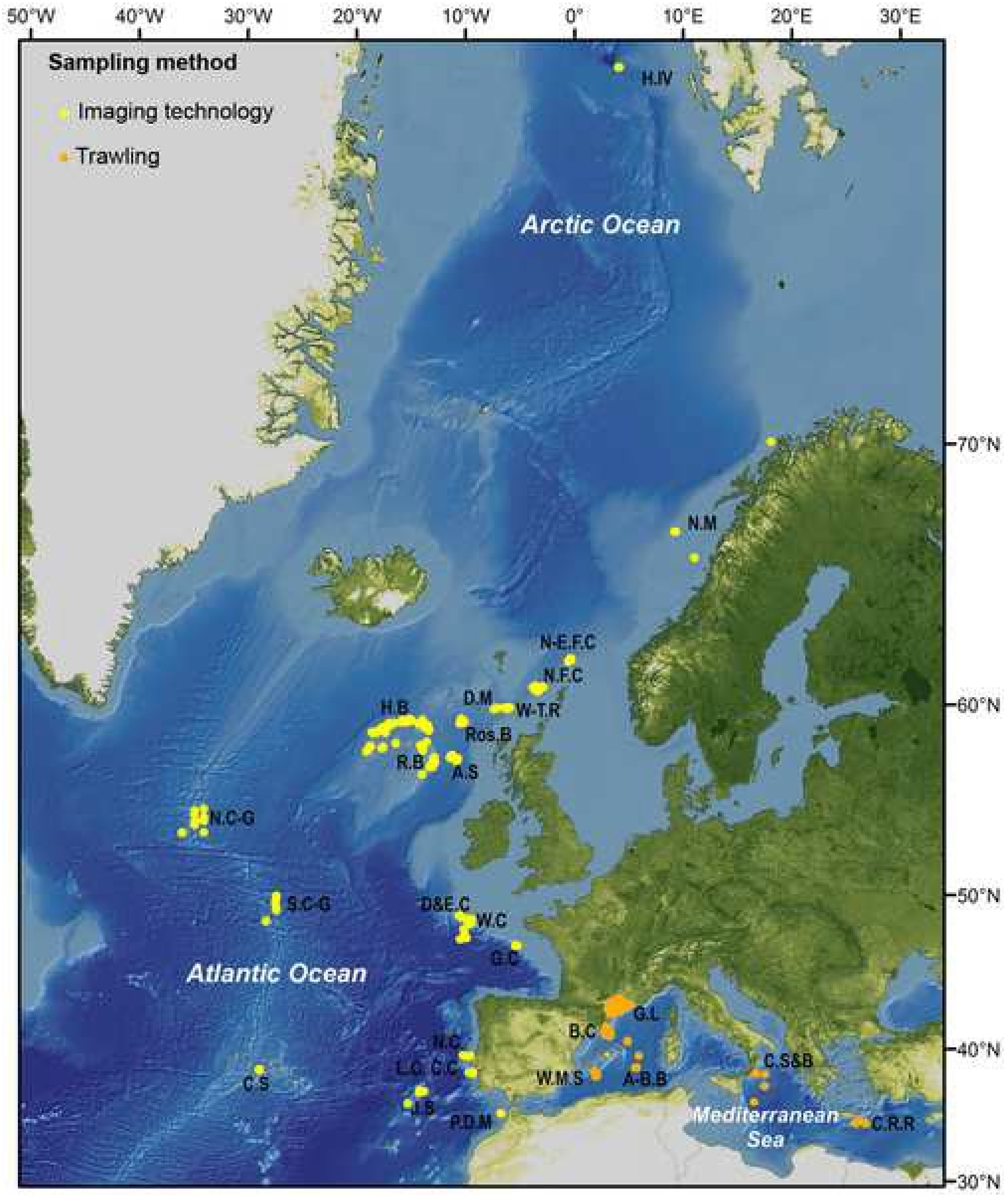


Figure 6. Locations of study sites sampled with imaging technology (ROVs, manned submersible, towed camera systems) and trawling

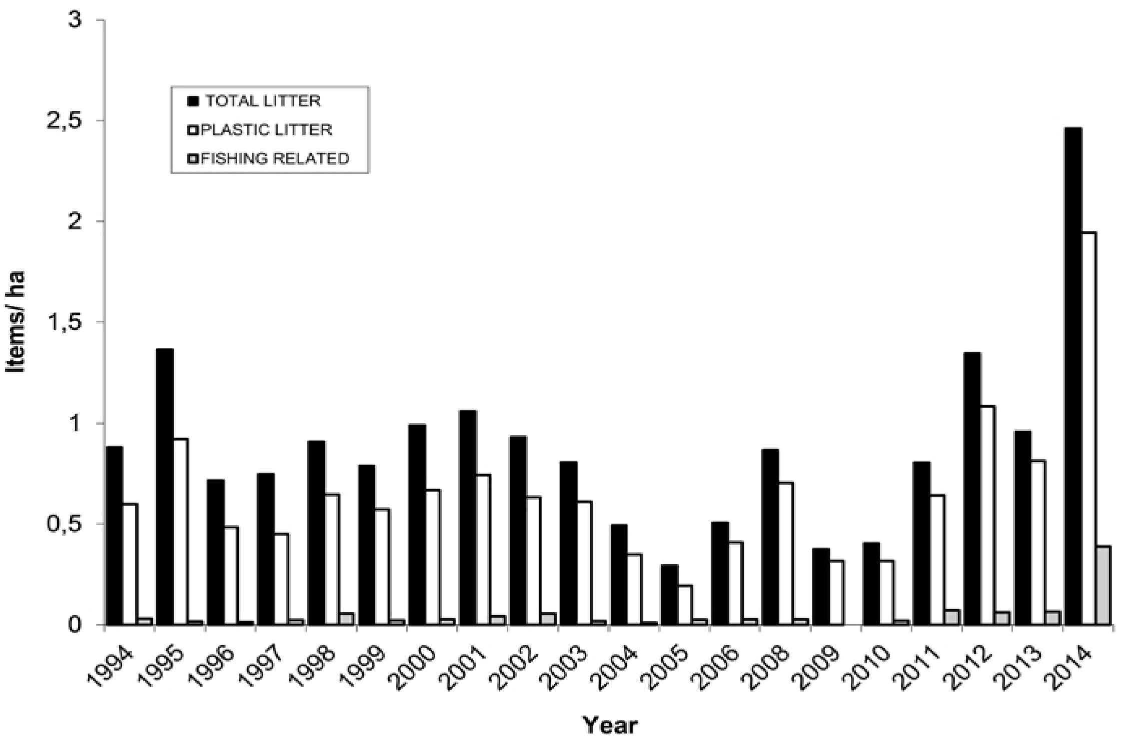
A-B.B = Algero-Balearic Basin (W. Med.), A.S = Anton Dohrn Seamount, B.C = Blanes Canyon (NW Med.), C.C = Cascais Canyon, C.S = Condor Seamount, Calabrian Slope & Basin = C.S&B, Crete-Rhodes Ridge = C.R.R, D&E.C = Dangeard & Explorer Canyons, D.M = Darwin Mounds, G.L.C = Gulf of Lion canyons (NW Med.), G.L = Gulf of Lion, G.C = Guilvinec Canyon, H.B = Hatton Bank, H.IV = HAUSGARTEN, station IV, J.S = Josephine Seamount, L.C= Lisbon Canyon, N.C = Nazaré Canyon, N.C-G = North Charlie Gibbs Fracture Zone, N-E.F.C = North-East Faroe-Shetland Channel, N.F.C = North Faroe-Shetland Channel, N.W = Norwegian margin, P.D.M = Pen Duick Alpha/Beta Mound, R.B = Rockall Bank, Ros.B = Rosemary Bank, S.C = Setúbal Canyon, S.C-G = South Charlie Gibbs Fracture Zone, W.C = Whittard Canyon, W.M.S = Western Mediterranean slope, W-T.R = Wyville-Thomson Ridge.

Source: Reprinted from Pham *et al.* (2014).

In addition to the monitoring methods described in Section 5.2.1, other approaches have been used to gain insight into the density and distribution of seabed litter. For example, the Fishing for Litter project in Dutch waters collected more than 500 tonnes of debris between 2000 and 2006, including tyres, refrigerators, packaging material, lost shipping items, fishing gear, ropes and many other items (Hammer *et al.* 2012; KIMO 2010).

The Korean government recently supported the removal of derelict fishing gear from the seabed of the East Sea. Fishers used bottom trawling with heavy hooks and ropes to remove the litter. In 2009 and 2010, 460 tonnes of debris was removed (Cho 2011).

In the Mediterranean Sea and within the European Data Collection Framework, the Mediterranean International Bottom Trawls Surveys (MEDITS) program[4] intends to produce basic information on the population distribution and demographic structure of benthic and demersal species on the continental shelves and along the upper slopes (80–800 m). The program will use systematic bottom trawl surveys and a common, standardised sampling method and protocols. The latest version of the protocol includes a common standard for the voluntary collection of data on marine litter, in agreement with the requirements of the European Commission's Marine Strategy Framework Directive and the Barcelona Convention Regional Action Plan on Marine Litter. This will facilitate the organisation and regular collection of data and allow assessments of litter at the basin scale. To date, 1,280 sampling stations are considered, covering mostly (but not only) the European coastline, and there remains the potential to extend the monitoring to the wider basin region. As an example, Figure 7 shows results from the Gulf of Lion where monitoring began in 1994, enabling the evaluation of trends through time.



[4] International Bottom Trawl Survey in the Mediterranean, http://www.sibm.it/SITO%20MEDITS/principaleprogramme.htm.

Figure 7. The density of litter collected on the sea floor between 1994 and 2014 in the Gulf of Lion, Mediterranean Sea

Litter (mean values for 70 sites) was collected during the Mediterranean International Bottom Trawl Surveys (MEDITS) cruises dedicated to fish stock assessments using a stratified sampling scheme and 20 mm mesh. The protocol is available at http://www.sibm.it/SITO%20MEDITS/principaleprogramme.htm. Results are expressed as items/ha.

Source: Galgani (2015), with permission; http://dx.doi.org/10.3389/fmars.2015.00087.

5.3 The shoreline and coastal margin

Understanding the sources of plastic litter on the shoreline and coastal margins and trends over time remains difficult. However, there has been extensive coastal sampling, largely through volunteer clean-up programs such as the International Coastal Cleanup (ICC) organised by Ocean Conservancy, which has been operating in more than 70 countries over nearly 30 years.[5] Other such work includes Project AWARE's underwater litter surveys, and many local, regional and national efforts around the world.

5.3.1 Monitoring methods

Many of these projects have significant spatial and temporal coverage. However, they are typically focused specifically on debris removal and they often do not follow sampling designs that readily lend themselves to analysis. Another challenge is a lack of methodological consistency for repeated surveys. Information collected by volunteers is often recorded and reported intermittently, incompletely or not at all. This irregularity can reduce our ability to perform large-scale, statistically robust analyses that can yield important insights about the effectiveness of litter policies, awareness campaigns or other activities. However, some statistical approaches could be used to overcome these challenges to some extent.

Coastal litter monitoring typically uses individuals (often volunteers but also trained participants or paid staff) to record information that is identified visually. This means that there tends to be a size bias depending on people's visual acuity. Also, items with monetary value or large items may be more likely to be collected than smaller particles, dangerous objects or objects that are difficult to identify.

However, some sampling methods focus on smaller items (such as micro- and nanoparticles). Those approaches require different survey methodologies, which are focused on the identification and capture of small particles and are similar to those mentioned in Section 5.2.1 on ocean-floor monitoring (see the review in Cole *et al*. 2011). NOAA also recently released a protocol for analysing and quantifying synthetic particles in the water and in sediments (Masura *et al*. 2015).

Coastal litter surveys remain one of the easiest and most cost-effective means of providing an index of marine litter (Dixon & Dixon 1981; Merrell 1985). However, a number of different methodologies and approaches are currently used to identify, count, sort and quantify the litter. While many methods overlap or are similar, few collect or report information using a systematic,

---

[5] Ocean Conservancy, http://www.oceanconservancy.org.

statistically robust approach, even though researchers pointed to the importance of standardising survey methodologies many years ago (Ribic et al. 1992).

Another approach to monitoring coastal litter that has recently gained popularity uses litter traps. Litter traps and boom systems are generally designed to collect floating litter and other debris and can be used to monitor and sample litter in coastal environments. Traps can continue to work regardless of tidal flux or other changes in water levels and without impending water flow. However, they can break down or reach their maximum capacity in high-flow situations, such as during extreme weather. Floating litter traps also require a system for collection and removal. One novel litter trap that has recently gained praise for its effectiveness in removing litter and in raising awareness of the issue is the Baltimore Water Wheel, which uses solar and hydropower to collect debris from the Jones Falls River (WPB, n.d.)

Remote sensing can also be used for shoreline and coastal monitoring. Options include drones, satellites and balloons equipped with cameras, as well as ground-based imagery (Kako *et al*. 2012; Jang *et al*. 2015). While the identification of small items can be challenging and image-processing time can be restrictive, automated image processing may make remote sensing a more practical monitoring tool in the near future.

5.3.2 Examples of coastal litter monitoring programs

Ocean Conservancy's International Coastal Cleanup is an annual event. In 2013, nearly 650,000 volunteers collected about 6,000 tonnes of trash in around 100 countries. The clean-up's collection and compilation methods have been systematically developed and integrated to quantify the results. While the dataset has broad geographical representation over nearly 30 years, there are missing data points. Some sites and regions are not represented every year, and monitoring methods have been modified over time. Furthermore, some volunteers do not record all the information, and in some cases data have been combined before they were submitted to Ocean Conservancy, thus losing spatial resolution and statistical power.

In Korea, there is a national beach monitoring program, which was initially led by the Korea Marine Rescue Center and subsequently by the Korea Marine Environment Management Corporation in collaboration with volunteers who collect coastal litter. The main aim of the program is to assess the level of beach debris pollution and to aid in the identification of management priorities for coastal debris in Korea (Hong *et al.* 2014). The project aims to collect data on the levels of marine debris pollution and to identify litter sources. The 20 monitoring sites are well distributed around the coastline (Figure 8).

The Korean Government has responded to the marine litter issue by investing in gear and litter retrieval programs and research projects since the 1990s. In 2008, the Marine Environment Management Act was revised, providing a legal basis for the management of marine litter. Without reliable scientific information on the sources, types, distribution and impacts of marine debris, however, identifying targets has been challenging.

Figure 8. Marine debris coastal monitoring sites in Korea

Source: Hong *et al.* 2014; used with the authors' permission.

In the United States, marine debris issues is within the mandate of NOAA. NOAA's Monitoring and Assessment Project has developed guidelines and a toolbox for coastal litter monitoring (NOAA, n.d.). It currently runs two long-term, repeated-survey monitoring programs. One is a standing-stock litter survey in which litter is observed and counted but not removed, and the other is a repeated coastal survey in which coastal debris is removed. However, the consistency and frequency of data collection among monitoring sites has been variable, and maintaining ongoing efforts will require sustained funding. The NOAA marine debris program also funds and has otherwise supported many outreach, education and research projects since the program's inception in 2006.

The Australian Government has a national policy and threat abatement plan to address marine debris, and particularly its impact on threatened marine vertebrates (DEE 2009). Many volunteer clean-up efforts and coastal care initiatives are led by not-for-profit groups, such as the Surf Rider Foundation, Tangaroa Blue, the Two Hands project, Take 3 and Clean Up Australia. In addition, many state and local governments host coastal debris clean-ups and litter prevention and awareness raising campaigns.

These efforts yielded some information on debris along the coastline, but no large-scale systematic dataset. In response, a research team from the Commonwealth Scientific and Industrial Research Organisation (CSIRO) developed a statistically robust coastal survey method that was then applied nationally (Figure 9). The method controlled for sampling bias to estimate the distribution of debris along the entire coastline of the continent (Hardesty *et al.* 2015). The team also investigated factors influencing the contribution of terrestrial sources to coastal debris and made recommendations to reduce coastal litter.

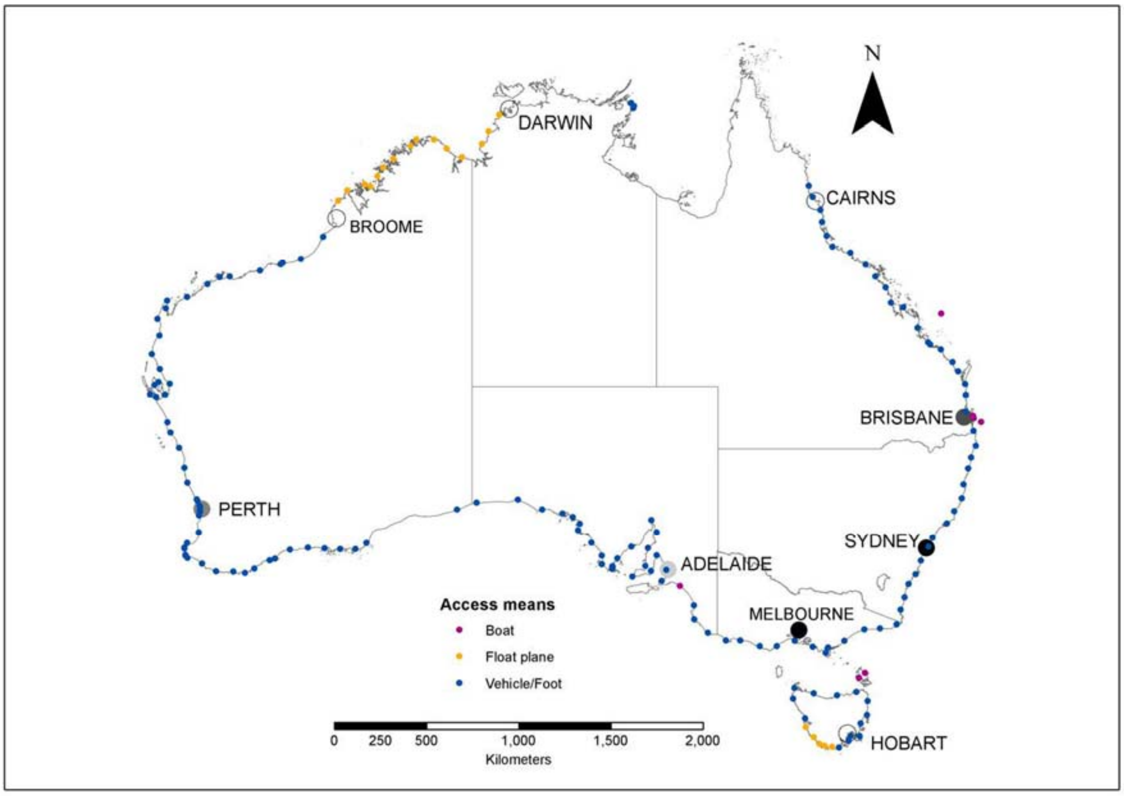


Figure 9. Locations of and means of access to CSIRO coastal debris survey sites around mainland Australia and Tasmania

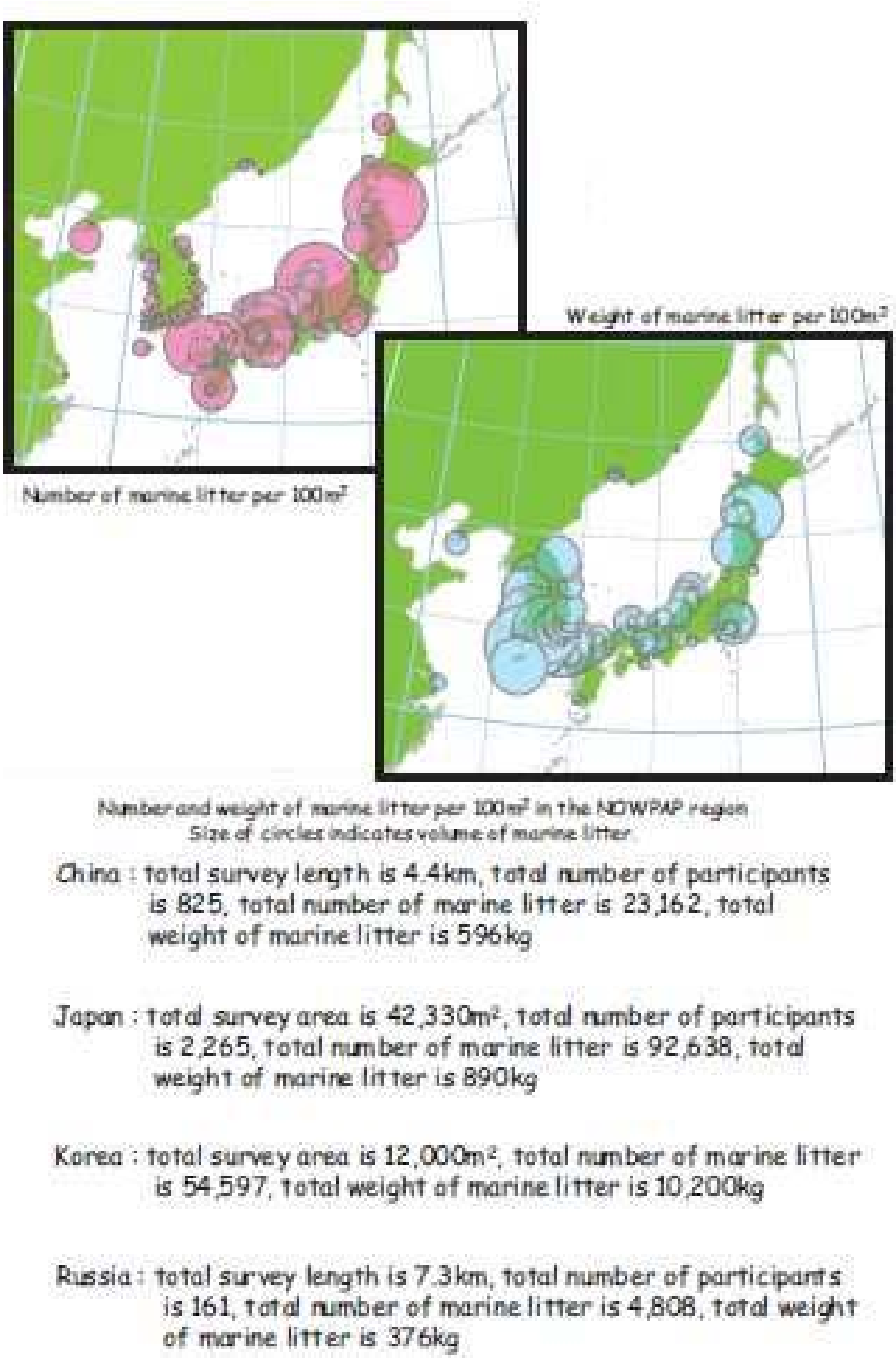
In Europe, the Marine Strategy Framework Directive (EC 2008), which decrees that European Union member states shall determine a set of characteristics that define 'good environmental status' for their relevant waters, based on a list of 56 indicators that includes four indicators specific for marine litter. In addition, regional plans have been adopted for the management of marine litter in most European waters.

The OSPAR Commission, which was created under the Convention for the Protection of the Marine Environment of the North-East Atlantic, has committed to reducing marine litter and protecting vulnerable species and habitats in those waters. It has developed the Marine Litter Regional Action Plan, which addresses both land- and sea-based litter sources (OSPAR 2014). Within the OSPAR region, more than 250 surveys take place in coastal areas around Europe (Figure 10). Surveys follow a standard data collection protocol, and the information is posted and freely available online.

Figure 10. Beach litter survey sites in the OSPAR region, where repeated monitoring uses consistent protocols

Source: OSPAR beach litter survey website, http://www.mcsuk.org/ospar/map.

Monitoring is also underway in the Northwest Pacific Action Plan (NOWPAP) region (East China, Korea, Japan and far eastern Russia). In this understudied region, a snapshot of marine litter was created from information collected at 82 beaches (2 in China, 20 in Korea, 53 in Japan and 7 in Russia) in 2009 (NOWPAP CEARAC 2009).



Numbers of items and weight of marine litter per 100 m^2 in the NOWPAP region.

Size of circles indicates volume of litter.

	Survey length/area	No. of participants	No. of litter items	Total weight of litter
China	4.4 km	825	23,162	596 kg
Japan	42,330 m^2	2,265	92,638	890 kg
Korea	12,000 m^2	Not known	54,597	10,200 kg
Russia	7.3 km	161	4,808	376 kg

Figure 11. Volume of marine litter in each NOWPAP member state, based on coastal litter surveys

Source: NOWPAP CEARAC (2009).

In China, nationwide monitoring started in 2007 at beaches, on the sea surface and at sea-floor sites from eastern to southern coastal areas (Figure 12). The number of sites has been increasing, offering snapshots of litter in multiple locations during different years. The results from the annual monitoring have been shared through the *Bulletin of Marine Environmental Status of China*. In addition, monitoring results from surveys carried out at nine beaches around the northern South China Sea in 2009 and 2010 were reported by Zhou *et al.* (2011).

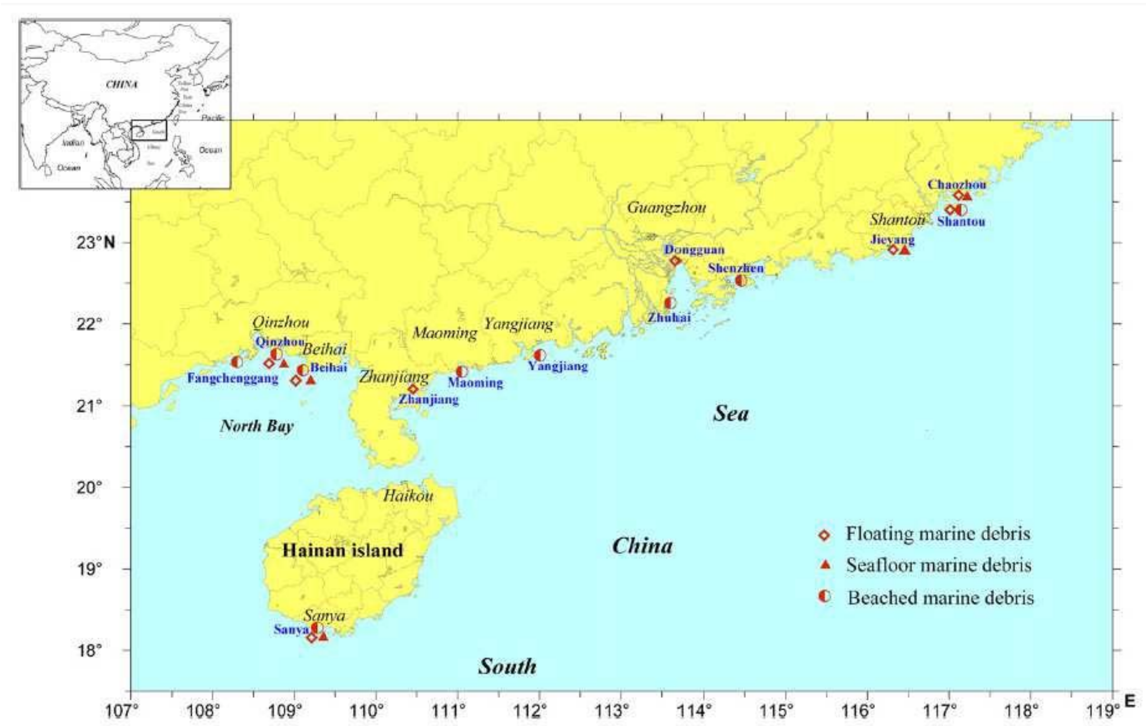


Figure 12. Survey sites in coastal China for floating, sea-floor and beached marine debris

Source: Zhou *et al*. (2011)

Many other clean-up efforts, litter prevention and reduction programs and surveys take place idiosyncratically, opportunistically and through well-organised programs around the world (Table 2).

Table 2. Summary of historical and recent coastal litter surveys

Location Sample site	Year	Frequency (no. of surveys	Sites (no.)	Citizen / scientist	Reference
South Africa					
Central Transkei (local)	1994–1995	Monthly (13)	6	S	Madzena & Lasiak (1997)
South Atlantic					
South Georgia, Bird island (local)	1995	Monthly (6)	1	S	Walker *et al.* (1997)
Falkland island (local)	2002	Monthly (4)	1	S	Otley & Ingham (2003)

Macquarie island (local)	2001	Monthly (12)	1	S	Eriksson & Burton (2003)
Candlemas island (local)	1997	Monthly (1)	1	S	Convey *et al.* (2002)
North Atlantic					
Europe (international)	2001–2006	Seasonally (24)	51	C	OSPAR (2007)
Nova Scotia (Canada, local)	2005	Monthly (6)	1	S	Walker *et al.* (2006)
North Pacific					
NOWPAP (international) Japan (local)	2009	Once	82	C	NOWPAP CEARAC (2009) Shimizu *et al.* (2008)
Japan (local)	2004–2005	Bimonthly (7)	3	S	Shimizu *et al.* (2008)
Japan (national)	2009	1–4	53	C	NOWPAP.org
Korea (national)	2008–	Bimonthly	20 (40 since 2015)	C	Hong *et al.* (2014)
China (national)	2007–	Once	14 (increase)	C	Hu (2010)
China (Taiwan, local)	2009–2010	Bimonthly	4	S	Liu *et al.* (2013)
	2012–2013	Seasonally	6	S	Kuo & Huang (2014)
South Pacific					
Australia (national)	2011–2013	Once	560	S, C	Hardesty *et. al* (2014)

Source: Adapted from Hong (2013).

Much of the world's coastline has not been subjected to coastal litter surveys or is undersurveyed, particularly where access is difficult, such as on remote islands and in parts of the developing world (Figure 13). This is partly due to logistical difficulties, transport challenges and the cost of access. Furthermore, many parts of the world lack infrastructure to support coastal litter monitoring.

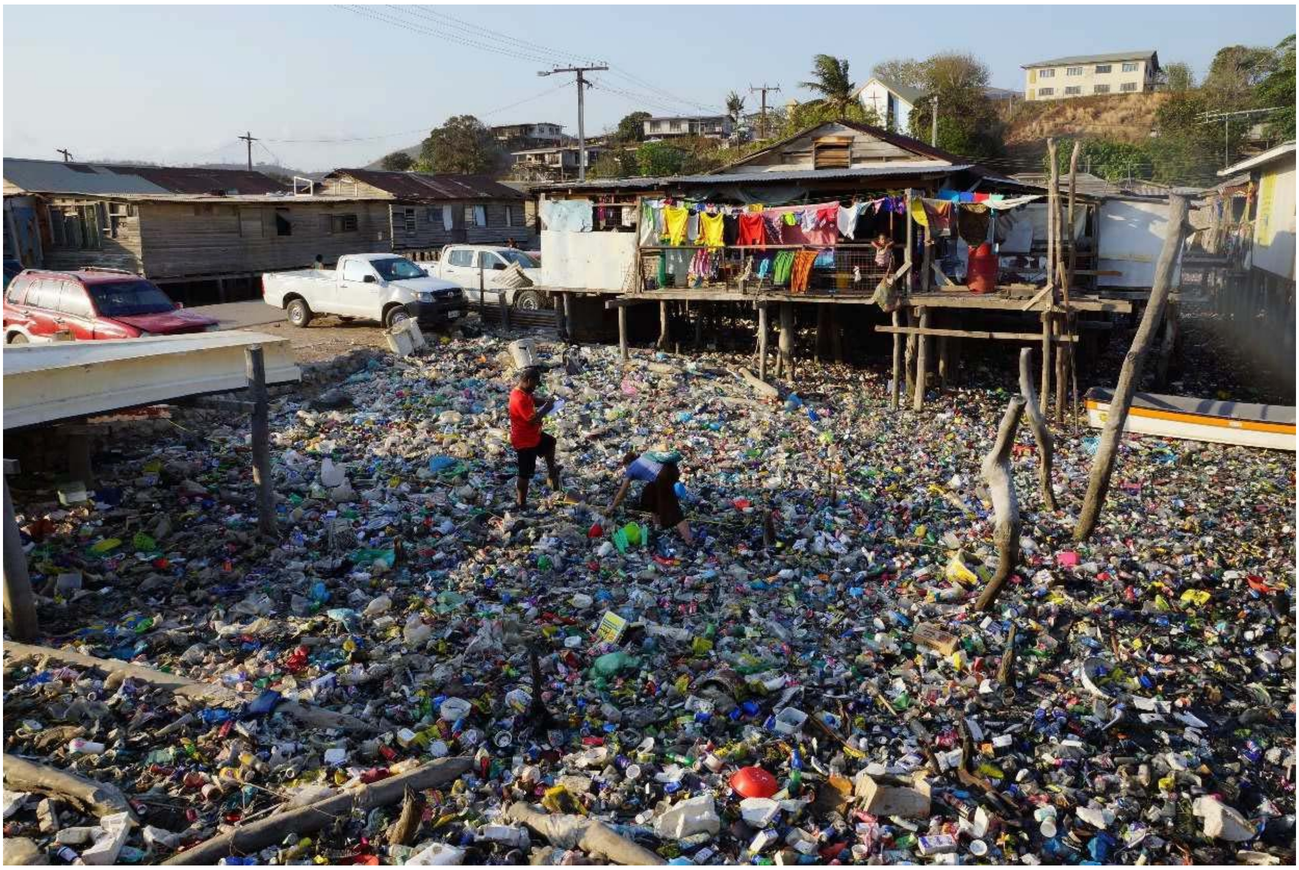


Figure 13. Coastal community in Papua New Guinea, surveyed for coastal litter in 2015

Source: © Sustainable Coastlines Papua New Guinea.

5.4 The water column

The most extensive sampling of the vertical distribution of plastics in the water column to date examined the abundance and mass of plastic debris with depth in the North Atlantic gyre, which is an area of high plastic concentration.

Overall, empirical studies, statistical models and physical models suggest that debris in the water column is mainly confined to regions near the surface and the ocean floor.

5.4.1 Monitoring methods

Our understanding of plastic litter in the water column has large knowledge gaps. To date, few have surveyed the vertical distribution of plastics throughout the water column, and most monitoring has been at the surface or within the upper 5–10 metres (Reisser *et al.* 2015).

There are no large-scale synoptic datasets that include vertical stratification in sampling (aside from the uppermost metres to tens of metres of the ocean surface). Sampling using bottom trawl and subsurface trawl nets is one approach to sampling the water column for non-buoyant litter items. Using video, still photography and other visual imagery in oceanographic surveys provides additional opportunities for sampling below the ocean's surface.

There are also opportunities to use video or other cameras in oceanographic sampling at fixed locations for other long-term survey projects. Continuous plankton recording surveys that are underway to evaluate marine ecosystem health, including in Australia's oceans, could also incorporate plastics sampling. Using below-surface sampling and towed behind ships of opportunity, this low-cost sampling approach is a promising survey tool that could be applied to gain a better understanding of subsurface and water column marine litter abundance, density and movement.

5.4.2 Examples of water column monitoring

Lattin and colleagues (2004) carried out monitoring at three depths in coastal California, where they surveyed the ocean surface (using a manta net), the mid-depths (using a bongo net) and the ocean floor (using an epibenthic sled). In general, an improved understanding of the vertical transport and movement of plastics between ocean compartments is needed to improve estimates of the size distribution, concentration and missing stock of plastics in the ocean (Kukulka *et al.* 2012; Law *et al.* 2014; Isobe *et al.* 2014).

5.5 Biota

In some cases, marine species can be used as indicators of ecosystem health and to identify hotspots of marine litter in the ocean.

5.5.1 Monitoring methods

Historically, seabirds were shot on the wing and plastics ingestion was identified as part of diet studies during necropsies (Ainley *et al*. 1989; Spear *et al*. 1995). Increasingly, dissections or necropsies to monitor marine plastics use beach-washed or beach-wreck birds (van Franeker *et al*. 2010, 2011; van Franeker & Law 2015; Carey 2011; Acampora *et al*. 2013; Ryan 2008).

Because marine species are often difficult to study on the ocean, surveys of faecal pellets and boluses (Hutton *et al*. 2008; Nilsen *et al*. 2014) have been used to identify the frequency of plastic ingestion and, importantly, to identify regions of the ocean with high concentrations of plastic. Animals in captivity for rehabilitation (such as marine turtles) have been known to excrete plastic that they have ingested. Linking foraging areas and the risk of plastic ingestion (*sensu* Wilcox *et al*. 2016; Schuyler *et al*. 2016) provides an excellent opportunity to identify risk hotspots, risk species, and regions in which to focus on reducing litter inputs.

Lavage of live animals (as reviewed in Karnovsky *et al.* 2012) or endoscopy (Sievert & Sileo 1993) can also be used to assess the frequency and quantity of plastic ingestion and has long been used for seabirds.

However, lavage can be stressful to birds and does not result in the voiding of the entire gastrointestinal content (Barrett *et al.* 2007; Neves *et al.* 2006). Endoscopy is difficult and time consuming and cannot yield indigestible matter below the stomach oil surface (Sievert & Sileo 1993) and necropsies of birds typically use biased samples (Hardesty *et al.* 2015).

However, recent advances show promise. A newly described method for assessing live seabirds' exposure to plastics through minimally invasive means (Hardesty *et al*. 2015) provides a way to assess the ubiquity of plasticisers occurring in multiple species with different body sizes, foraging strategies and geographical distributions. A similar approach has been trialled to detect phthalates in stranded whales (e.g. Fossi *et al.* 2012). Identifying traces of chemicals used in plastic production may increase our ability to sample additional species and geographical regions and to identify regions of greatest concern.

5.5.2 Examples of biota monitoring

Surveys of marine vertebrates for diet and plastic studies have included fish taxa (Boerger *et al.* 2010; Rochman *et al.* 2015), marine mammals such as whales (de Stephanis *et al.* 2013; Sechi and Zarzur 1999; Jacobsen *et al.* 2010) and dolphins (Baird and Hooker 2000), all of which have been found to ingest plastic. For the past several decades, researchers have also reported on plastic ingestion in seabirds (Ainley *et al*. 1990; Ryan 1987; Spear *et al*. 1995).

The OSPAR-initiated ecological quality objectives (EcoQOs) directly apply monitoring of seabirds to targets for acceptable ecological quality. The northern fulmar (*Fulmaris glacialis*) is the key EcoQO indicator species for long-term monitoring of plastic debris in the North Sea, based on the abundance of debris that the species ingests. The EcoQO target defined for plastic pollution in the North Sea is for fewer than 10% of fulmars to have more than 0.1 g of plastic in the stomach, based on sampling beach-washed birds. A similar approach is being considered for using loggerhead turtles (*Caretta caretta*) to act as ecosystem monitors in the Mediterranean (Hardesty *et al*. 2015).

6 Modelling marine litter

Research, educational, community engagement and outreach activities are underway around the world to understand, quantify, identify and reduce marine litter. With those activities comes a variety of monitoring opportunities. While monitoring is fundamentally important to assess the efficacy of measures aimed at solving the marine litter problem, it is complicated by the spatial and temporal heterogeneity in accumulation and movement of the debris and the multiple pathways that it can take.

Given those challenges, combining empirical data from monitoring and surveys with modelling approaches can be useful to help predict where plastic will occur in the marine environment. Numerical modelling can also be applied to back-track or hindcast where plastics in the ocean may have come from. Oceanographic current models can be used to identify where oceanic accumulation zones are most likely to occur.

Coupling such tools and approaches with species distribution maps and other ecological information, we can combine disparate data types to:

- predict or identify hotpots of risk to taxa or geographical regions of interest
- identify movement pathways or trajectories
- develop scenario analysis tools to identify potential sources and sinks
- evaluate the effectiveness of local actions and activities
- predict risks of invasion along pathways
- evaluate the costs of action and inaction.

6.1 The state of the models

6.1.1 Sources of data

Marine plastic litter is most often monitored in coastal areas, but monitoring can also take place at sea or through sampling animals that have encountered debris.

Along coasts, monitoring and surveys of litter are often part of clean-up activities or other community events. While this form of monitoring can provide crude estimates of debris types and abundance, it may use uneven sampling, involve only sporadic or patchy data collection and be biased in various ways. For example, it can be biased by the differential removal of litter items by beachcombing or beach dynamics.

At-sea monitoring, in coastal waters or on the high seas, can be expensive and difficult to replicate. Typically, oceanic monitoring uses surface trawl sampling, which is biased towards items in a particular size range—those that are small enough to be caught in nets but large enough to be discerned by the human eye. Surface sampling captures only floating objects and, given the vastness of the ocean, ocean circulation patterns and wind mixing, samples are often highly variable. At-sea sampling also requires large sample sizes to facilitate the statistical analysis needed to detect changes in distribution and abundance, given the high spatial and temporal heterogeneity of plastics in the ocean (Barnes *et al.* 2009).

Efforts have been made to survey subsurface marine litter (for example, Reisser *et al.* 2015) and the ocean floor (van Cauwenberghe *et al.* 2013a; Katsanevakis & Katsarou 2004; Galgani *et al.* 2000). This work has used a variety of methods, including bottom trawl nets, sonar, submersibles, snorkelling, scuba diving and manta tows (reviewed in Spengler & Costa 2008) but has lagged significantly behind coastal and surface sampling, probably because of the additional costs and time needed for such surveys.

Around the world, a number of different data collection strategies have been developed and used to monitor marine and coastal litter. While different questions require different monitoring approaches, the importance of standardising approaches cannot be overstated (Barnes *et al.* 2009). To date, monitoring methods and data recording have not been harmonised globally, and that remains an important goal.

Long-term monitoring is also costly, time consuming and difficult to sustain. Importantly, however, a number of long-term monitoring efforts are underway. They include the OSPAR Commission's[6] marine beach litter program in Europe, the International Coastal Cleanup organised by Ocean Conservancy[7] and NOAA's Marine Debris Program, which monitors coastal litter using multiple monitoring approaches.[8] These initiatives are important not only to detect long-term trends and patterns, but also to allow evaluations of the efficacy of legislation and to identify changes in sources, deposition, material types and impacts on wildlife. Furthermore, long-term monitoring can help to identify opportunities for impact through local actions.

6.1.2 The focus of modelling

A wide range of modelling related to marine debris has been undertaken, from models focused on the distribution of litter at sea (Maximenko *et al.* 2012) or emphasising sources (Lebreton *et al.* 2012) to models focused on ecosystem responses (e.g. Troost *et al.* 2015), ecological risk (Wilcox *et al.* 2015; Schuyler *et al.* 2015) or even ecological impact (Wilcox *et al.* 2014, 2016).

Here we focus on models concerned with the sources, transport, distribution and fate of debris. We largely leave aside questions of ecological impact, except as they relate to the ultimate fate of the debris.

---

[6] OSPAR Commission, http://www.ospar.org.

[7] Ocean Conservancy, http://www.oceanconservancy.org.

[8] Marine Debris Program, https://marinedebris.noaa.gov.

6.1.3 Reservoirs and fluxes of plastics

Plastic occurs throughout the marine water column, from the surface to the deep ocean floor. It can be in sediment, biota and ice, be trapped along the coastline or in estuaries, waterways and lakes, and be trapped in the atmosphere.

The reservoirs most relevant for modelling the movement of plastics in the ocean are the five compartments identified in Section 2 of this report: the ocean's surface; the sea floor; the shoreline and coastal margin; the water column; and biota.

Other reservoirs (such as the atmosphere, lakes and waterways) are less relevant to this modelling.

Movements of plastics between compartments are termed 'fluxes' (Figure 14).

RESERVOIRS & FLUXES

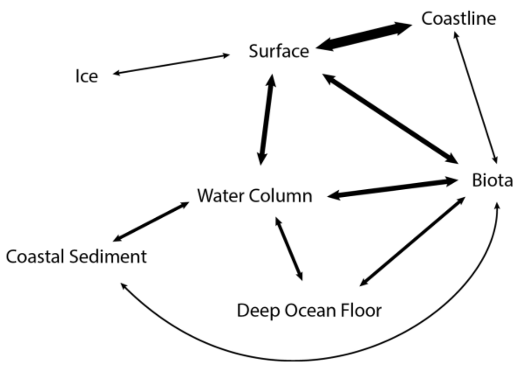


Figure 14. Reservoirs and fluxes for marine plastics

Note: The weight of the arrow indicates the magnitude of marine debris flux hypothesised to occur between compartments.

Evaluating budgets (losses, sources and sinks in the environment) or leakages between reservoirs requires an understanding of several key processes. The most important include:

- rates of fragmentation
- buoyancy, sinking and refloating rates
- rates and volumes of inputs of litter to the ocean
- time trends for plastics in the ocean.

Because our knowledge of reservoirs, fluxes and processes is incomplete, each of those elements involves uncertainty. Identifying uncertainties facilitates a ranking of the reservoirs, fluxes or processes to determine where effort should be focused, taking into account the key question—whether that relates to sources, losses between transition zones or impacts (Table 3).

Table 3. Transfers from reservoir to reservoir, and the approaches needed to increase our understanding and improve our models

	Surface	Ocean floor	Sediment	Ice	Biota	Coastline	Water column
Surface	Lagrangian modelling, field tracking experiments	Lab experiments / modelling / empirical	–	Modelling / field measure-ments	Field measure-ment / spatial analysis	Lab and field experiments	Lab experiments / modelling / empirical
Ocean floor	Lab and field experiments	Field experiments	Lab/field experiments	Field experiments	Empirical sampling	–	Lab/field experiments
Sediment	–	–	–	Field experiments	Lab experiments	–	Modelling / experiments
Ice	Modelling	–	–	Modelling / field observa-tions	–	Field observation s	Modelling
Biota	Lab/field	Lab / field / spatial analysis	Lab / field/ spatial analysis	–	Field / lab/ modelling	Lab / field / spatial analysis	Lab / field / spatial analysis
Coastline	Field, modelling	–	–	–	Field / lab / modelling	Field / lab/ modelling	Field / lab / modelling
Water column	Lab / modelling	Lab / modelling	Lab / modelling	–	Field / lab / modelling	–	Lagrangian modelling, field tracking experiments

Note: Dashes indicate a lack of direct interaction between compartments (for example, movement takes place through an intervening reservoir.

6.1.4 Identifying key fluxes

Four fluxes are considered to be of highest priority if we are to increase our understanding in this area:

- between the ocean (surface, water column or floor) and biota
- between the ocean and the coast
- from biota to the ocean
- at the coast-to-ocean interface.

The flux between the ocean and the coast is considered most important, as most plastics must pass through the nearshore environment to reach the ocean. This is also a zone of high biodiversity, where much of the biological impact is likely to occur.

This does not rule out the importance of ocean-to-ocean movement between reservoirs or movement between the surface and water column, but it highlights the critical need for a better understanding of movement between key reservoirs. Fluxes between ice and other reservoirs are considered to be of less importance, although there is agreement that modelling fluxes between ice and other reservoirs might not be particularly difficult.

It is widely believed that information can be gathered to evaluate fluxes between the ocean surface and water column, surface-to-coastline fluxes and litter in coastal reservoirs. In contrast, due to a lack of data, fluxes from biota to the water column (and other reservoirs) would be difficult to constrain, as would be movement from the deep ocean. One of the main challenges is the disparity between what are recognised as the most important fluxes to understand, and our current knowledge not only of fluxes, but of the plastic in those key reservoirs.

Both for mass balance modelling and to evaluate impacts, an understanding of the accumulation of plastic in biota is needed. Importantly, this is a 'sink' for which empirical data *can* be collected—whether through necropsies of deceased animals, through excreta, or with non-invasive sampling techniques. The number of papers reporting on the interactions between plastics and marine fauna has grown (see Gall & Thompson 2015), and the ingestion of debris, entanglement and chemical contamination are increasingly reported in the literature. It might now be reasonable to estimate microplastics in biota, but so far an estimate of the overall mass of debris in wildlife has yet to be made.

6.2 Models of surface-to-surface fluxes

The most well developed research on the flux of plastics in marine systems is on the flux between coastal regions and the surface layer of the ocean. This is partly due to the availability of models for representing the dynamics in this portion of the system, the availability of data and the relative ease of study. In the sections below, we focus on this portion of the system and dynamics.

6.2.1 Published models

A number of models have been applied to predict the distribution of plastic debris in the ocean. The initial modelling effort by Maximenko *et al.* (2012) focused primarily on describing the distribution of plastics, using a particle tracking model that could represent the effects of surface currents. The Maximenko model applies a transition matrix approach, which is based on the probability of particle travel between ½° bins. The bins are calculated from historical trajectories of a global set of satellite-tracked drifting buoys (NOAA 2016). In this model, microplastics are represented as a virtual tracer and are advected through the ocean by iterating the transition matrix for 10 years. The Maximenko model employs a uniform distribution over the global ocean as the source function. Results from the model showed that a high concentration of microplastics builds up in the five subtropical gyres in 2–3 years. In this approach, microplastics have the potential to persist for hundreds of years before washing ashore.

The Lebreton model (Lebreton *et al.* 2012) uses ocean velocity fields from the 1/12° global HYCOM circulation model.[9] Virtual microplastics are sourced from major river mouths in proportion to

[9] HYCOM, https://hycom.org.

urban development within individual watersheds. This model considers coastal input as a function of coastal population. The Lebreton model releases microplastics continuously, in increasing amounts, based on the global plastic production data (*sensu* Plastinum 2009). Particles are advected by the ocean surface velocity field for 30 years.

The van Sebille model (van Sebille *et al.* 2012; van Sebille 2014) advects microplastics in ocean currents captured in a transition matrix built from the trajectories of drifting buoys, in a manner consistent with the Maximenko model. Here, the source function is assumed to be proportional to the human population within 200 km of the coast, scaled to the amount of plastic waste available to enter the ocean, on a country-by-country basis in 2010 (based on Jambeck *et al.* 2015). The van Sebille model continuously releases microplastics at each coastal location for 50 years (1964–2014), increasing in time based on global plastic production data (Plastics Europe 2013).

The three ocean circulation models treat microplastics sinks differently. The Lebreton and van Sebille models allow for *no* sinks (all released particles remain in the ocean indefinitely). In contrast, microplastics in the Maximenko model can 'wash ashore' or beach when they enter grid cells with a shoreline. None of the three models allows for loss from the surface due to ingestion, biofouling or sinking. Furthermore, none of the three incorporates fragmentation. Hence, they treat particle count concentrations similarly to mass concentrations.

6.2.2 Integration with observational data and comparisons among models

One of the major applications of the models described in Section 6.2.1 is in providing a surface that can be used to interpolate the global distribution of plastics, given limited at-sea observations. Three research teams have taken this approach. Erikssen and colleagues (2014) used the model developed by Lebreton *et al.* (2012) to estimate that there are more than 5 trillion plastic pieces in the ocean (or 66,000 tonnes); Cozar and co-authors (2014) used a simplified surface derived from one of the models to estimate that between 7,000 and 35,000 tonnes of plastic occurs in the open ocean (2014). However, one major uncertainty is the mismatch between the amount of plastic these models predict to be in the ocean and the estimates of annual input. Jambeck *et al.* (2015) estimated that 6–12 million tonnes of plastic enters the ocean each year. The differences in approaches, reporting methods and data collection methods can make it challenging to understand the discrepancies in reports.

Recently, van Sebille and colleagues (2015), compared estimates of microplastic abundance and mass using a rigorous statistical framework. They standardised a large global dataset of plastic marine litter based on surface trawl surveys (of more than 11,000 samples). They also compared the three ocean circulation modelling approaches of Lebreton *et al.* (2012), Maximenko *et al.* (2012) and van Sebille (2014), using each to estimate the global standing stock of small floating plastic litter. Importantly, they resolved sampling biases and other variations by applying a statistical model to standardise the dataset to appropriately scale the three model solutions. They compared where the models converge and identified regions where discrepancies between the modelling approaches need to be resolved.

The resulting estimates of plastic litter are roughly similar, which is very encouraging, given the methodological differences between the modelling approaches. However, the model solutions do vary: the estimates of small floating plastic in the ocean range from 93,000 to 236,000 tonnes, depending on the model used (van Sebille *et al.* 2015). The variations in model solutions emphasise the undersampling that persists in oceanic sampling of floating plastic, particularly in the Southern Hemisphere. The least well sampled regions are those with low plastic concentrations—where models predict that anywhere from 30% to 70% of plastic particles may occur. Quantifying the densities in these regions will be critical for understanding the global load of plastic, and potentially for resolving the mismatch between the estimated annual input of 6–12 million tonnes and the standing stock of only 236,000 tonnes.

6.3 Critical assumptions

Modelling marine debris necessarily involves some assumptions. Five sets of assumptions are critical:

- inputs to the ocean
- transport only by surface currents
- particle traits
- feedback from particles to currents
- losses of particles from the system.

6.3.1 Inputs to the ocean

The representation of the sources of plastic inputs to the ocean has become more realistic with time. Early models, such as those of Maximenko *et al.* (2012), ignored the source dynamics altogether, starting with plastic particles uniformly distributed throughout the ocean. This is a reasonable starting point if one assumes that the system is largely in equilibrium. In that case, the starting distribution of plastics is fairly irrelevant, as the strong dynamics of surface transport will rapidly erase the effect of the starting distribution. This model was successful in reproducing the general patterns observed, with the highest frequencies of plastic particles concentrated in the oceanic convergence zones.

Subsequent modelling efforts, including Lebreton *et al.* (2012) and van Sebille *et al.* (2012), have used source functions that attempt to integrate information about land-based sources into the analysis. This is a critical improvement, as recent research has shown that there is significant variation in the expected inputs to the ocean (Jambeck *et al.* 2015). In particular, countries with large coastal populations, increasing incomes and relatively underdeveloped waste management practices and infrastructure are predicted to make disproportionately large contributions to the plastic input to the marine system.

Given that the predicted input to the ocean of 6–12 million tonnes per year is substantially larger than the predicted standing stock in the ocean of 236,000 tonnes, it is very likely that the underlying dynamics of marine debris are very far from the equilibrium assumptions initially used in modelling the system. This non-equilibrium suggests that both source dynamics and the possible sinks for inputs of plastic from land will be critical considerations in developing accurate models for the dynamics of marine debris. Ultimately, the mismatch between the estimates of the standing stock and the annual inputs suggests that the ocean may be a transitory location for plastic, not a storage location.

The most recent modelling analysis of plastic distribution, by van Sebille *et al.* (2015), uses the plastic losses to the ocean from Jambeck *et al.* (2015) to scale the estimated inputs by country in simulating the distribution of plastic. This scaling accounts for the differential inputs by country; however, this approach does not use the input volumes from Jambeck *et al.* (2015), but instead scales the inputs in the same relative way across countries. The resulting predicted distribution of plastics in the ocean is then scaled using a dataset of more than 11,000 at-sea observations that has been standardised statistically to remove the effects of sampling conditions such as wind, sampling year and other factors and to address sampling variation across sites. This rescaled surface is then used to estimate the distribution of plastics at the global scale and, by integrating across it, the standing stock of plastics in the oceans.

6.3.2 Transport only by surface currents

The Maximenko *et al.* (2012), Lebreton *et al.* (2012) and van Sebille *et al.* (2012) models all use surface currents inferred from a variety of sources to simulate the transport of plastic particles in the ocean surface layer. These models, and the more recent expansions, have generally ignored wind forcing on floating plastics, aside from that encapsulated in the surface currents. They have also ignored wave transport, or Stokes drift. Wave transport could be relatively important, particularly near coastal regions where it is consistently towards shore and thus transports material out of the marine environment. The models have also ignored direct wind transport, which is potentially relevant for items that are larger and more buoyant. During high winds, those items may be lofted by waves and subsequently transported in the air, largely losing contact with the water. This effect would presumably be biased towards larger and more buoyant items, due to both their chance of leaving the water surface and the cross-section exposed to the wind.

There is some evidence to suggest that wind and Stokes drift are important, particularly for larger particles. Surveys of marine debris off Africa and Japan suggest that the size distribution of items decreases in samples further from shore (Isobe *et al.* 2014; P. Ryan, pers. comm.). In some cases, this has been attributed to biofouling, which shows some signs of being more important on larger items (P. Ryan, pers. comm.). However, modelling and analysis in the Japanese case both suggest that this is probably due to increased shoreward transport of larger items, fragmentation due to wave, wind, and exposure to sunlight in shallow coastal regions, and subsequent oceanward transport of larger items (Isobe *et al.* 2014).

Note that most analysis to date has focused on floating plastics. However, around two-thirds of plastic produced is negatively buoyant. This material is likely to follow very different trajectories, either remaining concentrated around sources such as stormwater and sewer outfalls, coastal sites, and areas of high vessel traffic or moving offshore driven by currents and wind. There is some possibility of transport by currents and in particular by tides in coastal margins; however, there has been little investigation of negatively buoyant material to date, and even less work on its transport and fate.

6.3.3 Particle traits

Model tracking of Lagrangian particles requires decisions about particles' traits, including their buoyancy (density), windage, size and shape and the extent to which wave action (Stokes drift) affects them.

So far, there has not been a comprehensive modelling study on the sensitivity of particle pathways to these traits. It is therefore unclear which of the traits are the most important to incorporate and which may be less important to consider. While the answer to the trait importance ranking will depend on the question or process being investigated, it would be insightful to perform such comprehensive modelling studies for a few selected tracking scenarios. Those studies could include different scales (local to global, weeks to decades) and source functions.

A comprehensive set of sensitivity studies can help guide research priorities into parameterisations of particle traits, similarly to approaches used to evaluate the historical movement of plankton. Van Sebille *et al.* (2015) quantified the lateral distance that planktonic species can move, incorporating regional variability based on surface currents and variations in surface current movements. They further included life history traits of the target species, such as depth, sinking speed and lifespan. Such approaches can also be applied to particle tracking of plastics, incorporating plastic traits, rates of degradation or breakdown, the likelihood of ingestion and other 'behaviours'.

6.3.4 Lack of feedback from particles

Almost all plastic particle modelling to date assumes that particles carried by ocean currents do not in turn feed back on those currents. It is this assumption that allows for offline particle tracking. However, the validity of the assumption is not entirely clear. There are myriad factors that influence particle movement. The behaviours of microplastics in the ocean have recently been reviewed, focusing on physical, chemical and 'bio' behaviours of plastic (Wang *et al.* 2016).

There is some literature suggesting that plankton has an effect on ocean circulation through shading. As sunlight penetrates the upper few metres of the ocean, the depth at which it is absorbed affects the stability of the water column. If there is suspended matter in the upper ocean (plankton, and possibly also plastics), that changes the penetration depth and thereby the stratification of the upper ocean.

It is as yet completely unclear whether shading by plastics has a discernible effect on the ocean. However, if it turns out that plastic does shade sunlight on scales that matter, that would mean that particle models might need to be run online so that they can feed back on the hydrodynamic model itself. Whether these 'active particles' are needed should be further investigated.

6.3.5 Losses of particles from the system

None of the global scale models published to date (Maximenko *et al.* 2012; Lebreton *et al.* 2012; van Sebille *et al.* 2012; van Sebille *et al.* 2015) represent the loss of particles from the system in any detail. In fact, most do not incorporate loss of particles at all (Lebreton, van Sebille 2012, van Sebille 2015).

Given the significant mismatch between the estimated standing stock and the estimated annual input of marine plastic, Cozar *et al.* (2014) attempted to estimate losses from the system and the sinks to which they are moving. They concluded that the most likely sink is settling to the benthos due to biofouling. However, the patterns of benthic plastic distribution appear to reflect local deposition around sources, such as areas of fishing, coastal urban regions and river outlets (Corcoran 2015).

There is some evidence that coastal regions may be a major sink, particularly given the shoreward bias of wind transport (Isobe *et al.* 2014; Kako *et al.* 2014). Experimental studies suggest similar results, including substantial local retention near coastal sources (Carson *et al.* 2013).

There has also been some suggestion that the biota may be a sink for debris (Cozar *et al.* 2014). A wide range of studies suggest significant levels of debris ingestion by hundreds of species (Gall & Thompson 2015), from zooplankton (Desforges *et al.* 2015) to whales (Fossi *et al.* 2012), but it is not clear whether this biological reservoir is significant in terms of the unaccounted for losses in the system.

6.4 Uncertainties in modelling and data

Empirical measurements of floating debris at sea have been used to infer the distribution of plastic at the global scale, throughout the world's oceans (Lebreton *et al.* 2012; Maximenko *et al.* 2012; van Sebille *et al.* 2015; Cozar *et al*. 2014). Predicted distributions have then been used to estimate the exposure of wildlife to plastic pollution (Wilcox *et al.* 2015; Schuyler *et al.* 2015).

Yet, despite the significant inferences being made from the at-sea sampling data and the models used to extrapolate global densities from them, there has been very little examination of the uncertainty underlying the data, models or resulting predictions. A notable attempt to include uncertainty is the recent global estimate of debris by Cozar *et al.* (2014). The researchers aggregated patterns of predicted plastic density into high-, intermediate-, and low-density regions, then used mean values for those regions based on samples to infer the average across the region. The resulting density estimates were then integrated across the entire area for each density level to produce a total debris estimate at the global scale. Cozar and colleagues did attempt to bring error estimates through in their analysis, but the estimates were primarily derived from the estimates of the means of the 1,127 net tows they averaged for each of 442 spatial blocks, which were then averaged to produce debris estimates in 15 global-scale polygons (3 accumulation levels by 5 ocean basins). This ignores a range of sources of error, from sampling error in their at-sea trawl data to spatial errors in their model-derived accumulation zone boundaries.

A recent analysis has compared predictions made by three of the models available for interpolating the debris surface at the global scale based on at-sea sampling data (van Sebille *et al.* 2015). That analysis found the models in general concordance for major zones of debris concentration in the open oceans. However, there are substantial differences in closed basins, such as the Mediterranean, and in coastal zones. Both of these differences appear to be due to structural differences in the models, such as the inclusion of debris source dynamics and the spatial resolution of the underlying models (van Sebille *et al.* 2015). The analysis also attempted to incorporate sampling error explicitly, using a statistical model to correct the 13,000 trawl samples for the effects of wind, time and other variables that affect either debris density or the efficiency of sampling. Estimates of sampling error at the individual trawl level were then used to put uncertainty estimates on the projections emerging from the models. Nevertheless, this analysis still ignores a number of important sources of uncertainty in the analysis.

Although researchers have not propagated uncertainty through their analyses as they have made global projections of the debris fields, there is good evidence that both uncertainties in the models and sampling error in the data could have significant effects on the projections. For example, Reisser *et al.* (2015) examined the role of wind- and wave-driven mixing in reducing plastic debris at the ocean surface, finding that up to 70% of plastic can be below the surface and unavailable to typical surface trawls. Moreover, this downward mixing varied with the shape and density of the items, resulting in potentially complex biases in surface samples. Reisser *et al.* (2015) also noted cases in which the observed mixing did not match theoretical models that have been used to correct surface trawl data for mixing effects (Kukulka *et al.* 2012). The models underlying the global debris surfaces also have errors associated with them. In some cases, those models are based on drifter trajectories (for example, Maximenko *et al.* 2012), which can contain errors due to aspects of the conversion of drifter trajectories to drift rates driving the modelled surfaces (for example, Katsumata & Yoshinari 2010). These empirical models also contain uncertainty due to sampling error, driven by the number of drifters available to estimate transitions between locations and the coverage of the underlying variability in the drift trajectories due to changing conditions. For example, seasonal changes such as the strengthening of the trade winds increase the sampling needed to accurately estimate surface velocities in affected regions.

We explored three aspects of the gaps in our current knowledge of the distribution of plastic pollution in marine systems. First, we examined which zones of the ocean have been well sampled and where sampling is either absent or sparse. Second, we used a spatial statistical model to estimate the distribution of debris (taking sampling biases into account) and its variability to identify locations with sampling but at which there remains significant uncertainty. Third, we compared these standardised data to the distribution of debris predicted by an oceanographic model, identifying areas where there was a lack of concordance between the predicted and observed densities. Together, these results provide a picture of where additional sampling or further analytical effort should be allocated.

6.4.1 Methods

Data assembly

We used data assembled for a recent global estimate of the global standing stock of debris, described in van Sebille *et al.* (2015). This data included 11,854 surface trawls from 27 studies carried out between 1971 and 2013 in all major oceans except the Arctic. Samples were collected using plankton nets varying in mesh size from 0.15 mm to 3.0 mm, although more than 90% of the observations were collected with manta or neuston nets with mesh sizes between 0.333 mm and 0.335 mm. All data were converted to counts per volume of water sampled for analysis. There is an established effect of wind-driven mixing on debris at the ocean surface, but many studies did not report wind speeds during sampling. We used daily average wind speeds from the European Centre for Medium-Range Weather Forecasts ERA-Interim global atmospheric reanalysis (Dee *et al.* 2011) to interpolate the wind velocity for each trawl date – location combination. ERA-Interim reanalysis data are available from 1 January 1979, so 222 trawls before that date were excluded from the analysis. For further details on the dataset, including a list of data sources, see van Sebille *et al.* (2015).

Quantifying observational variability

We used a generalised additive model implemented in the mgcv package of the R statistical language to model the observational data on plastic density in the ocean (Wood 2008; R core team 2014). We investigated a number of possible variables that could account for variation in the data, including the year of the survey, wind speed, trawl length, and study. We used a smooth term to initially explore the relationship between the continuous predictor variables to evaluate the potential for non-linearity. We subsequently fitted both second and first order polynomials for terms, as appropriate, based on the smooth term.

All models we evaluated included a spherical smooth to represent position on the globe. This smooth forces values near to each other to have some relationship, and assists with estimations of the relationships of the variables driving sampling error, such as wind velocity. We also evaluated the importance of allowing a discontinuity at the Americas, at the boundary between the Caribbean Sea and the tropical Pacific Ocean, to account for the lack of connection, given the proximity of the basins. We fitted each of the possible models incorporating the potential covariates and identified the best fitting model based on the Akaike information criterion (Burnham & Anderson 2002). We checked the best fitting model for overdispersion and tested the fit of the final model using a goodness-of-fit test on the deviance residuals.

We used the residual deviation between the at-sea samples and fitted values from our best model to evaluate the accuracy and bias in estimates of the debris density at the global scale. The residuals represent the remaining variation around the mean value predicted by the model for each observation. We fitted a spatial model to the residuals to look for spatial patterns among them. We used the absolute value of the residuals to measure the unexplained variability in the observational data and the signed value of the residual to measure bias. Regions with strong bias have strongly negative or positive residual values.

Identifying deviations from expectations

We used a predicted distribution of plastic density in the ocean, based on methods described in van Sebille *et al.* (2015). The model we used was the one adapted from van Sebille *et al.* (2012). The method assumes that plastic is lost from the coastline in proportion to the population within 200 km of the coast, scaled by the amount of plastic estimated to enter the ocean, by country, from Jambeck *et al.* (2015). Simulated plastic was released from the coastline on a monthly basis, starting from the year 1950. The volume of plastic released from each coastal location increased on an annual basis, in proportion to the increase in global production as estimated by Plastics Europe (2013). Drifting trajectories of this plastic were then modelled using a statistical model estimated from the global drifter dataset, on a bimonthly basis. Modelled distributions were interpolated to a 1° latitude by 1° longitude grid.

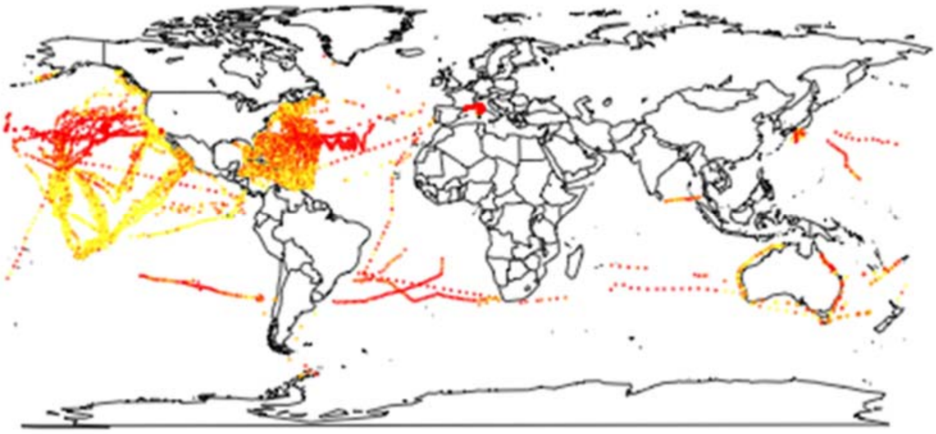
We fitted this predicted distribution to the standardised global dataset of plastic observations using a linear regression of the predicted densities on the observations. We then evaluated the patterns of mismatch based on the residuals of the linear regression. We evaluated the residuals for both a global model and for a model allowing separate regression coefficients for each ocean basin. As for the standardisation model, we used a spatial model of the absolute value and the signed value of the residuals to investigate patterns of precision and bias.

6.4.2 Results

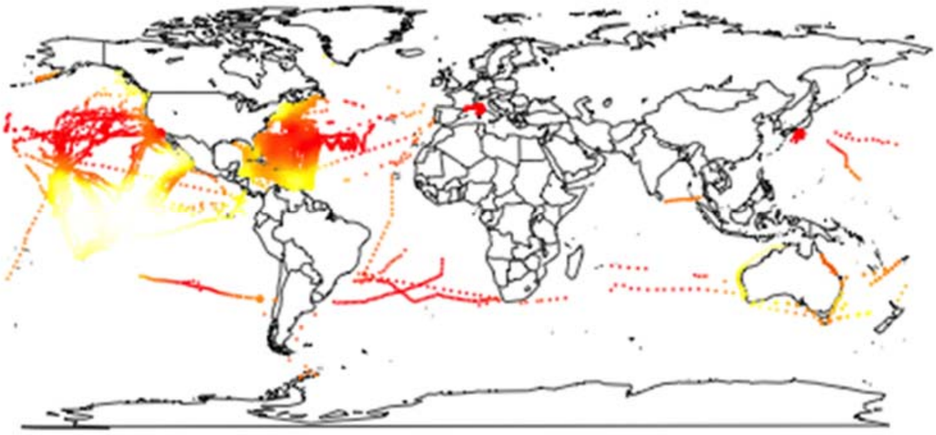
Data coverage is by far the most extensive in the western North Atlantic Ocean and the eastern tropical and North Pacific oceans (Figure 15a). There is relatively less coverage in the eastern and southern Atlantic. The southern and western portions of the Pacific and the Indian Ocean have the least coverage. Neither the Arctic nor the Antarctic has significant sampling at this point.

Figure 15. Observed, standardised and residual values for the global plastic observations

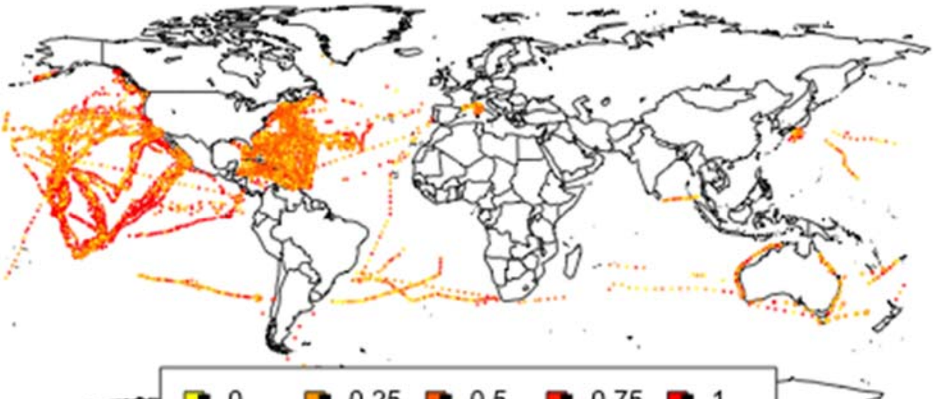
A



B



C



Panel A Relative density of plastic debris in surface trawl data around the globe. Data are presented in rank order of density, instead of raw densities, due to the long-tailed distribution of density values.

Panel B Rank-ordered fitted values for the density of plastics in samples from the best fit model for the observational data.

Panel C Rank-ordered residuals between the best model and the observed data.

Sampling is relatively poor in coastal regions and is extensive only along the central coasts of North America, off Japan, in the Mediterranean, and around the Australian continent (Figure 15a). Of the five major accumulation zones in the world's oceans, only the North Atlantic and North Pacific zones have been extensively sampled. Even in those two cases, intensive sampling covers only a portion of the expected area of high plastic densities.

The best fitting statistical model for the observational data included terms for wind speed and study year and a nonlinear term at the boundary dividing the Pacific and Caribbean basins at the Americas (Table 4). The wind speed in the model was represented as a second order polynomial, with a decreasing incremental effect of wind speed at higher velocities. There was a significant positive and linear trend with time. Allowing for a discontinuity at the Caribbean Sea – Pacific Ocean boundary improved the overall model fit, although it was not significant in its own right as a term in the model. Overall, the intercept term had a larger effect than any other term at the median of the covariate value. Following the intercept, wind speed was next most important in determining the plastic density observed during sampling, followed by study year.

Table 4. Adequacy of the candidate standardisation models and coefficients of the best fitting model

A. Model fit		**B. Best fit model coefficients[b]**				**Median effect**
Model[a]	**AIC**	**Coefficient**	**Estimate**	**Std. error**	**p value**	
SAyWWsqBd2	159533.3	Intercept	7.3	3.4	0.033	7.3
SAyWWsqBd	159537.7	Year (since 1950)	0.016	0.005	0.0012	0.86
SAyWWsq	159538.2	Wind speed	–0.34	0.045	1.40E-13	–1.8
SAyWBd	159541.9	Wind speed squared	0.011	0.0044	0.015	0.32
SAyWBd2	159541.9	Atlantic–Pacific boundary squared	3.7	8.4	0.67	1.4
SAyW	159542.4					
SWWsq	159592.8					
SW	159598.4					
SWsq	159727.7					
S	160546.3					
0	177503.4					

AIC = Akaike information criterion.

a Model codes in panel A are 0—intercept only, S—spherical smooth, W—wind speed, Wsq—wind speed squared, Bd—Atlantic–Pacific discontinuity, Bd2—Atlantic–Pacific discontinuity squared, Ay—year (since 1950).

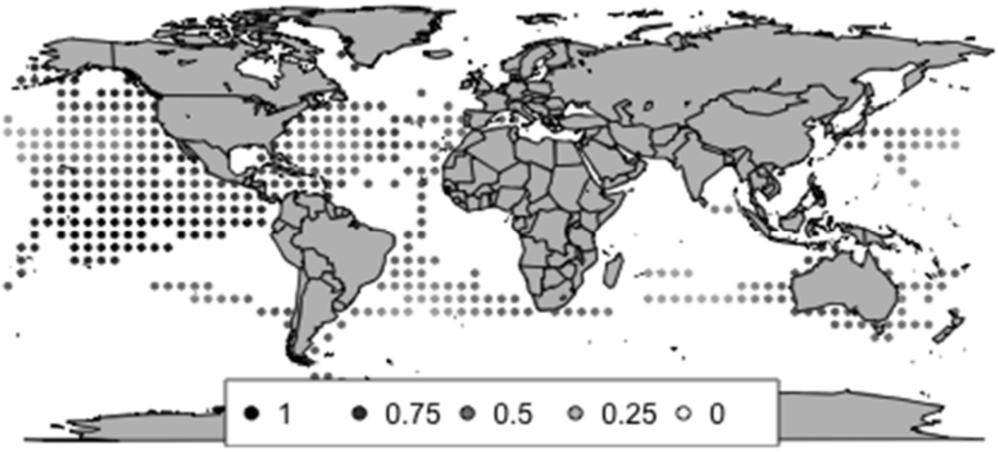
b The median effect column in panel B is produced by multiplying the coefficient estimate by the median value of the corresponding covariate, and gives a measure of the relative magnitude of the effect of each term in the model.

While the global model of debris distribution does in general match the patterns in the observational data, with most of the major oceanic accumulation zones modelled, there are a number of locations where the fitted and observed values differ, particularly at the coastal margins, such as near Japan, in the California Bight (near Los Angeles) and along the north-eastern and north-western coasts of Australia. In the highly populated regions, such as the California and Japanese coasts, the model suggests that the observations should be higher than they are, once standardised (Figure 15a&b). By contrast, the coasts of Australia are predicted to have lower debris densities relative to other samples after standardisation. For patterns of fit, the model of the observations appears to fit relatively well to the sample dataset around the globe, with the exception of samples in the central tropical Pacific and in the central North Atlantic at the edge of the main region of sampling effort (Figure 15c).

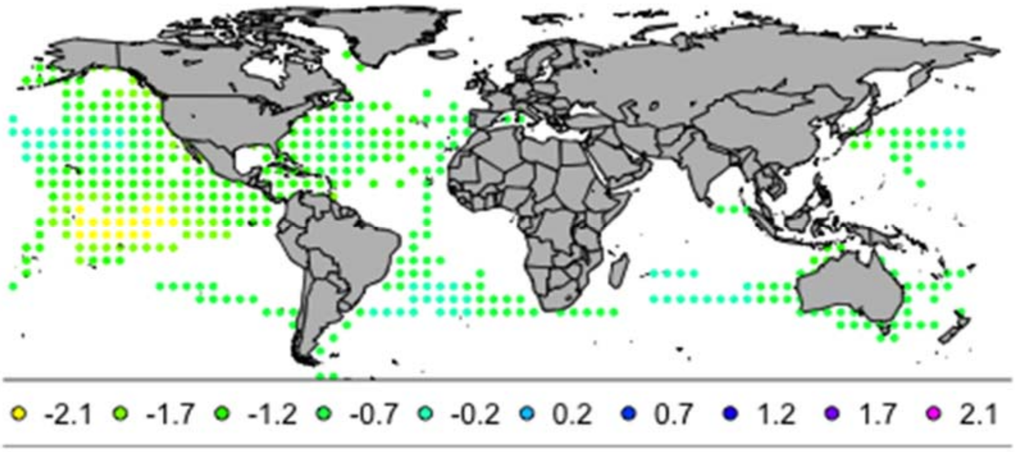
Looking at the spatial patterns in these residuals as a measure of variation in the data, after the sampling effects from wind and other sources have been controlled for, we see elevated variation in the observations in the central tropical Pacific (Figure 16a). There is also some evidence of high variability in the observations in the North Pacific, along the coastal margins of Alaska and Canada (Figure 16a). For bias in the residuals, there appears to be a negative bias (that is, the predicted values in the standardisation model are greater than the observed values from the sample) in the central tropical Pacific (Figure 16b). The bias analysis suggests that there might be a slight negative bias elsewhere, particularly along the coastal regions. There was no bias in areas of high debris densities, such as the North Pacific gyre.

Figure 16. Accuracy and bias in the statistical model used to correct the global debris data

A



B



Panel A The magnitude of the residuals. Values are estimated from a global surface fitted to the residuals from the model of debris density in the at-sea surveys. Points are displayed for locations with empirical measurements within 5° latitude or longitude. The scale bar shows the relative magnitude of the absolute residual values, scaled from 0 to 1.

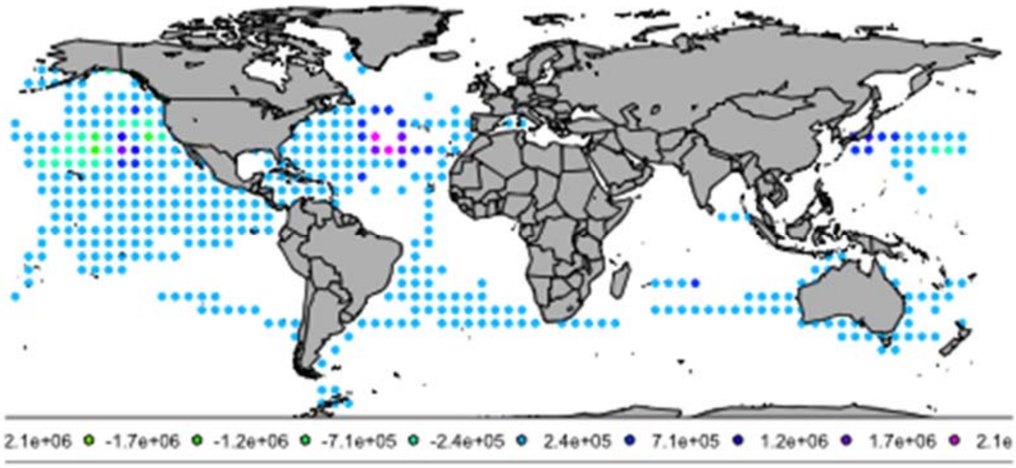
Panel B The bias in the residuals from the statistical model fitted to the global debris surveys. Although the model of the residuals is global, only data points within 5° of a survey location are displayed.

Evaluating the fit of the predicted debris distribution from the oceanographic model to the standardised at-sea survey data, one can see areas of high variance and bias, particularly concentrated in the central and western North Pacific (Figure 17). Most regions of the globe appear reasonably approximated by the predicted debris distribution from the oceanographic model. However, the two regions of poor fit appear to have both significant unexplained variation (Figure 17a) and a bias towards underestimating the observed values (Figure 17b).

Figure 17. Accuracy and bias in the fit of the oceanographic model used to infer the global distribution of plastic densities

A

B



Panel A The magnitude of the residuals. Values are estimated from a global surface fitted to the residuals resulting from the fit of the oceanographic model of debris density to the standardised data. Points are displayed for locations with standardised observations within 5° latitude or longitude. The scale bar shows the relative magnitude of the absolute residual values, scaled from 0 to 1.

Panel B The bias in the residuals from the statistical model fitted to the global debris surveys. Although the model of the residuals is global, only data points within 5° of a survey location are displayed.

6.4.3 Discussion

There are clear limitations to our current ability to accurately predict the distribution of marine debris at the global scale. Data coverage is limited, with only the western North Atlantic and the eastern Pacific having substantial sampling. These data are highly variable due to sampling conditions such as wind mixing and due to changes in the availability of plastic in the ocean. Even after correcting for sampling conditions and changes in debris densities, some areas of the ocean, such as the western tropical Pacific, retain significant amounts of unexplained variation, with some bias in the corrected values. Finally, when an oceanographic model is used to interpolate the corrected observations to make global estimates of the distribution of debris or its total amount, there are areas of poor underestimation in the centres of the gyres.

These uncertainties are driven to some extent by the underlying character of the data. The data was collected across 27 different studies, potentially involving multiple vessels, over more than a 40-year period. In our analysis, we noted a number of possible drivers of variation in the sampling process. First, there appears to be a difference between day and night samples, which we detected in the subset of data with time. Vertical migration of the planktonic community could lead to variation in sampling efficiency, due to additional material in the nets used for collecting debris. We explored the relationship with daylight, proximity to dawn and dusk and a number of other hypotheses. However, we were not able to detect a strong enough signal to establish whether vertical migration might be causing this effect. We also noted a strong effect of the 27 studies on the debris sampled. However, this effect is confounded with the distribution of debris, as the studies did not generally overlap with each other in space and time. Thus, there is an inherent trade-off in examining the differences among vessels and estimating the spatial distribution of the debris from the samples. Clearly, a range of factors related to the studies could affect their detection rates, including the configuration of the net deployment (tow angle, tow height, vessel wake proximity and so on), sample processing facilities and methods, and competing work demands for the research team, to name a few.

One clear source of uncertainty in the data arises due to the sampling effort in the data collection process. The mean area trawled across the 11,854 samples was 1,769 square metres, suggesting an average lower detection limit of 565 items/km^2. Examining the raw data, there do appear to be missing observations between 0 and 500 items per square kilometre. There were 4,247 trawls with debris densities of 0 and 6,829 trawls with densities greater than 500 pieces/km^2, but only 241 surveys with densities in the interval 0 to 500. Given that densities are continuously distributed and otherwise right skewed (smaller values are more frequent), this suggests that a large portion of low-density locations are recorded as zeros.

This is known as right censored data in the statistical literature. Censored data, including right censored data, can be used in statistical modelling, but requires specialised approaches to account for the uncertainty in the censored region. These methods could potentially improve the estimates we present; however, many of the statistical tools we used, in particular spherical smooths and the Tweedie error structure, are not readily available together with tools for censored data. Censoring is one likely explanation for the area of poor fit observed in the standardisation model in the tropical Pacific. This area is predicted to have low densities, and there are many zero observations but also some non-zero ones, making it difficult for the statistical model to fit this bimodal distribution.

Local heterogeneity at fine scales may also be a significant factor in generating additional variation in the data. Marine debris accumulates in surface circulation patterns, creating narrow linear features with significantly elevated levels of debris in them. Sample estimates of densities in this environment will vary widely, depending on the angle of sampling relative to the feature. If a trawl is parallel to the direction of a surface feature, it will either underestimate or overestimate the density, depending on whether it intersects the feature or not.

We found that the global surface predicted for marine debris, based on coastal inputs, provided a reasonable fit, with the exception of the high-concentration zones at the eastern edge of the North Atlantic gyre, in the North Pacific gyre and off the coast of Japan. Recent analysis by van Sebille *et al.* (2015) also noted this pattern, which suggests that there is an issue with the underlying dynamics used to generate the debris surface. Researchers have divided their models into basins to allow more flexibility in fitting, using this as a method to address this problem (for example, Cozar *et al.* 2014; van Sebille *et al.* 2015). However, this is only an approximation, as the lack of fit implies either a mis-specified error term or incorrect dynamics in the underlying physical model.

We also found that the regression fit showed evidence of heteroskedasticity, in this case larger variance in the observation data in areas with high density. This is presumably due to surface circulation concentrating debris at small scales, which would make sampling increasingly variable as debris concentrations increase.

A number of practical steps could be taken to improve both the underlying data and the modelling approaches used to predict the distribution and abundance of plastic drifting in the sea. Future at-sea sampling could be improved by using replicate trawls at each sampling station, allowing variation within sampling stations to be estimated. This would assist in removing effects of small-scale spatial heterogeneity in the samples. Sampling strategies could also be structured in an adaptive manner, with increasing effort in low-density areas to reduce issues with censoring. Future analysis of debris data should address censoring, using appropriate statistical modelling of the data in the standardisation process. Consideration should also be given to the error distribution used in fitting predicted debris distributions to empirical data. The increase in the variance in samples in areas with high average values is typical of counting type processes, where variance frequently increases with the mean of the distribution (Hilborn & Mangel 1993). This could be addressed in future work by using an error distribution in which the variance scales with the mean for the fitting, or explicitly modelling the variance in addition to the mean.

6.4.4 Model uncertainties associated with litter deposition

Identifying regional hotspots for litter deposition using a dispersal model requires accurate environmental forcing components. The forcing terms used in the framework presented above are derived from archive products of numerically modelled sea-surface current, wave and wind data. Some uncertainties are associated with these theoretical models. In particular, understanding the issue from a global to a regional or national scale requires different tools, models and assumptions. Timescale dependency is also an important factor. For example, a global geostrophic current model with timescales of 3–24 hours, as used in this example, does not consider tidal circulation.

Coastal circulation processes are rather complex and site specific. They are best understood at smaller timescales (one hour or less) and spatial scales (metres to kilometres). Therefore, while global models are useful to draw probabilistic conclusions, regional models at higher resolution (in both time and space) should be used while conducting a deterministic analysis. There are many existing options for environmental forcing data and circulation models at global and regional scales. In any case, the accuracy of dispersal forcing components should be validated against observations.

The main source of validation for these circulation models is location data collected by drifting buoys, such as in the Global Drifter program. While drifting buoys provide very useful information, other technologies to better understand the dispersal of debris are available (high-resolution imagery, synthetic aperture radar, LiDAR, Doppler scattering).

Further research and investment in monitoring debris displacement should be conducted in targeted regions. The type of polymer and the size and shape of debris have an impact on its floatability. High-floatability items, such as packaging products, bottles or fishing buoys, will be much more subject to wind forcing than less buoyant items, such as fishing nets or plastic bags.

The dispersal model shows that the windage coefficient (set at a standard 0.5% for this assessment) has a very significant impact over time on debris mass transport and connectivity between the different accumulation zones worldwide. A rigorous assessment should consider different types of debris with the windage coefficient gradually varying between no forcing (0%, fully submerged) to strong wind-forcing (2–3%, semi-submerged). Ideally, the windage coefficient should vary in time to reflect the change in floatability over time (for example, from degradation or biofouling). A framework integrating this component would require empirical formulations based on observations and experiments for different types of polymer, debris shapes, water biochemistry, solar radiation and so on.

The modelling approach presented here has other limitations. For example, the model treats only the sea surface, which is considered as a sink for marine debris. However, no interaction with other sinks, such as the shoreline (stranding), the water column and sea bed (sinking) or the biota (ingestion) are considered. There is still room to improve the various models of marine litter presented to date, particularly in regard to the long-term fate of plastic material in the ocean.

Tables 5 and 6 in Appendix 1 list available datasets and the capabilities and limitations of available particle tracking models.

6.5 Expert workshops on modelling

As part of this UNEP project, we ran two expert workshops on modelling marine debris transport, reservoirs, accumulation and fluxes. The workshops were associated with the UNEP/CSIRO 'Modelling and monitoring marine litter movement, transport and accumulation' collaboration project.

At the workshops, we set out to summarise the current state of knowledge, based on the participants' expertise in oceanographic modelling, to identify key areas in need of further research.

Recommendations to overcome challenges identified in the workshops are set out in Section 8.8 of this report. Workshop participants are listed in Appendix 2.

6.5.1 April 2015 workshop

Immediately following a Joint Group of Experts on the Scientific Aspects of Marine Environment Protection (GESAMP) microplastics working group meeting hosted by the Food and Agriculture Organization in Rome, CSIRO organised a one-day modelling workshop that included some of the working group participants. The workshop took place on 24 April 2015.

The aim was to identify approaches, knowledge gaps and data needed to increase awareness on marine litter.

The group:

- reviewed the state of knowledge about marine litter
- discussed the utility of combining empirical data with modelling to identify the sources, sinks, distribution and movement of marine litter, particularly microplastics
- considered key areas where new data would be most informative.

In a vigorous discussion about the utility and appropriateness of models, participants noted that, while models will not tell us where the plastic is, inverse modelling can be used to interpolate and predict where it is going.

The participants identified some important practical and theoretical questions:

- *Fragmentation of debris:* Do the factors that affect fragmentation (wind, UV radiation and so on) affect the spatial distribution of differently sized fragments? Where do we find large versus small fragments?
- *Modelling sources, fluxes and sinks:* Can inverse modelling be used to solve the mass conservation problem identified by Thompson *et al.* (2004)? How important are physical traits, such as specific gravity? Can models accommodate 'black boxes' and gaps?
- *Modelling processes:* How well can we model in three dimensions, by including upwelling, downwelling and other important processes?
- *Hotspots and accumulation zones:* Should we prioritise hotspots (Gulf of Biscay, Caribbean etc.) over oceanic gyres?
- *Epistemology:* Is it appropriate to base assessments on grey literature or to include only peer-reviewed journal articles? What can we learn from ocean movement research into such areas as larval dispersal, iceberg movements, mercury transportation in biota and extreme events (such as tsunamis)?

Some shortcomings in current models were identified:

- Many current models, such as ADRIFT, retain all particles (that is, no loss is modelled). While it may not be difficult to take into account sinking, fragmentation and other processes, those models require data or parameterisation to make these improvements.
- There are data gaps in many models due to no or poor drifter data in some areas.
- Many current models include surface drifters only.
- Time-series resolution needs to be appropriate for the question or region being studied, particularly because of seasonal variability in litter movement and deposition.

However, models such as ADRIFT are flexible; for example, sources can be added to the model, labelled, tracked and followed.

One of the first and most significant improvements would be to add terms for losses in the environment and for suspension and resuspension rates. To establish a reasonable loss term for coastal regions, it would be critical to use data from standing stock surveys to look at coast–ocean–coast (C-O-C) suspension and resuspension. Additional information needed might include data on winds, tides, forcing, advection and solar radiation.

To improve modelling, it would be ideal to have a comprehensive list of datasets that can be used. Those datasets would be geographically dispersed, long term, and with a high frequency of data collection.

One way to address the C-O-C knowledge gap would be to develop a transfer function from the coast to the ocean and back again. Perhaps the best way to incorporate this into existing models is to find a few locations where there are long-term data on coastline litter stocks. Analysis of such an empirical dataset, coupled with relevant covariates (wind speed, direction, tides and so on), would be useful. The ideal dataset would be a long time series with frequent sampling intervals.

Specific datasets that may be useful for modelling plastic movement include:

- North Sea fisheries data (a high-quality long-term dataset includes records of birds' nests that contain fishing debris)
- data from long-term experiments on Midway and Tern islands, including around 20 years of coastal debris data from bi-weekly cleanings of sites (a time-series analysis has looked at when debris arrives on shore; extreme events appear to drive debris deposition; there are non-linear processes that result in local deposition)
- OSPAR long-term dataset
- NOAA data (long-term time series with high frequency and reasonable geographical spread)
- regional data from the North Pacific, Japan and Korea.

The participants noted some emerging issues:

- *Ageing, fragmentation and biofouling.* Some researchers (Delft, Netherlands) are running experiments on fragmentation and incorporating modelling with fragmentation and biofouling experiments.
- *Consumption of plastics by deep-sea bacteria.* A team of scientists in Brazil is running an experiment in oligotrophic environments at depths of 1,500 m and 3,000 m to determine whether, and at what rates, bacteria consume oil (in the form of plastics).
- *The importance of the nearshore zone.* Most researchers ignore the zone between shore and 25 km or 50 km offshore due to a lack of data in global models. Global models are poor at incorporating regional processes, and current regional models cannot be scaled up to the global level.
- *Vertical and temporal resolution.* This is a problem in our current movement/transport models.

The workshop also discussed opportunities to engage with citizen scientists. For example, the Sailing with a Purpose group engages with about 30 boats around the world. Sailors are taking photos of the water to aid studies of chlorophyll. A similar approach could be used to look at debris as well.

6.5.2 August–September 2015 workshop

The second modelling workshop was held at the UNESCO offices in Paris from 31 August to 3 September 2015. It began with an overview of UNEP and GESAMP activities by Heidi Savelli.

The participants, most of whom work on large-scale (global and regional) models of marine litter movement, aimed to identify gaps and key areas on which to focus future research.

They agreed that research into plastic marine litter has two ultimate goals:

- to improve our understanding of plastic budgets (where, how and why plastic enters and leaves the ocean)

- to improve our understanding of the impacts of marine debris on biodiversity, economies and the environment.

The two tasks are very different. However, an understanding of budgets and impacts informs the development of policy responses. Importantly, whereas modelling may be at the global or regional scale, waste management policy happens at small spatial scales. Striking a balance between the scale at which the research takes place and the scale at which policy decisions are made requires thinking about outcomes and impacts at both scales.

The marine litter problem is a source, pathway and sink problem. If there is a knowledge gap in any of three, models can aid in its resolution. While multiple modelling approaches can be used to solve the problem, a fundamental first step is to clearly identify the scale, focal question, key issues and aims.

Participants noted that process models could be improved, but that it is useful to consider whether improvements are worth the effort in areas where there may be insufficient or particularly noisy data. Some of the noise at large scales can be smoothed if the aim is a mass balance (whereby the noise becomes a statistical anomaly).

There was discussion on the key issues, with a focus on the following questions:

- What are the sources?
  - What is the source of the litter or microplastic?
  - Is the plastic or microplastic primary or secondary microplastic?
  - In the absence of knowledge of sources, can we model the behaviour of microplastics from coast to ocean and back to coast (C-O-C)?
  - What are the rates of inputs to the ocean (for which we need better empirical estimates)?
- How does it move?
  - How can laboratory experiments improve models of plastics in the oceans?
  - On what time/spatial scale do we need information to be able to address issues of risk or harm?
  - What improvements can be made on litter budgets and losses in the marine environment?
  - What are rates of fragmentation?
  - What are the buoyancy, sinking and refloating rates?
  - What are the priorities in understanding movement through the ocean?
- What is the fate?
  - Where are the plastic reservoirs?
  - What is the impact or harm that results?

– How can we apply the knowledge gained for policy impact?

Central to improving our understanding at all scales, and in relation to each of the priority research actions identified, is a core question: Would it be possible to have a *global, centralised data repository* where data could be made available? Such a repository could be useful not only for researchers, but for countries, governments and policymakers.

The workshops discussed recent improvements in modelling efforts, including our ability to use increasing computer power to incorporate additional parameters into marine debris modelling. Currently available approaches include circulation models, risk models and bioaccumulation models (ecosystem-scale modelling). Each has a role in increasing our knowledge and understanding of marine litter transport, and the choice to use a particular type depends upon the question asked, the region studied and the overall aim of the research.

One of the advantages of applying modelling to the marine litter issue is that it can allow us to use a variety of approaches at different scales. With models, we can focus on major drivers at a global scale that can be scaled down to consider local processes. For example, global data on wind, tides, waves, pressure and other processes can be scaled down to achieve modelled solutions at more local scales. While there may be some loss in resolution through such scaling, these approaches nevertheless improve our ability to map risks and impacts to marine biota, regions and ecosystems.

Workshop participants agreed on the following points:

- Where possible, researchers should aim to validate models with independent data in order to increase the usefulness of the research, confidence in the results and our understanding of uncertainty.
- Coastal models should ideally have very high spatial resolution (for example, 10 m in the horizontal and less than 1 m in the vertical), include parameterisations of several biogeochemical processes (such as fragmentation and beaches deposition), work on timescales that cover short-term effects (periods of a few minutes) up to seasonal and decadal variabilities, and include interactions with the atmosphere, rivers, land and deep ocean areas.
- Tracing plastics to their sources is critical but difficult, partly because of variability between and within regions, so models should ideally be tuned to consider empirical data collected in various regions (for example, incorporating country-, region- or basin-specific inputs, waste mismanagement and other covariates).
- Overlapping spatial mapping (for example, with accumulation models) with species distributions increases our ability to quantify the risk of plastics to biodiversity and marine ecosystems. Dynamic modelling of risks or impacts becomes critically important not only for individuals and populations, but also for marine species that are exposed to multiple threats to their survival and persistence. Identifying key geographical regions and taxa under higher or lower threat from marine plastics (see, for example, Wilcox *et al.* 2015; Schuyler *et al.* 2015) can provide a useful lever to drive policy.

7 Regional hotspots for litter generation, pathways and deposition

Jambeck *et al.* (2015) reported litter inputs from land to the sea based on estimates of waste mismanagement from 192 countries around the world. The top countries ranked by mass of mismanaged plastic waste were China, Indonesia, the Philippines, Vietnam and Sri Lanka, which together account more than half of the total coastal input of marine litter. In Asia and other parts of the world, emerging nations that have benefited from rapid growth in gross domestic product, improved quality of life, reduced poverty and significant increases in demand for consumer goods have not always met modern standards in waste management infrastructure and policies. However, there has not been a consistent long-term monitoring effort in those regions.

One means of identifying regional hotspots for litter generation and deposition is to develop dispersal models that simulate marine litter trajectories in the ocean. Debris is represented by particles that are continuously released from source locations. The distributions are computed using proxies such as levels of waste generation per inhabitant, as presented by Jambeck *et al.* (2015). Using data on waste management infrastructure and population density for identified regions of interest, Lagrangian particle release scenarios were created for:

- China
- Japan, North Korea, South Korea and Russia
- Southeast Asia
- Australia and the South Pacific region
- The Pacific coastline of North and Central America
- The Atlantic coastline of North and Central America (including the Caribbean)
- West Africa
- East Africa
- India, Bangladesh, Sri Lanka and the Arabian Peninsula
- Western and Northern Europe.

Figure 18 shows the mass of mismanaged waste generated per inhabitant and per day in the regions of interest. The average quantities range in average from 5 g/person/day in Europe to 92 g/person/day in China.

Figure 18. Average mismanaged waste generation per inhabitant, by region (kg/person/day)

Source: Adapted from Jambeck *et al.* (2015).

In this framework, the contribution to marine litter input of an individual region is function of its coastal population density. The population living within 50 km of the shoreline comprises more than 260 million people in China and around 400 million people in Southeast Asia. This is highly reflected in the estimate of total mismanaged waste production in those two areas compared to the rest of the regions of interest (Figure 19).

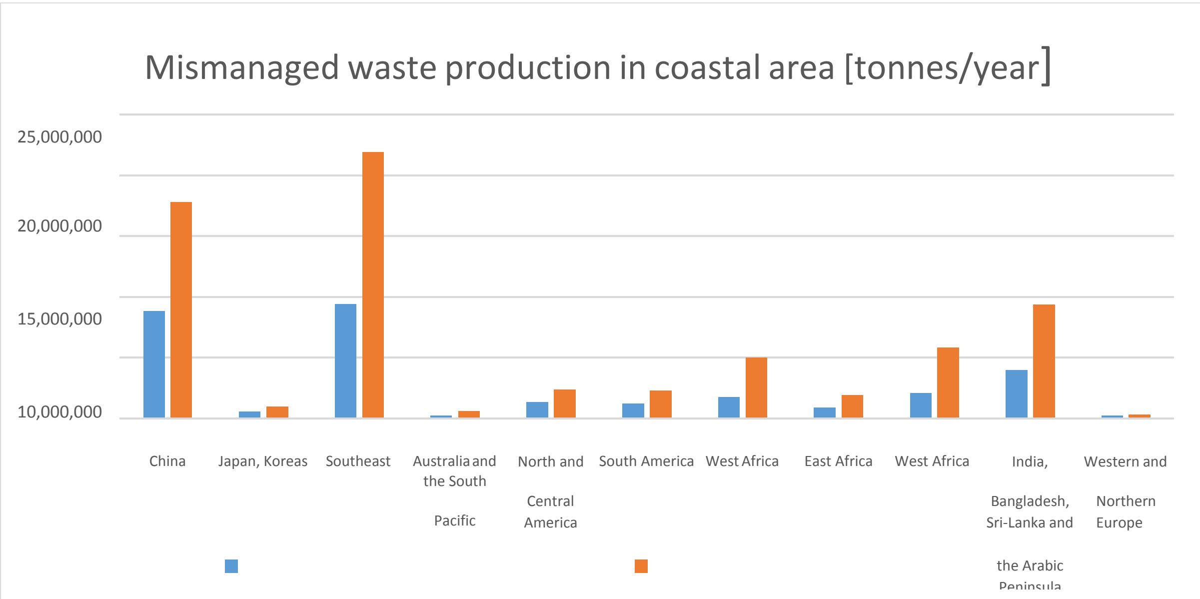


Figure 19. Total mismanaged waste generation per year, by region, 2012 and 2025 (tonnes/year)

Source: Adapted from Jambeck *et al.* (2015).

Data on population density and growth rates from IPCC scenario SRES B2 (Yetman *et al.* 2004) were extracted to produce the particle source distributions. Modelled particles were continuously released in the ocean and advected using several environmental forcing terms such as sea surface current, sea surface wind and wave-induced Stokes drift. For this simulation, wind forcing was considered equal to 0.5% of sea-surface wind speed (windage coefficient), representing debris with roughly 98% of its frontal cross-area immersed in water.

The sea-surface currents were sourced from a 2004–2014 composite database of model outputs from the data-assimilating and eddy-resolving HYCOM 1/12° reanalysis (experiment 19.0, 19.1, 90.9, 91.0 and 91.1; Cummings & Smedstad 2013; Cummings 2005; Fox *et al.* 2002) distributed by the Naval Research Laboratory of the US Navy. Wind speed and direction data were sourced from the 1948–present NCEP/NCAR global reanalysis (Kalnay *et al.* 1996) distributed by the Earth System Research Laboratory of NOAA. Finally, wave-induced Stokes drift was calculated using wave spectrum bulk coefficients from Wavewatch III model outputs (Tolman 1997) sourced from the NCEP Climate Forecast System Reanalysis and NOAA Marine Modelling and Analysis Branch.

Marine litter pathways are represented by modelled particle trajectories. Each trajectory is linked to a source origin (region, country and city) and a date of release. A stochastic analysis of particle trajectories, densities and ages allows us to better understand dispersal dynamics for individual regions. Three types of metric are reported:

- the frequency of particle visits per model cell as a percentage of total particles, describing zones with high probabilities of occurrence and deposition of marine litter
- the minimum age of a particle that visited a model cell, depicting how fast marine litter can spread from its origin
- the average age of particles contained per model cell, from 0 to 10 years, showing timescale dependencies and movements of marine litter masses in time.

7.1 China

China, with 260 million of its citizens living within 50 km of the coast, contributed over a quarter of the estimated global amount of mismanaged plastic waste entering the ocean in 2010, generating 1.323 million to 3.528 million tonnes of plastic marine litter per year (Jambeck et al. 2015).

Modelled particle frequency and travel time analyses are shown in Figure 20. The dispersal model suggests that particles released from China are found mainly around the country's shoreline in the Yellow Sea and in the East and South China seas but also in the North Pacific convergence zone, indicating that the destination of marine litter originating in China is eventually far offshore. Model particles can travel to the edge of the Kuroshio Current along the coast of Japan within six months and then enter the eastern part of the North Pacific Ocean within one year. Depending on the season and prevailing winds, the model shows that marine litter can also enter the Gulf of Thailand and the Java Sea within one year.

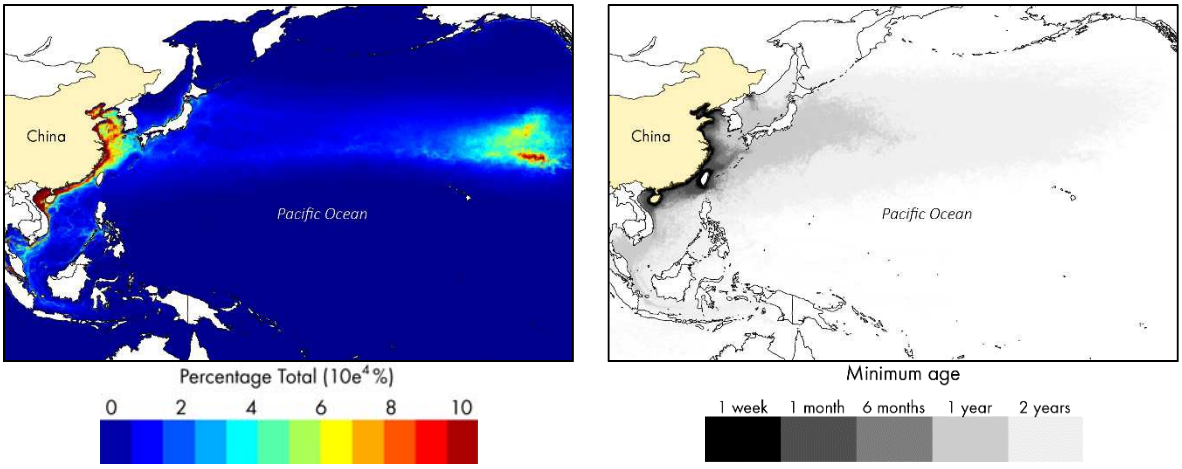
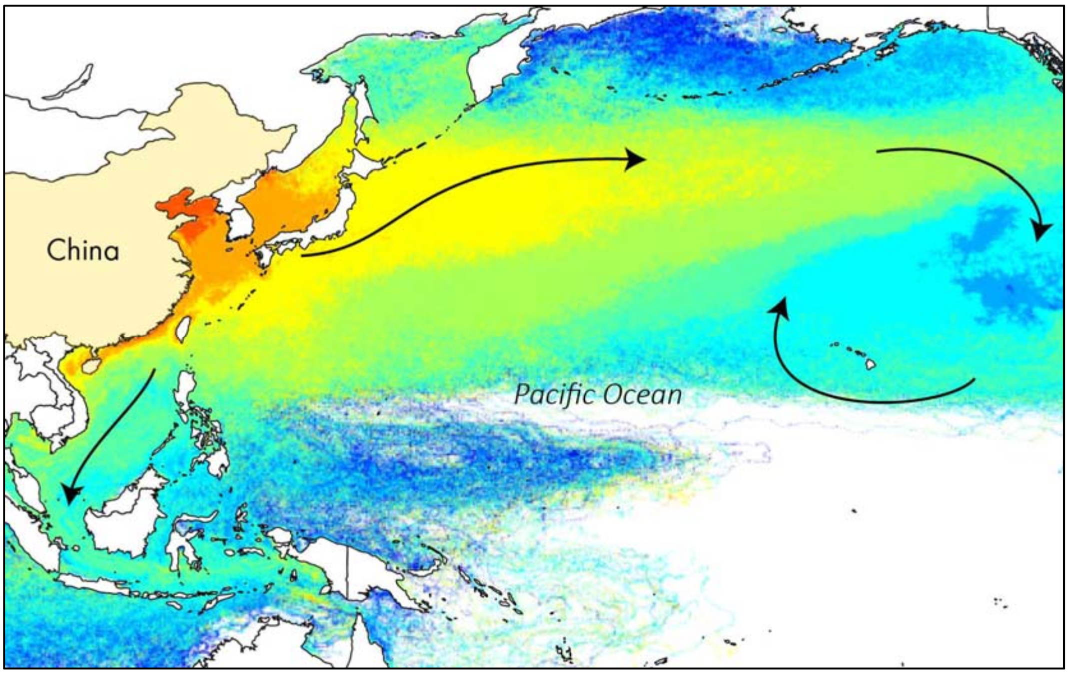


Figure 20. Frequency of particle visits as a percentage of total particle number for 1994–2014 (left) and minimum travel time for particles from China's coastline to the ocean (right)

Investigating the average age of particles in individual model cells allows us to better describe the movement of marine litter masses over time. Figure 21 shows the main modelled pathways of marine litter released from the Chinese seas. While most young particles (0–2 years) circulate inside the Yellow and Japan seas and along the coastline of South China, older particles (above 5 years) are found in three different oceans (the North Pacific, Indian and South Atlantic). Most particles enter the North Pacific Ocean through the Kuroshio Current and slowly drift in the subtropical convergence zone. However, some are transported south inside the Java Sea, where they can stay for several years, and eventually enter the Indian Ocean through the Malacca Strait and other straits of the Indonesian archipelago.



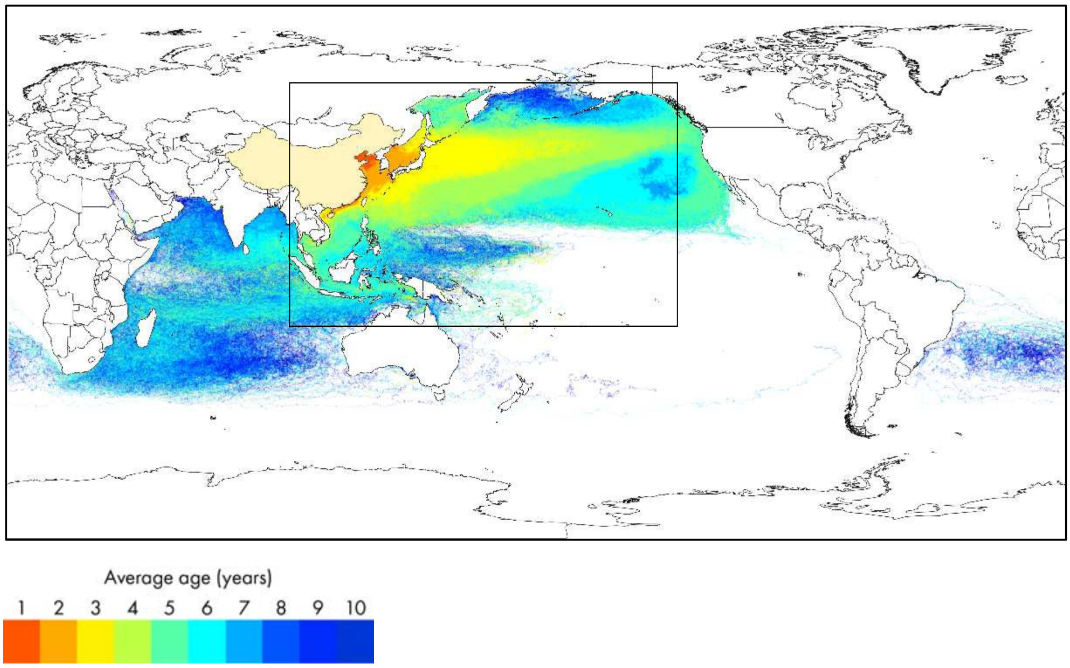


Figure 21. Average age in years of particles originating from China in the North Pacific Ocean (top) and globally (bottom), 1994–2014

7.2 Japan, South Korea, North Korea and Russia

Despite a relatively high population concentration in Japan (115 million coastal inhabitants) and South Korea (42 million coastal inhabitants), both countries generate a significantly smaller amount of plastic debris than China (annual predictions are between 21,000 and 57,000 tonnes and between 5,000 and 13,000 tonnes, respectively). Jambeck and colleagues (2015) estimate that North Korea (17 million coastal inhabitants) is the largest litter producer in this region, sending 46,000 to 122,000 tonnes of plastic into the ocean every year. This is assuming a rate of litter generation of 48 g/person/day, which is considerably higher than the rates of its neighbours (1 g in South Korea and 3 g in Japan).

Similarly to China, the model particles released in this region mainly enter the North Pacific Ocean and accumulate in the subtropical convergence zone. The ratio between particle frequency in the North Pacific gyre and the coastal areas is even higher than in the scenario for China (Figure 22). This suggests that debris released north of the Kuroshio Current is more likely to rapidly enter the North Pacific belt and travel eastward.

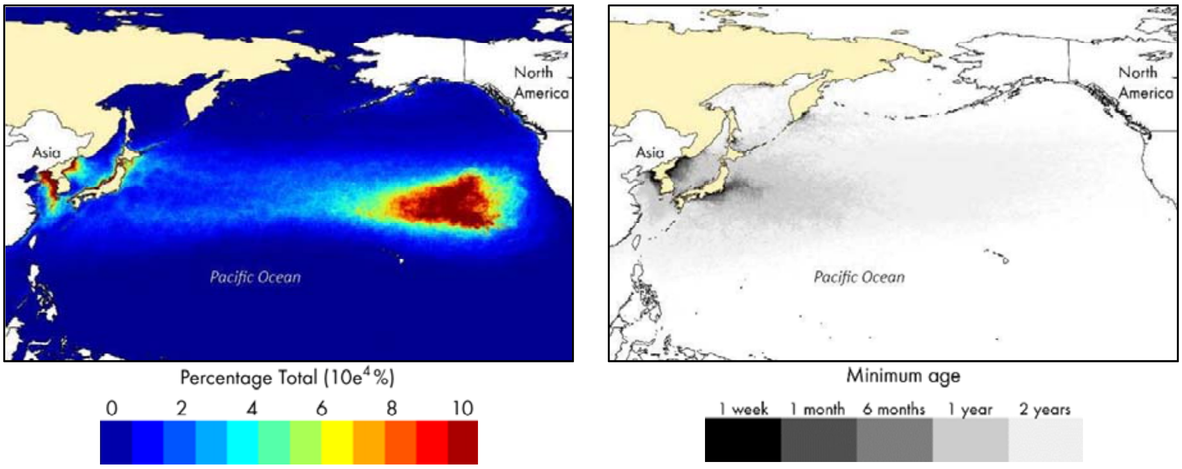
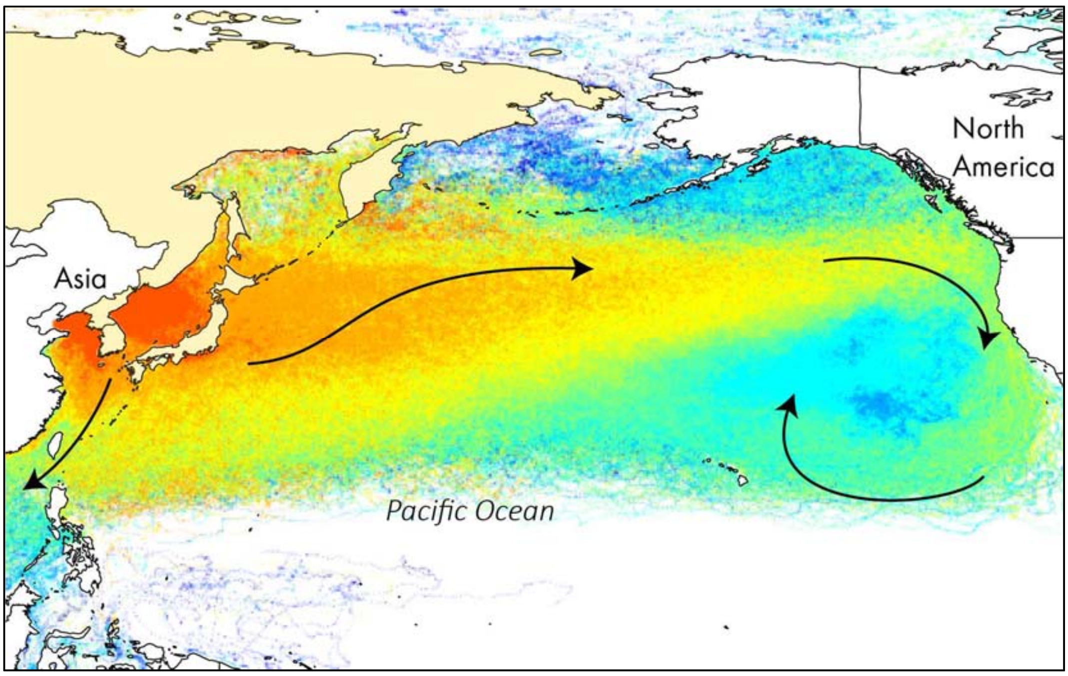


Figure 22. Frequency of particle visits as a percentage of total particle number for 1994–2014 (left) and minimum travel time for particles from Japan, Korea and the east coast of Russia to the ocean (right)

The dispersal model suggests that particles released from Japan, the Koreas or the east coast of Russia can reach the North American continent within three years (Figure 23). Note that this result is highly dependent on debris shape and buoyancy, which determine its exposure to wind. The vast quantity of floating debris produced by the 2011 Tōhoku earthquake-induced tsunami has been well documented (Bagulayan *et al.* 2012; Lebreton & Borrero 2013; Calder *et al.* 2014). Vessels and other high-windage objects of Japanese origin where observed in Alaska and British Columbia within only one year after the catastrophe. However, for this study, less buoyant debris is considered, as a lower floatability is more representative of litter generated by coastal populations from consumer goods.



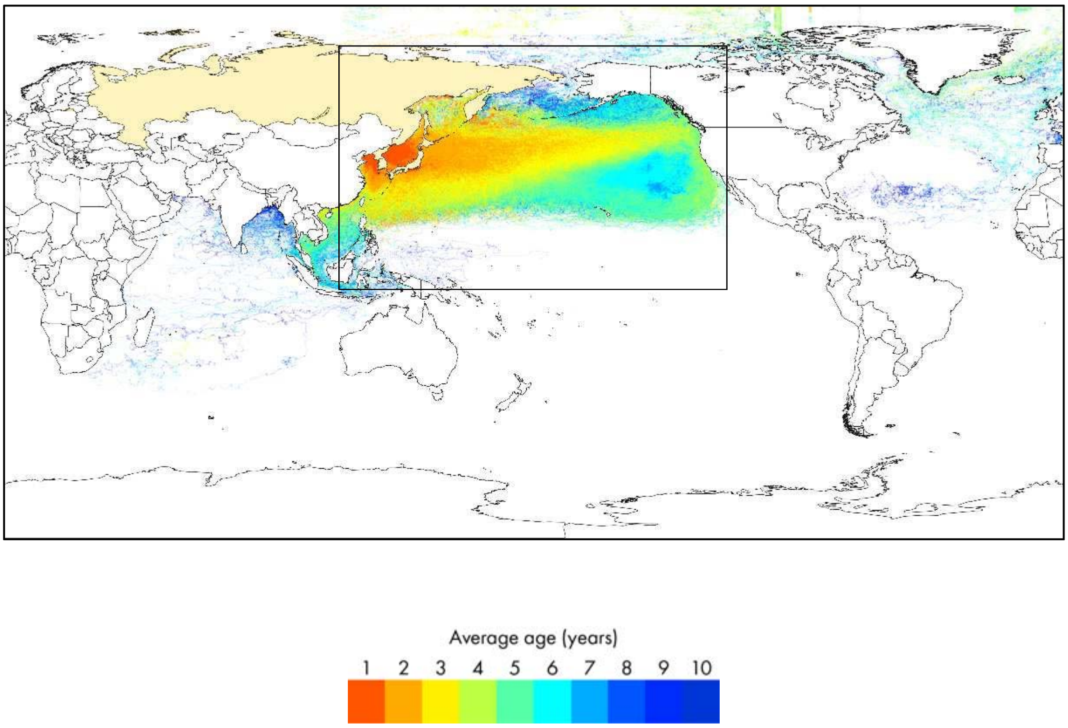


Figure 23. Average age in years of particles originating from Japan, Korea and the east coast of Russia in the Pacific Ocean (top) and globally (bottom), 1994–2014

7.3 Southeast Asia

Southeast Asia has 400 million coastal inhabitants, who are citizens of Brunei Darussalam, Myanmar, Indonesia, Malaysia, the Philippines, East Timor, Singapore, Thailand, Cambodia and Vietnam. Three of those countries (Indonesia, Philippines and Vietnam) are in the top five countries generating mismanaged waste in coastal areas. The annual amounts of plastic marine debris produced by these countries were estimated at 483,000 to 1,287,000 tonnes for Indonesia, 283,000 to 753,000 tonnes for the Philippines and 275,000 to 734,000 tonnes for Vietnam (Jambeck et al. 2015).

Model-predicted trajectories show that marine litter from Southeast Asia is likely to enter the Indian, Pacific and Atlantic oceans within less than 10 years. However, normalised particle visit frequencies (Figure 24) show that a significant amount of material is still found around land masses, between the various islands and straits of the archipelago. In particular, the model shows the Bay of Bengal, the Gulf of Thailand, the Malacca Strait, the Gulf of Tonkin and most east-facing shores of Indonesian islands as regional deposition hotspots. However, marine litter can escape the Southeast Asian archipelago relatively quickly, as some of the particles travel as far as in the subtropical latitudes of the Indian Ocean or near the start of the Kuroshio Current within one year.

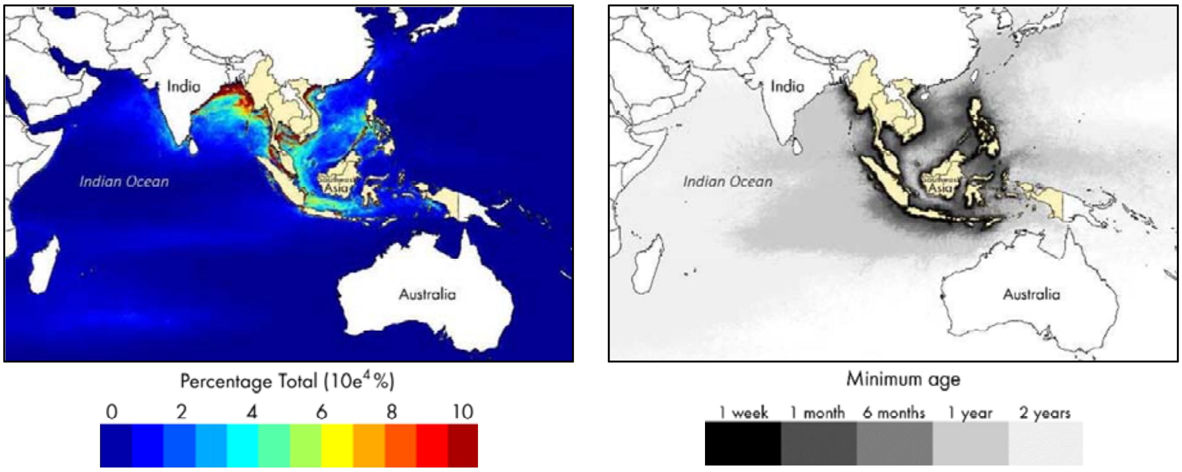
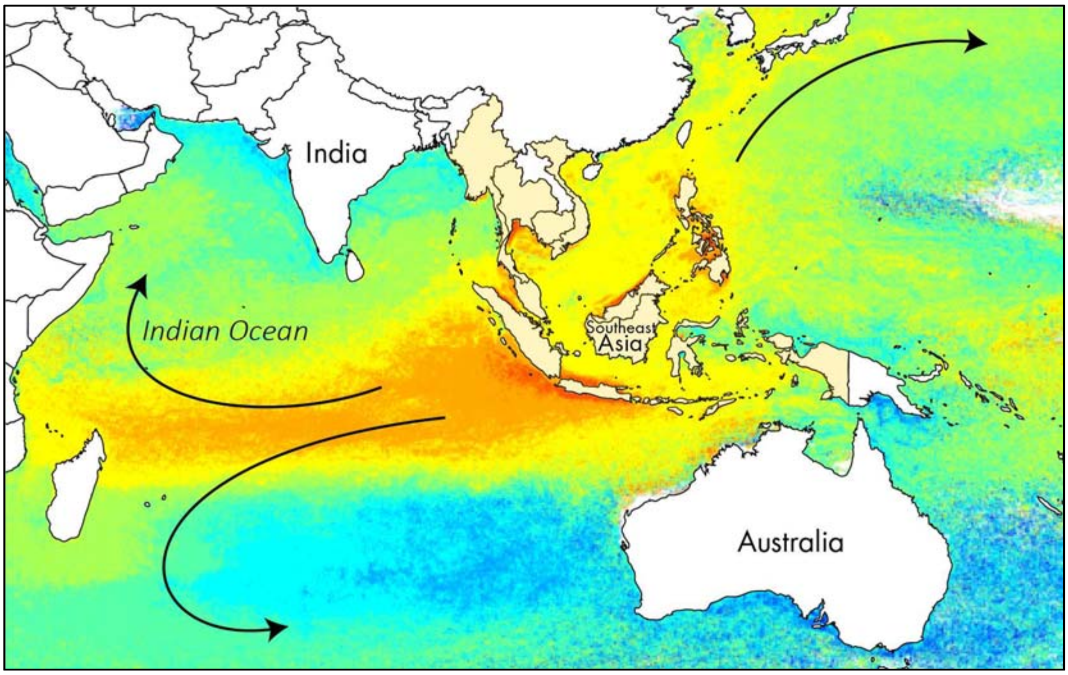


Figure 24. Frequency of particle visits as a percentage of total particle number for 1994–2014 (left) and minimum travel time for particles from Southeast Asia's coastline to the ocean (right)

Modelled marine litter usually enters the Indian Ocean from Southeast Asia within one to two years (Figure 25). From there, it drifts away from the equatorial latitudes towards the subtropics. In the south, the particles enter the Indian Ocean convergence zone south-east of Madagascar and can eventually leak into the South Pacific Ocean or the Atlantic Ocean within 5–10 years. In the north, the particles are contained between the equatorial countercurrent and the Indian subcontinent. Episodes of monsoon in the region regularly push the material back to the coastline, producing an average age of particles in the Arabian Sea and the Bay of Bengal above five years. Particles can also enter the North Pacific Ocean from Southeast Asia through the Kuroshio Current and eventually accumulate in the subtropical convergence zone.



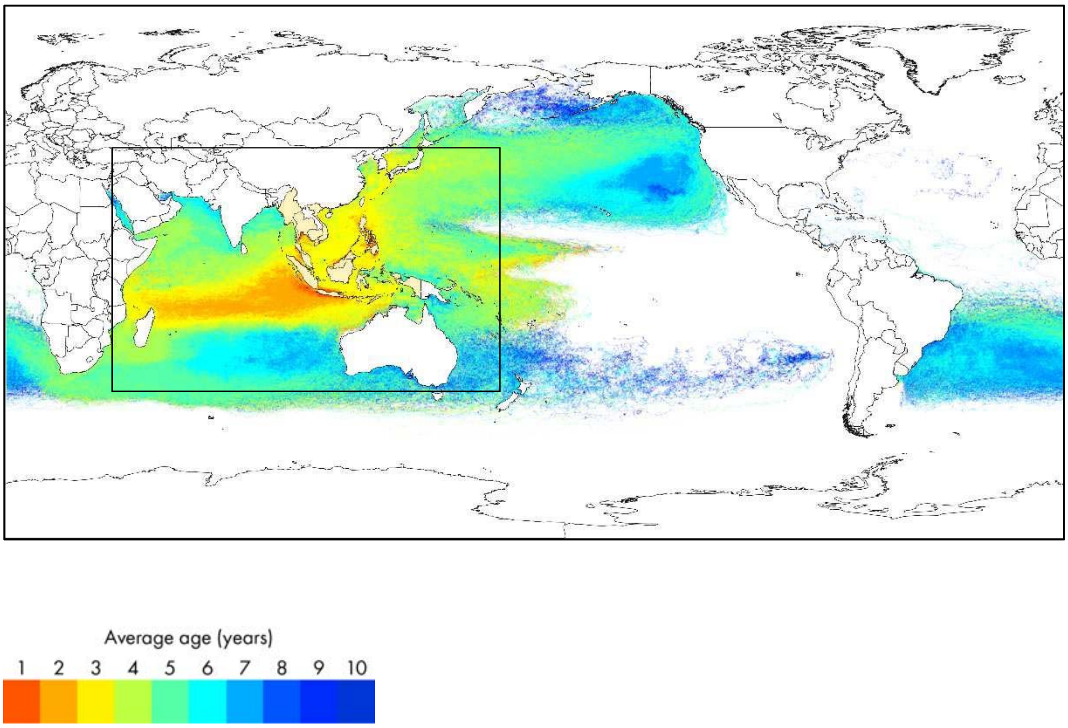


Figure 25. Average age in years of particles originating from Southeast Asia in the Indian and Pacific Oceans (top) and globally (bottom), 1994–2014

7.4 Australia and the South Pacific

The South Pacific region is far less populated than the Asian continent and is reasonably regarded as a minor source of marine litter at the global scale. However, the estimated quantity of mismanaged plastic waste generated per inhabitant is relatively high in the region, particularly for Pacific islands. The total amount of marine litter entering the ocean in the South Pacific region is expected to triple by 2025.

Particles were released from Australia, New Zealand, Solomon Islands, Fiji, French Polynesia, New Caledonia, Vanuatu, Papua New Guinea and Samoa. The trajectories extend to four different oceans within 10 years of simulation: the North and South Pacific oceans, the Indian Ocean and the South Atlantic Ocean. Locally, Australian waters north of the Great Barrier Reef and also the Gulf of Papua show high frequencies of model particle visits (Figure 26) suggesting potential regional hotspots for accumulation.

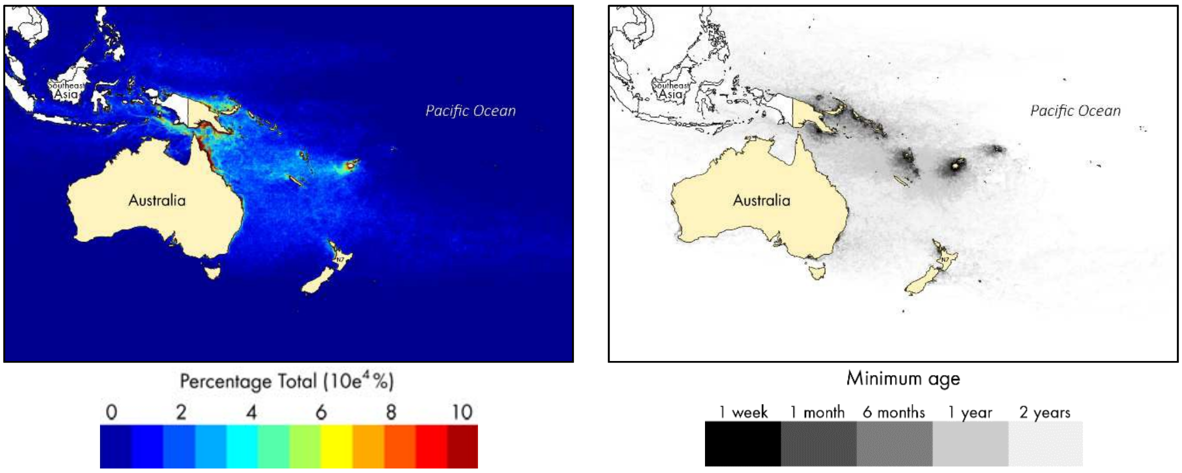
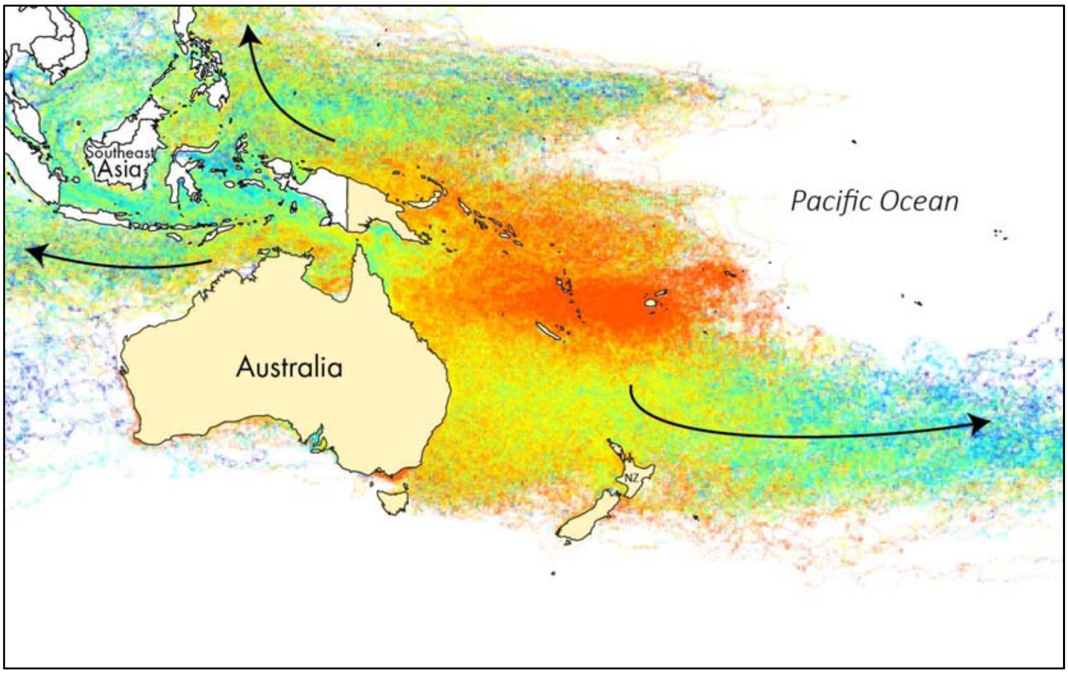


Figure 26. Frequency of particle visits as a percentage of total particle number for 1994–2014 (left) and minimum travel time for particles from the South Pacific region's coastline to the ocean (right)

The model predicts that marine litter from South Pacific islands will usually circulate at tropical latitudes within the first year, pushed by trade winds towards the Coral Sea and Australia. Some particles travel south following the East Australian Current to enter the Tasman Sea and eventually the South Pacific subtropical convergence zone. However, other particles drift north towards the Southeast Asian archipelago and reach the same fate as debris from that region by eventually entering the Indian Ocean or the North Pacific Ocean. Particles released from New Zealand mostly escape eastward towards the South Pacific. In Australia, very few model particles released in the west or south leave the continental shelf within the first few years. The particles are regularly pushed back to the landmass by episodes of swells and storm winds and travel along the coastline in the predominant direction.



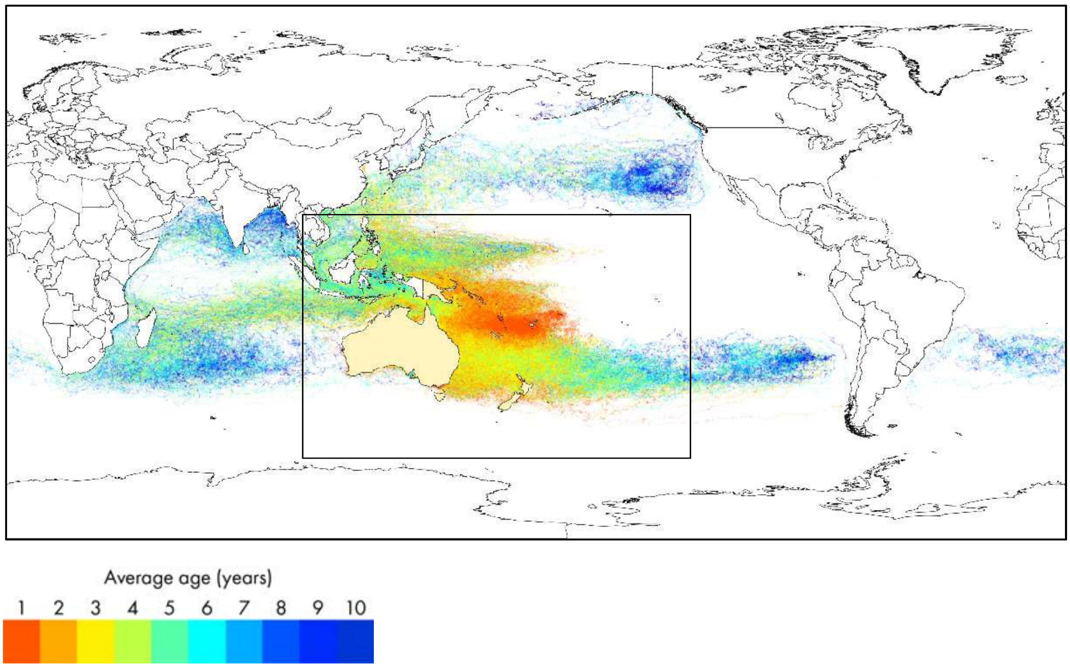


Figure 27. Average age in years of particles originating from the South Pacific region in the Pacific Ocean (top) and globally (bottom), 1994–2014

7.5 North and Central America (Pacific coastline)

North and Central American countries facing the Pacific Ocean are Canada, the United States, Mexico, Guatemala, El Salvador, Honduras, Nicaragua, Costa Rica and Panama. In Central America, El Salvador is considered to be the largest producer of marine plastic litter, producing 18,000 to 47,000 tonnes of annual input, assuming a rate of 51 g per coastal inhabitant per day (Jambeck *et al.* 2015). In North America, the United States is the largest emitter of litter, producing 41,000 to 110,000 tonnes of plastic litter per year. However, that figure is for both sides of the country (the Pacific and Atlantic sides), and the eastern side is significantly more populated.

Model particles released in the Pacific waters of the North American continent travel southward with the California Current and rapidly migrate offshore when they reach the tropical latitudes. From there, the particles cross the Pacific Ocean and can reach Southeast Asia within two years. Eventually, most particles then travel back east towards the North Pacific convergence zone, where they accumulate after 5–10 years on average (Figures 28 and 29).

As a comparison, between 2012 and 2014, the boat of fisherman Jose Salvador Alvarenga was reported lost off the coast of Mexico and was only found 13 months later in the Marshall Islands on the other side of the Pacific Ocean.

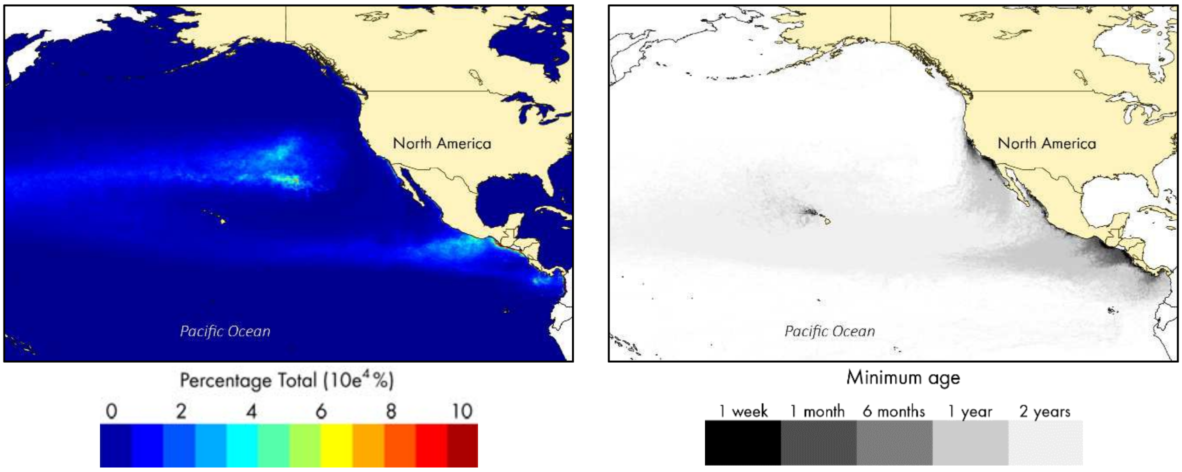
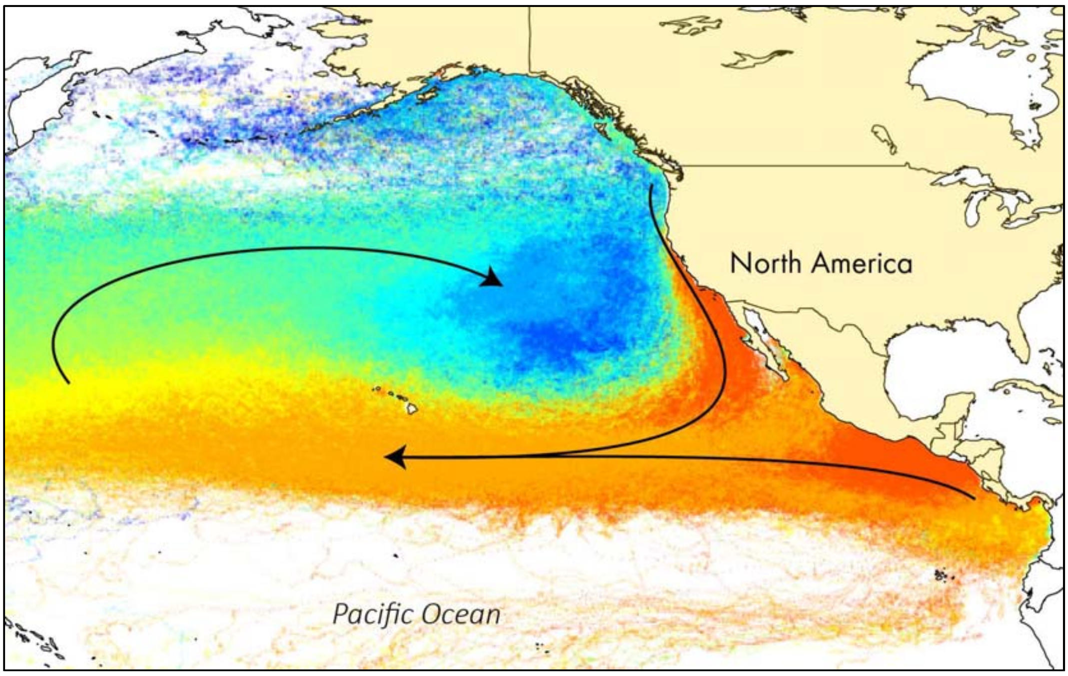


Figure 28. Frequency of particle visits as a percentage of total particle number for 1994–2014 (left) and minimum travel time for particles from North and Central America's Pacific coastline to the ocean (right)



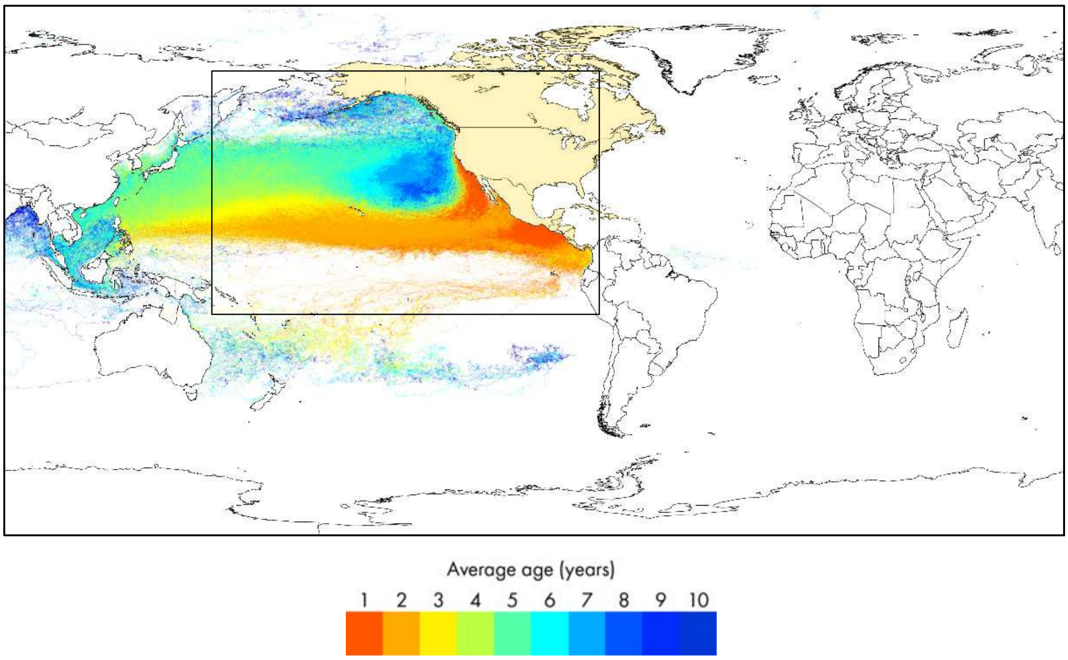


Figure 29. Average age in years of particles originating from North and Central America's Pacific coastline in the North Pacific Ocean (top) and globally (bottom), 1994–2014

7.6 North and Central America (Atlantic coastline)

Countries of North and Central America sharing a stretch of coastline inside the Caribbean Sea, the Gulf of Mexico or the Atlantic Ocean were selected for this scenario, including Mexico, Belize, Guatemala, Honduras, Nicaragua, Costa Rica, Panama, Colombia, Venezuela, Bahamas, Barbados, Cuba, the Dominican Republic, Guadeloupe, Haiti, Jamaica, Martinique, Trinidad and Tobago, Guam, Puerto Rico, the United States and Canada. For these countries, Jambeck *et al.* (2015) reported an annual plastic litter input of between 171,000 and 457,000 tonnes for 2010. The major regional producers are the United States (41,000–110,000 tonnes, Pacific and Atlantic coastline), Haiti (22,000–59,000 tonnes) and the Dominican Republic (18,000–47,000 tonnes).

Most model particles released inside the Caribbean Sea and the Gulf of Mexico eventually escape the region through the Gulf Stream along the coast of Florida. Particles can reach the North Atlantic Ocean within one year and finally accumulate inside the subtropical convergence zone known as the Sargasso Sea (Figure 30). However, some particles never leave the marginal seas of the Caribbean region. Regional hotspots were identified in the eastern part of the Gulf of Mexico and the Bay of Honduras. In the Caribbean islands, the model usually predicts higher accumulation in the southern side of an island. Far away from the North American continent, the Bay of Biscay in Europe shows a relatively high rate of particle frequency, suggesting a potential accumulation of marine debris in that area.

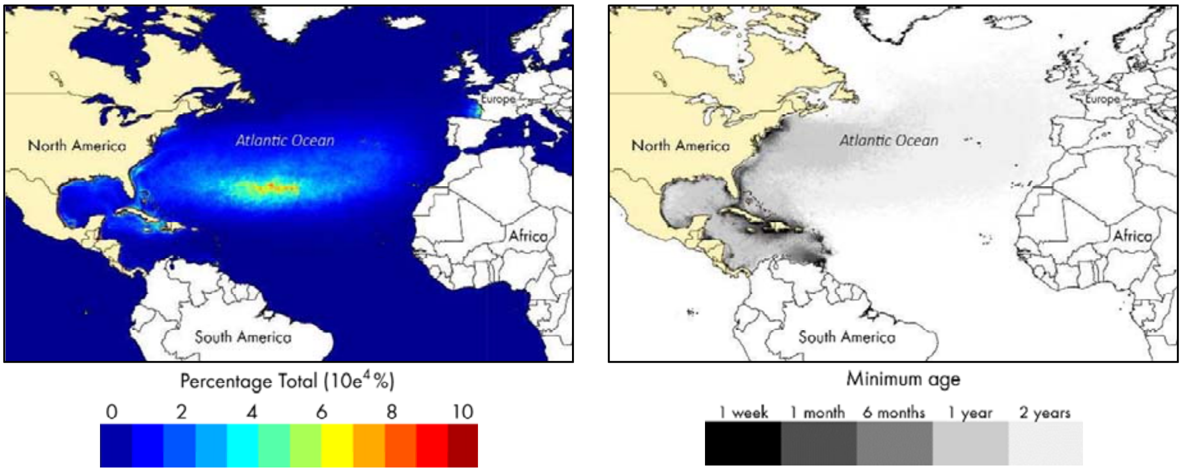
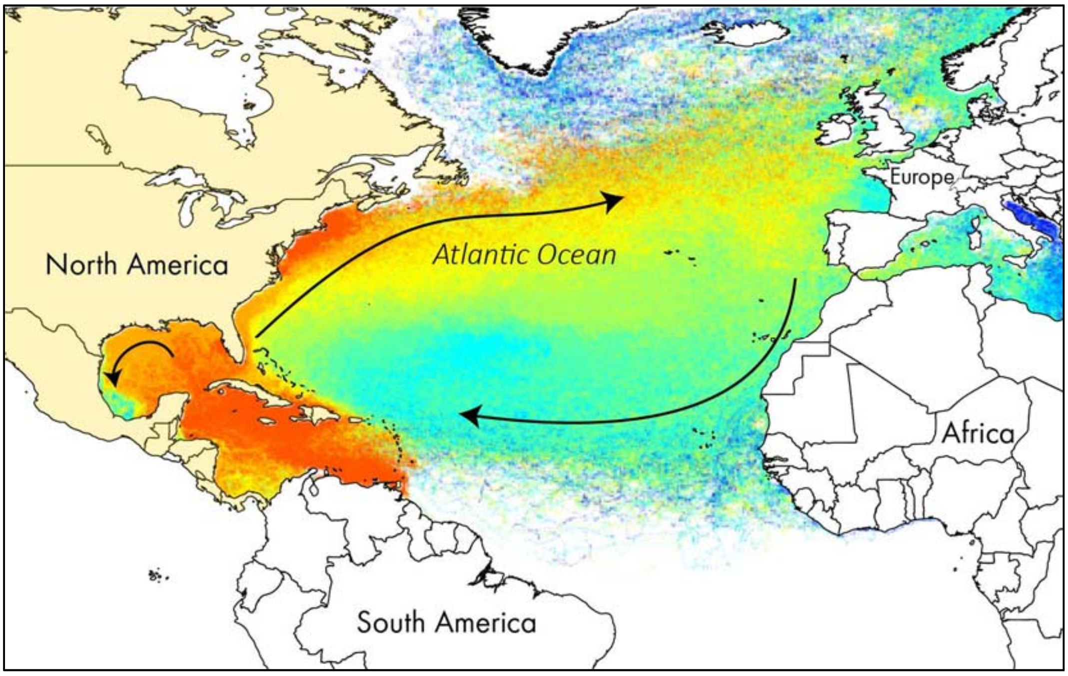


Figure 30. Frequency of particle visits as a percentage of total particle number for 1994–2014 (left) and minimum travel time for particles from North and Central America's Atlantic coastline to the ocean (right)

Modelled particles usually stay in the Caribbean Sea and the Gulf of Mexico for one to two years and leave the region by entering the Atlantic Ocean within three years. From there, within four years, they follow the Gulf Stream towards northern Europe and circulate either north along the Scandinavian peninsula into the Arctic or south along the Iberian peninsula, West Africa and back into the subtropical convergence zone. The model shows a significant proportion of old particles in the southeast part of the Gulf of Mexico, suggesting an important accumulation rate in this area (Figure 31).



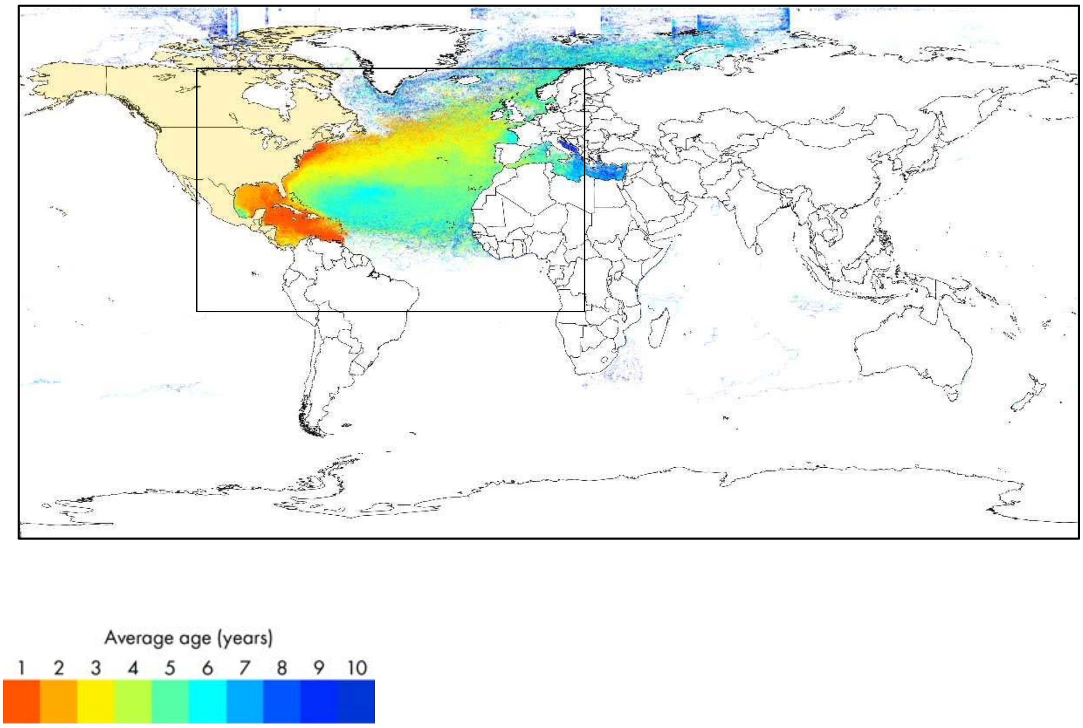


Figure 31. Average age in years of particles originating from North and Central America's Atlantic coastline in the North Atlantic Ocean (top) and globally (bottom), 1994–2014

7.7 South America

Nearly 150 million people live within 50 km of the ocean in South America. According to Jambeck *et al.* (2015), the largest emitter of marine litter in the region is Brazil, which sends an estimated 71,000–189,000 tonnes of mismanaged plastic into the ocean every year. Brazil is followed by Peru (29,000–78,000 tonnes per year) and Argentina (24,000–63,000 tonnes per year).

Model particles released from South Africa where found in all five main oceans of the world, but mainly in the South Atlantic and South Pacific oceans (Figure 32).

On the west coast, particles are transported towards the equator in the Humboldt Current. When they reach equatorial latitudes, they migrate towards the west with the South or North Equatorial Current and accumulate in the Pacific Ocean. Similarly to the North and Central American Pacific coastline scenario, the particles can reach Southeast Asia or Oceania within one to two years (Figure 33).

On the east coast, model particles released north of the equator are likely to drift northward into the Caribbean Sea and later into the North Atlantic (after three years on average). Particles released south of the equator are transported south along the continent in the Brazil Current and eventually accumulate in the South Atlantic convergence zone.

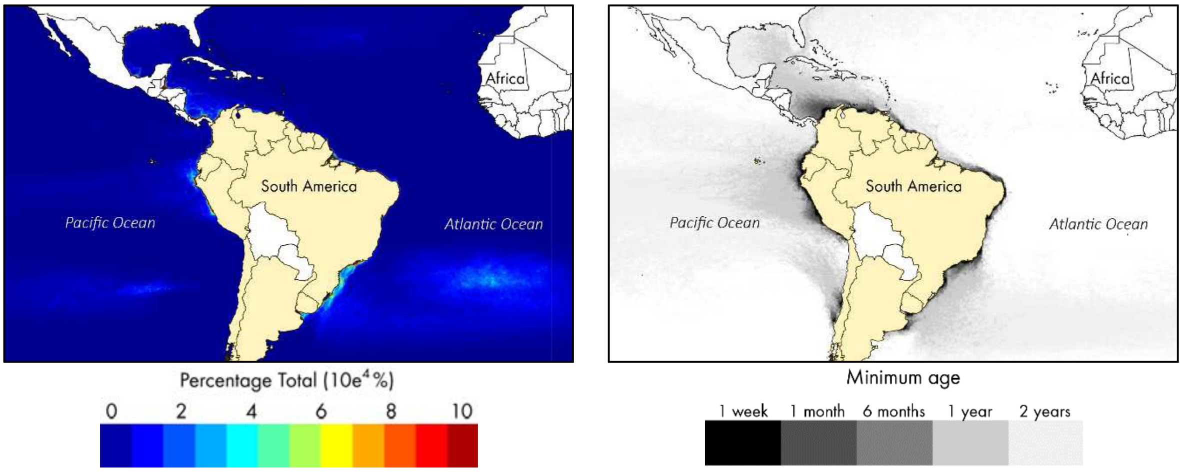
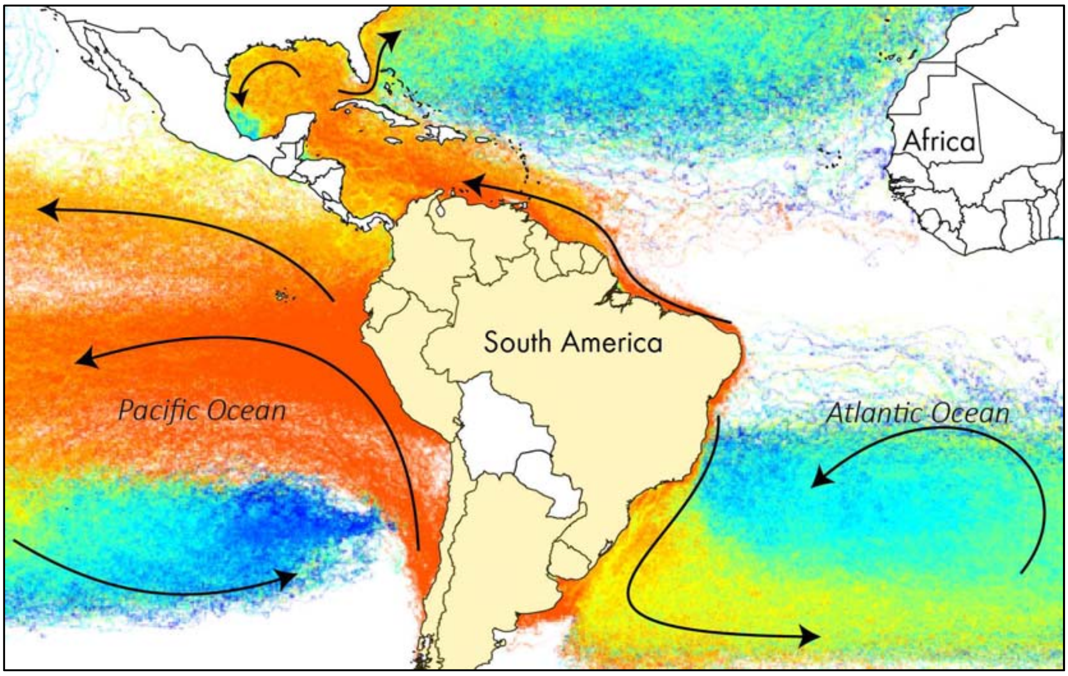


Figure 32. Frequency of particle visits as a percentage of total particle number for 1994–2014 (left) and minimum travel time for particles from the South America's coastline to the ocean (right)



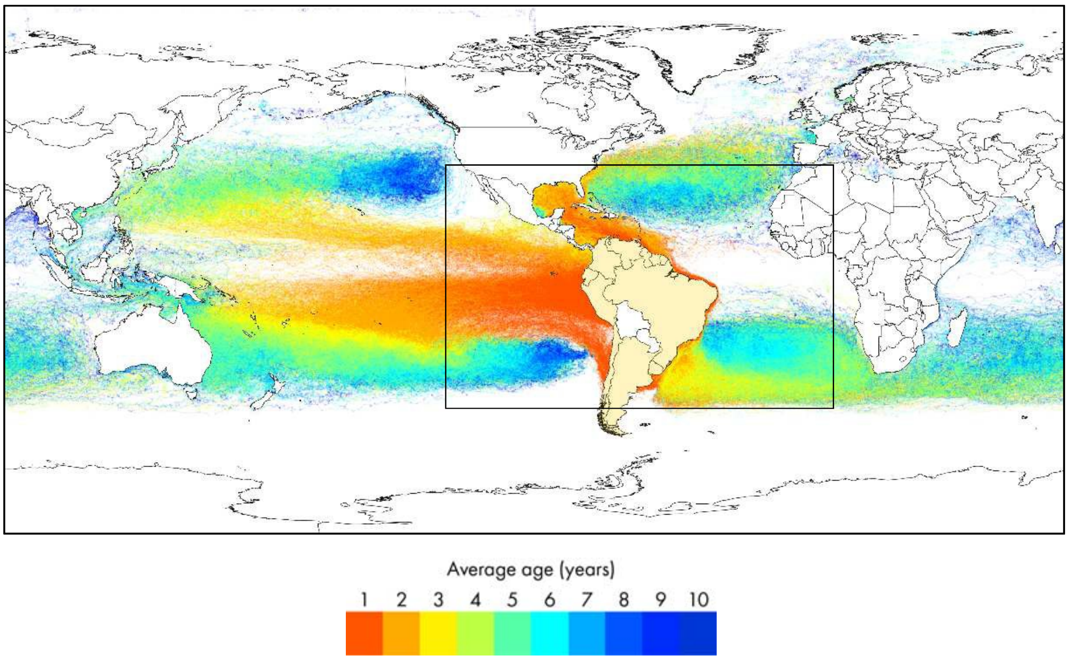


Figure 33. Average age in years of particles originating from South America in the Pacific and Atlantic Oceans (top) and globally (bottom), 1994–2014

7.8 West Africa

The region of West Africa from Morocco to Namibia has more than 92 million people living less than 50 km from the Atlantic Ocean. The largest emitter of mismanaged litter is Nigeria, which sent a predicted 128,000 to 341,000 tonnes of plastic into the marine environment in 2010 (Jambeck *et al.* 2015), followed by Morocco (47,000–124,000 tonnes). Other significant contributions in sub-Saharan Africa are those of Senegal (38,000–102,000 tonnes per year) and Ivory Coast (29,000–78,000 tonnes per year)

Surprisingly, the dispersal model shows that West Africa has a relatively low impact on the South Atlantic Ocean but instead affects the North Atlantic Ocean (Figure 34). Most model particles released in West Africa leave the continent within one year and travel west towards the South American coastline. Because most particles arrive in South America north of the equator, they follow the Caribbean Current and enter the Gulf of Mexico within three years and eventually leak into the North Atlantic gyre after five years on average.

Notably, the model predicts the Gulf of Guinea as a local hotspot for marine litter accumulation. A significant amount of particles remains trapped by the Guinea Current, which flows eastward, and can stay in the region for 5–10 years.

The dispersal model shows that particles from West Africa can reach the Arctic Circle within 10 years (Figure 35).

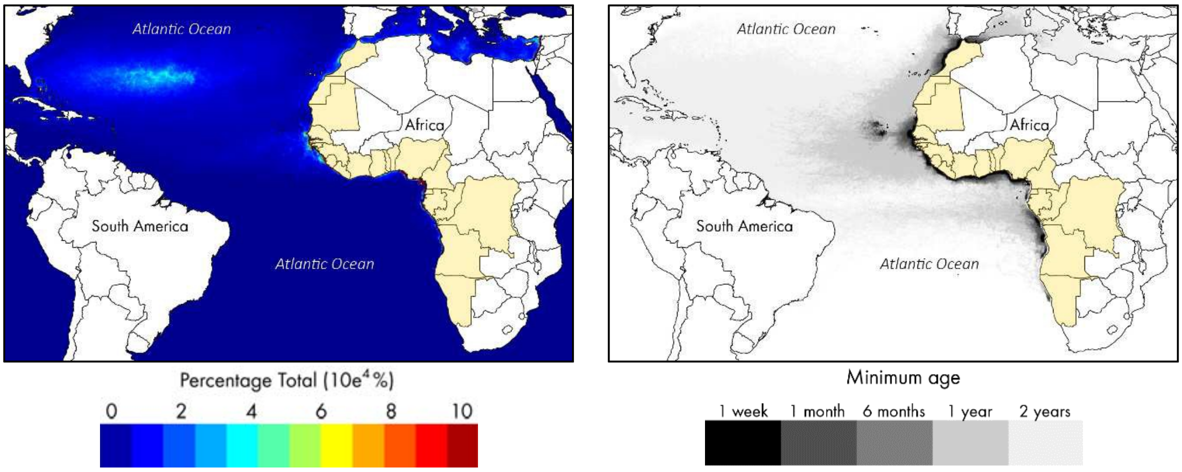
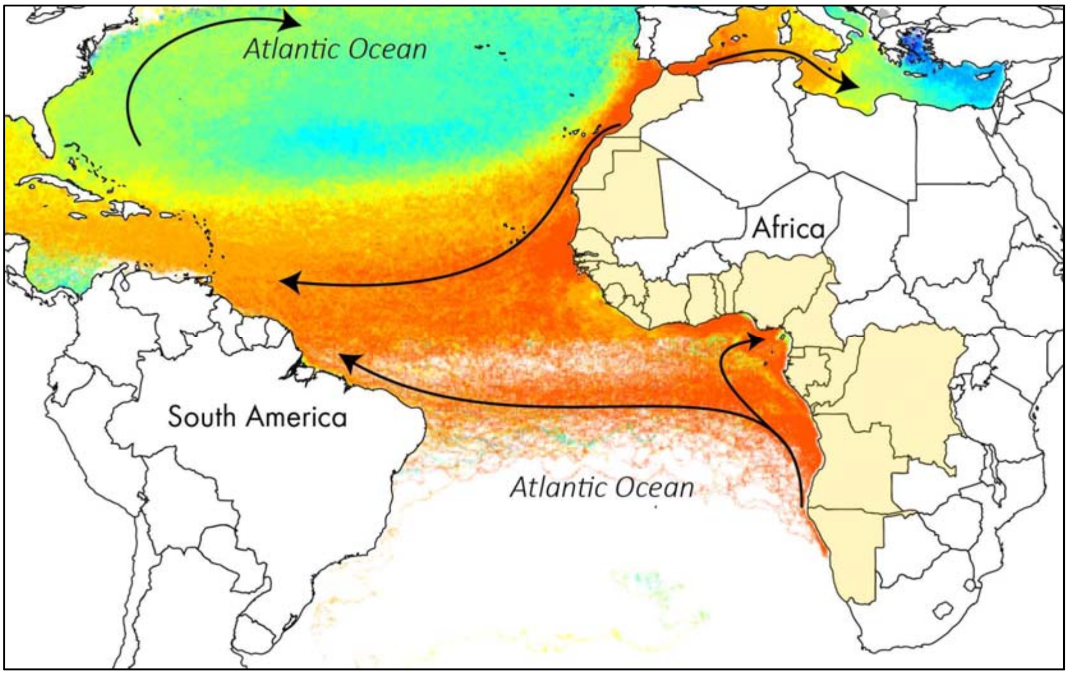


Figure 34. Frequency of particle visits as a percentage of total particle number for 1994–2014 (left) and minimum travel time for particles from West Africa's coastline to the ocean (right)



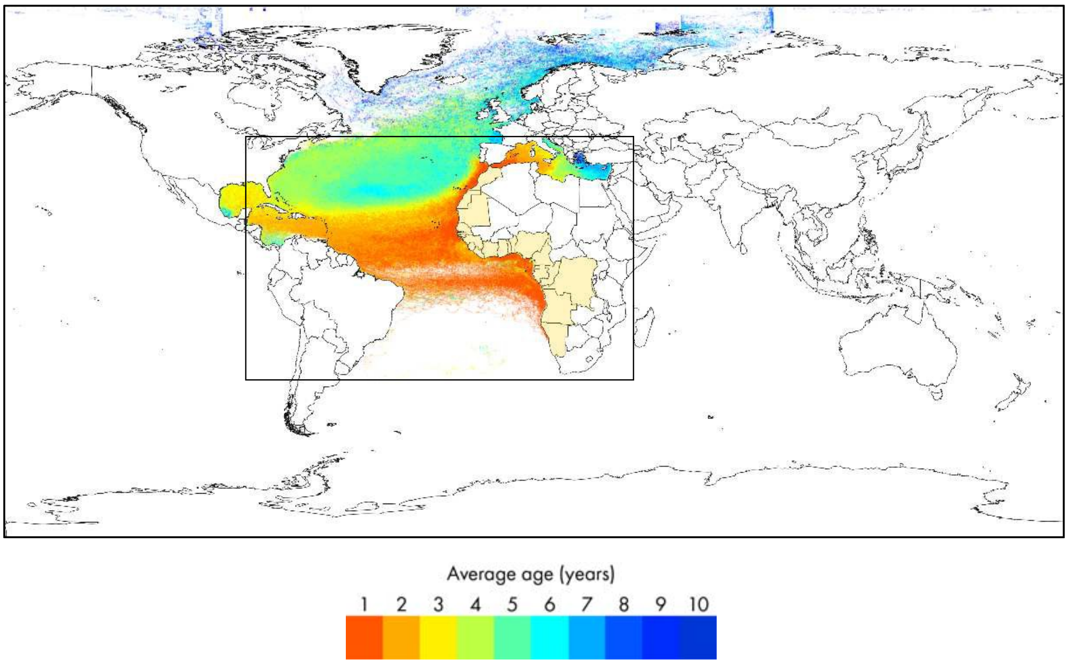


Figure 35. Average age in years of particles originating from West Africa in the Atlantic Ocean (top) and globally (bottom), 1994–2014

7.9 East Africa

More than 46 million people live in the coastal areas of East Africa. Data on mismanaged waste for Djibouti, Somalia, Kenya, Tanzania, Mozambique, South Africa, Madagascar, Mauritius and the island of La Reunion suggest an annual marine litter input from East Africa of between 135,000 and 361,000 tonnes for 2010 (Jambeck *et al.* 2015). The largest plastic litter producer in the region is South Africa, at 95,000–252,000 tonnes per year.

In East Africa, the dispersal model predicts that particles released south of Kenya, including from Madagascar, are likely to be transported by the warm Agulhas Current at the southern tip of the African continent. From there, they either follow the west wind drift and enter the Indian Ocean or enter the Atlantic Ocean through the colder Benguela Current. However, particles released north of the equator are more likely to drift in the Arabian Sea and later in the Gulf of Bengal. The model predicts a significant probability of accumulation along the eastern coastline of Africa and Madagascar, particularly the sections that face south in Somalia, Kenya and Mozambique (Figure 36). Particles in the area are transported by the south-easterly trade winds and remain very close to the shoreline.

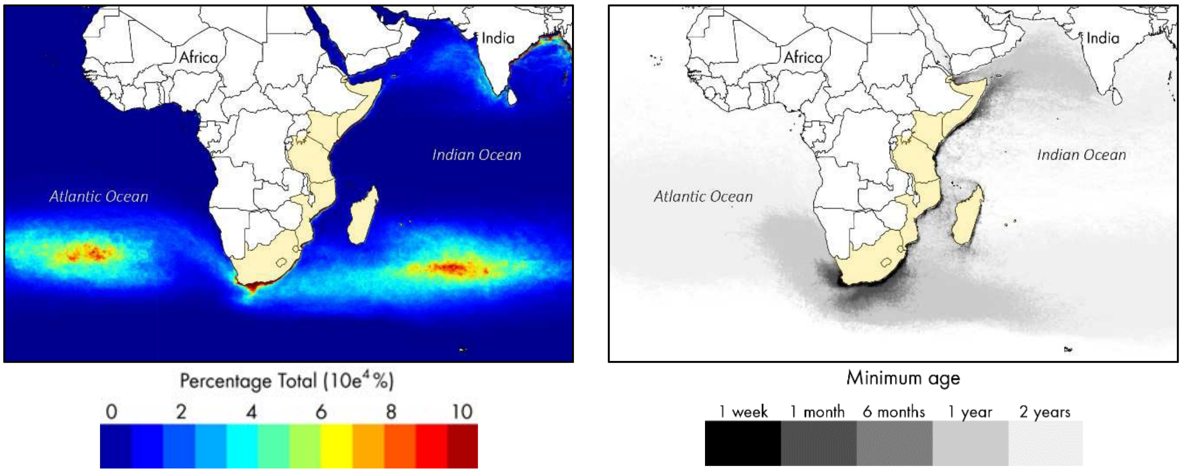


Figure 36. Frequency of particle visits as a percentage of total particle number for 1994–2014 (left) and minimum travel time for particles from East Africa's coastline to the ocean (right)

Modelled particles in East Africa usually leave the region within one to two years. When not stranded, they escape the continent at the southern tip where the Indian and Atlantic oceans meet, or at the north in the Arabian Sea. Within five years, most drifting particles reach the subtropical convergence zones in the Indian and South Atlantic oceans or the Bay of Bengal, which appears as a regional hotspot for accumulation. The model predicts that marine litter from East Africa could travel as far as the North Atlantic and the South Pacific's eastern side within 10 years (Figure 37).

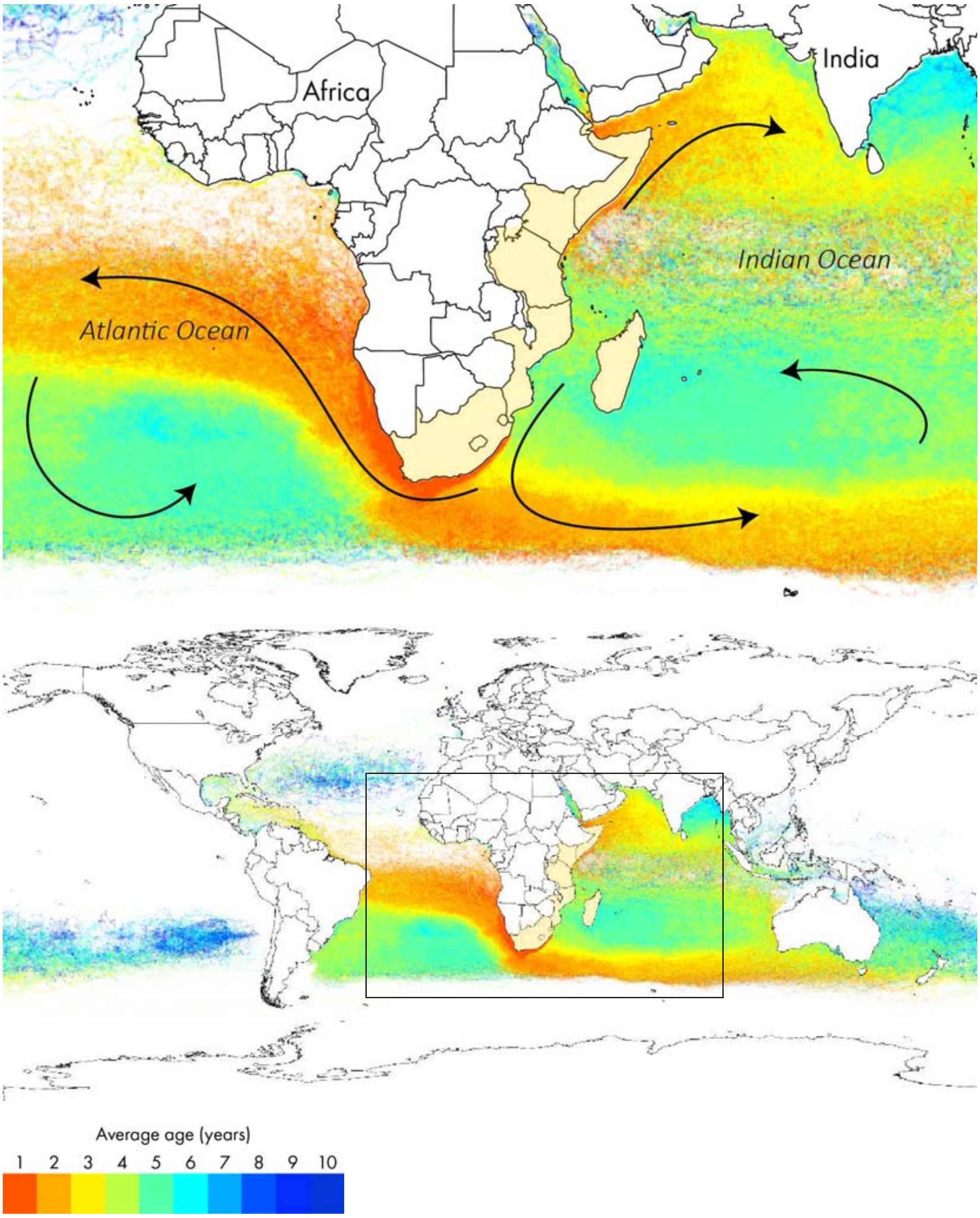


Figure 37. Average age in years of particles originating from East Africa in the Indian and Atlantic Oceans (top) and globally (bottom), 1994–2014

7.10 India, Bangladesh, Sri Lanka and the Arabian Peninsula

According to data on waste management and coastal population density (Jambeck *et al.* 2015), Sri Lanka ranks first as an emitter of marine litter in this region, with an estimated annual input of between 239,000 and 636,000 tonnes. Overall, this region could leak from 594,000 to 1,585,000 tonnes of marine plastic litter every year, with significant contributions also from Bangladesh (118,000–315,000 tonnes per year) and India (90,000–240,000 tonnes per year). Countries of the Arabian Peninsula that face the Indian Ocean (Iran, Iraq, Kuwait, Oman, Qatar, Saudi Arabia, the United Arab Emirates and Yemen) produce between 71,000 and 189,000 tonnes of litter each year.

Of all the regions covered in this modelling assessment, the Indian subcontinent and Arabian peninsula involve the least dispersive scenario. Model particles mostly remain in the area between the strong westward equatorial current and the landmass (Figure 38). Periodic episodes of monsoon conditions constantly push the material back to shore, particularly on the west side of India and inside the Bay of Bengal.

Some particles eventually find their way into the Southern Hemisphere with the Agulhas current along the eastern side of Africa (Figure 39). During the 10 years of the simulation, some model particles were recorded in the South Atlantic and Pacific oceans.

The model also suggests that, while the Persian Gulf is leaking material (average particle age below one year), the Arabian Sea is a distinct accumulation zone (average particle age up to 10 years).

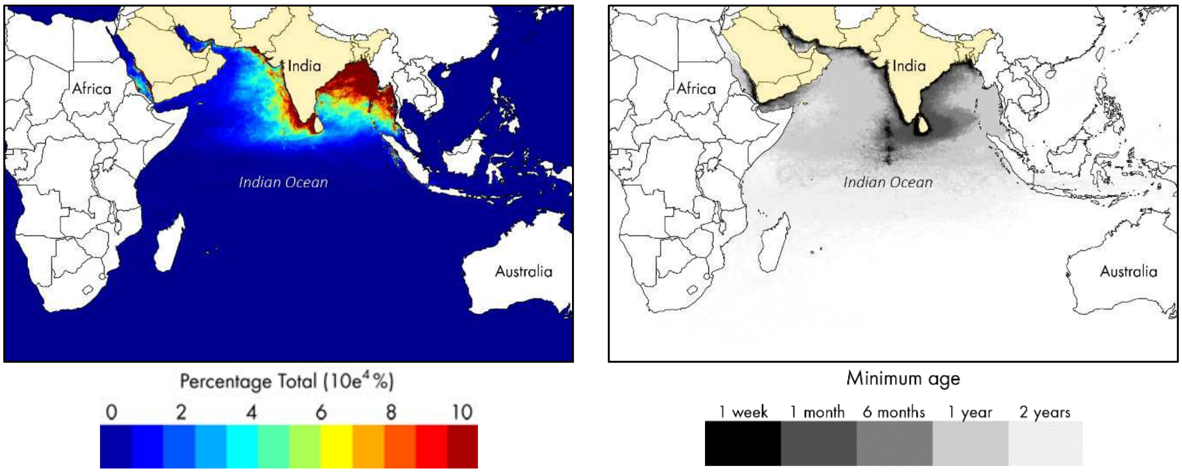
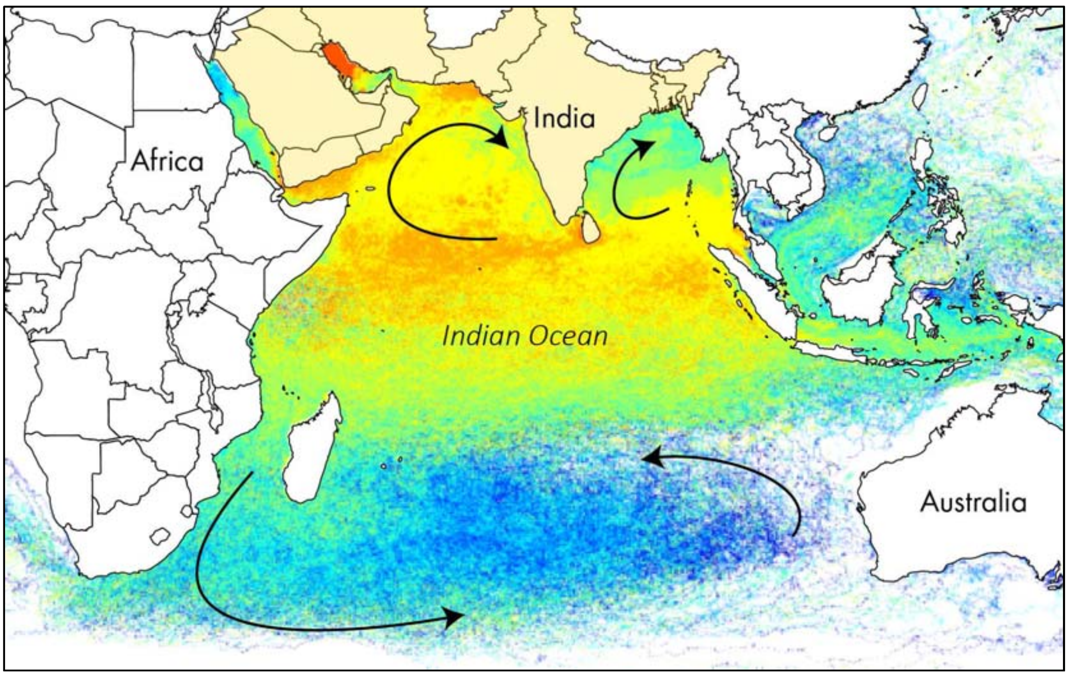
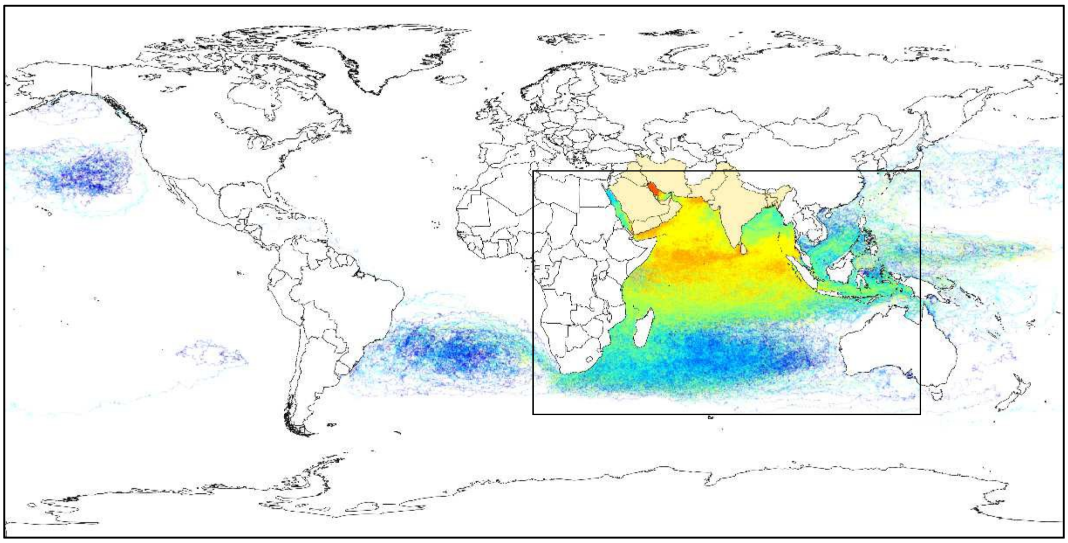


Figure 38. Frequency of particle visits as a percentage of total particle number for 1994–2014 (left) and minimum travel time for particles from India, Bangladesh, Sri Lanka and the Arabian Peninsula's coastline to the ocean (right)





Average age (years)
1 2 3 4 5 6 7 8 9 10

Figure 39. Average age in years of particles originating from India, Bangladesh, Sri Lanka and the Arabian Peninsula in the Indian Ocean (top) and globally (bottom), 1994–2014

7.11 Western and Northern Europe

Western and Northern Europe have more than 130 million coastal inhabitants in Belgium, Denmark, Finland, France, Germany, Iceland, Ireland, Norway, Portugal, Spain, Sweden, the Netherlands and the United Kingdom. The region released an estimated 37,000 to 98,000 tonnes of plastic litter into the marine environment in 2010 (Jambeck *et al.* 2015). On average, the countries of Northern and Western Europe have a relatively low rate of mismanaged waste production per inhabitant, mainly because they have modern municipal solid waste management infrastructure. The lowest rate is in Denmark and Sweden (1 g/person/day). Germany has the highest rate (10 g/person/day).

The dispersal model shows the Gulf of Biscay in the south-west of France and the western shores of the United Kingdom and Northern Europe from Belgium to Norway as local hotspots for frequency of marine litter (Figure 40). However, a significant amount of material leaks into the North Atlantic Ocean and accumulates in the Sargasso Sea. In the Mediterranean, particles from Spain and France but also other western countries accumulate around the African coasts (Figure 41).

Particles from Western and Northern Europe often reach the Arctic Circle within three to four years on average.

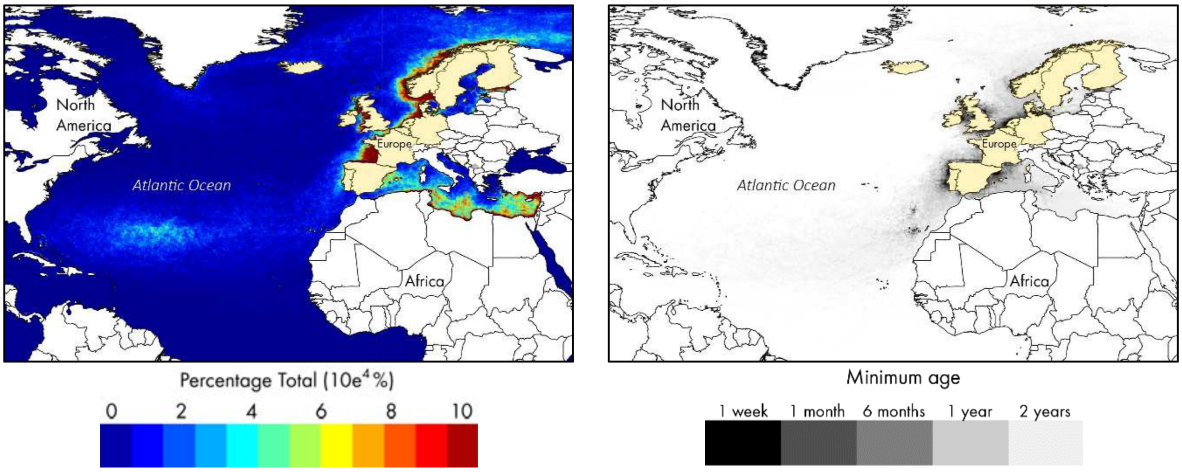
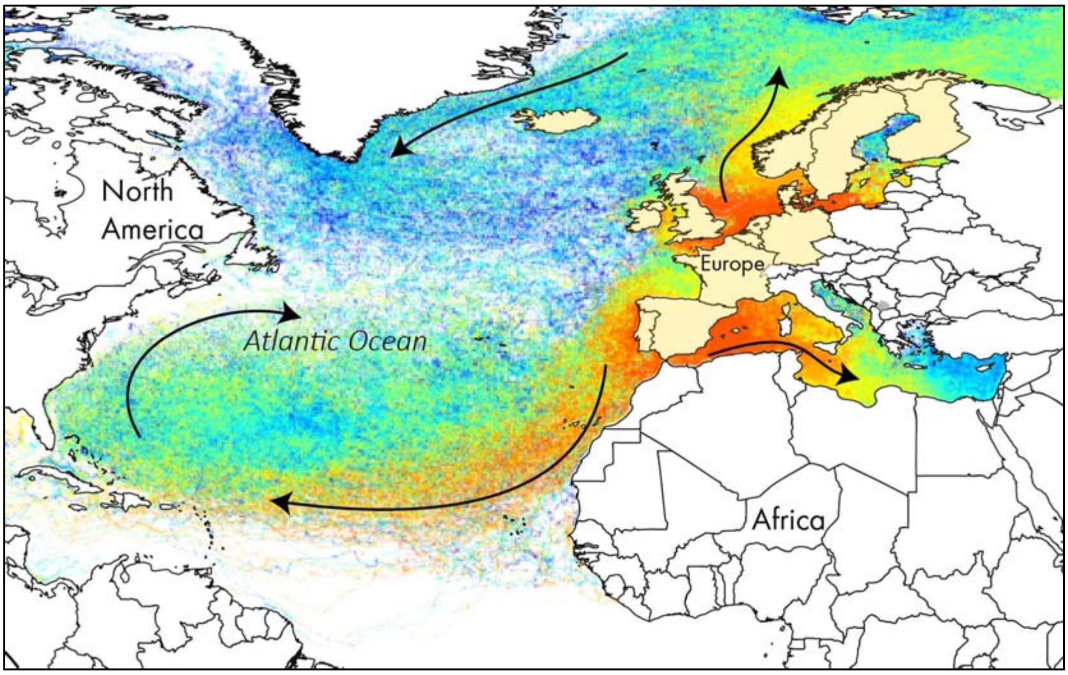


Figure 40. Frequency of particle visits as a percentage of total particle number for 1994–2014 (left) and minimum travel time for particles from Western and Northern Europe's coastline to the ocean (right)



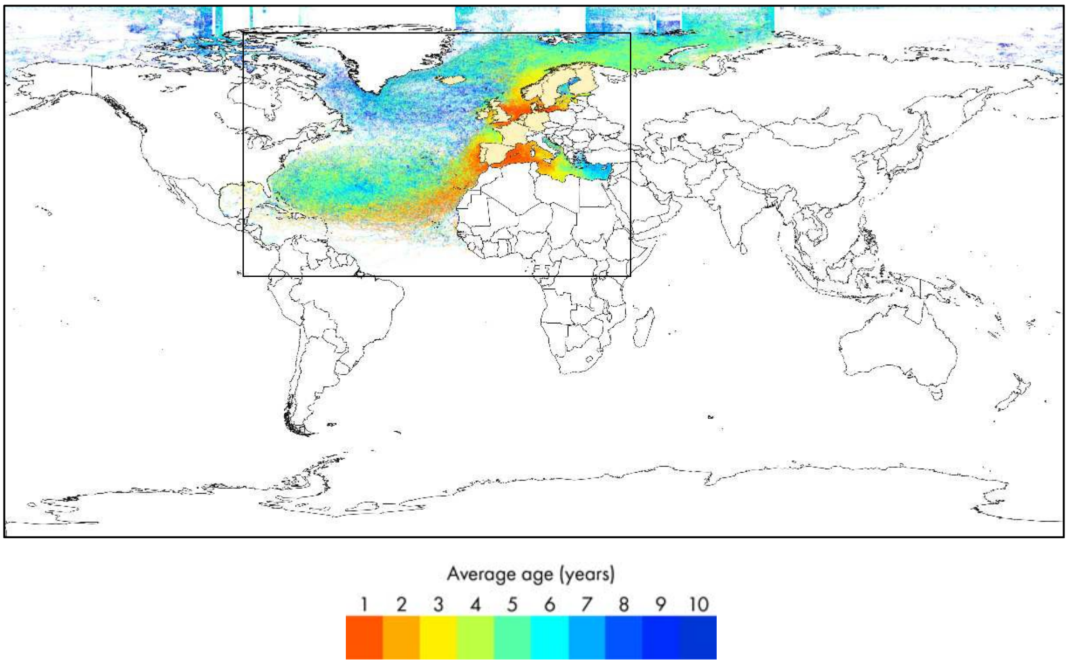


Figure 41. Average age in years of particles originating from Western Europe in the Atlantic Ocean (top) and globally (bottom), 1994–2014

7.12 The Mediterranean

The Mediterranean Sea has been described as one of the areas most affected by marine litter. Some of the largest quantities of municipal solid waste per person are generated annually in the region (208–760 kg/year).[10] The Mediterranean is particularly sensitive to debris accumulation because of urbanisation, tourism, shore use, important riverine inputs (from the Nile, Po, Rhone and Ebro rivers, because it carries 30% of the world's maritime traffic (UNEP, n.d), and because items less dense than the surrounding seawater cannot drift out at the Strait of Gibraltar. Semi-arid climates in the south, where annual rainfall is concentrated into just a few months, and the resultant spreading of litter during periods of intense rain, mean that river transport and uncontrolled discharges act as major sources of litter entering the marine environment.

Surveys of marine litter conducted to date show considerable spatial variability. However, the highest densities of marine litter stranded on the sea floor, sometimes reaching more than 100,000 items/km^2 (Galgani *et al.* 2000), and floating microplastics, reaching mean values of 1,050,000 particles/km^2 (maximum 4,680,000/km^2; Suaria *et al.* 2015), are in the southern Adriatic Sea. The two main sources of litter are land-based (up to 69%) and vessel-based (up to 26%) (Galgani et al. 2015).

The variability of surface circulation in the region is high, as instabilities occur in the basin. No global dataset on floating marine debris currently exists, although scenarios can be hypothesised to evaluate realistic distributions of litter. To date, only a few large sub-basins appear as potential retention areas, such as the north-western Mediterranean and the Tyrrhenian sub-basins, the

[10] Waste Atlas, http://www.atlas.d-waste.com/.

southern Adriatic and the Gulf of Syrt (Poulain *et al* 2012; Mansui *et al.* 2015). However, those regions lose their retentiveness for longer particle journeys, as no permanent gyres exist in the region (local sub-gyres typically persist for months, but seasonal and inter-annual variability alters water movements and litter distribution).

If the western Mediterranean coasts are regions of low impact, the southern coastal strip of the eastern Mediterranean basin appears to be a preferential beaching destination for marine litter (Figure 42). Litter stagnating along the Tunisian and Libyan coasts may result from accumulation in the Gulf of Syrt (Erikssen *et al.* 2014; Mansui *et al.* 2015). In contrast, the Levantine sub-basin appears to be a more local potential source for the nearby coast (Mansui *et al.* 2015).

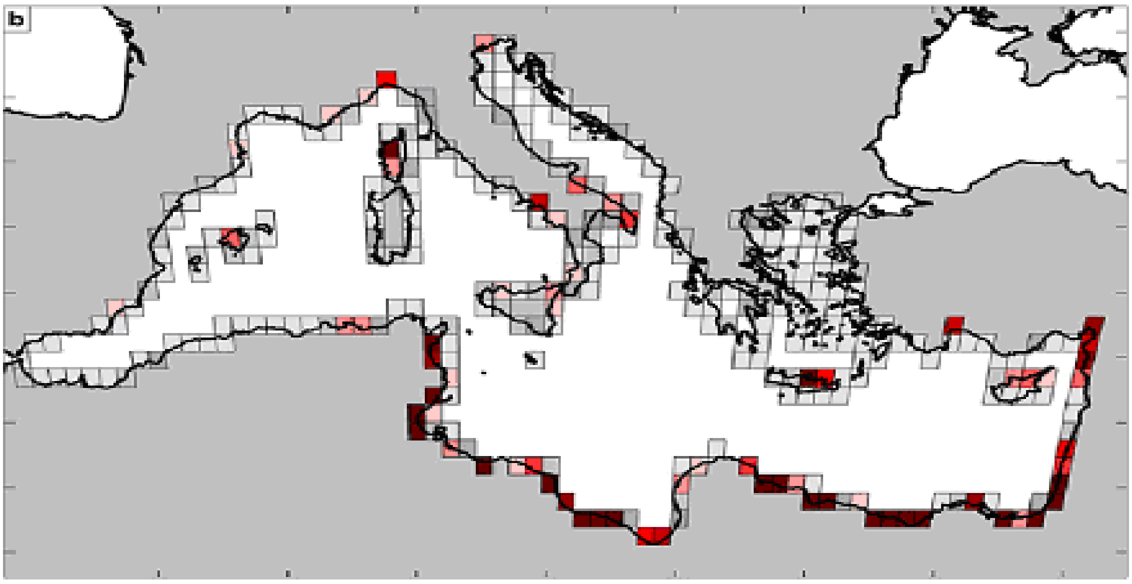


Figure 42. General predictive scheme of litter stranding on Mediterranean beaches

Source: Mansui *et al.* (2015).

8 Knowledge gaps, uncertainties and research priorities

This project has highlighted gaps and uncertainties in data on marine litter and in systems for monitoring and modelling its production, movement, accumulation and fate.

8.1 Litter sources

Recent global estimates of sources of marine debris rely on inference from statistics on waste production and waste management. However, there has been relatively little connection between the estimates of the sources, their relative magnitudes and sampling of the marine environment to validate or improve the estimates.

The large mismatch between estimates of litter inputs from land and the standing stock of plastics in the ocean (Jambeck *et al.* 2015; Cozar *et al.* 2014) raises many questions. A significant proportion of plastic (as much as 60% of production) cannot be accounted for as having been disposed of on land, and thus is likely to be lost into the environment (Kim *et al.* 2015). However, the standing stock may be overestimated, which would point to unaccounted for factors in the movement of litter between land-based sources and the ocean.

The transport of land-based solid waste into drainage systems during heavy rainfall and floods, and its impact on plastic transport to the ocean, remain relatively understudied. Similarly, it is very difficult to get data on the waste disposed of by vessels at sea, or on rates and volumes of losses of equipment and other debris at sea (Gilman 2015).

The modelled assessments of regions in Section 7 of this report rely heavily on the accuracy of the particle source distribution. In this case, a proxy based on coastal population density grids (Yetman *et al.* 2004) and estimates of mismanaged waste generation per country (Jambeck *et al.* 2015) were used. However, to fully understand litter generation, other source proxies should be used in parallel to reflect the contribution of specific sectors of human activity, such as fishing, aquaculture, tourism, shipping and transportation.

Riverine input, which was not considered in the assessments, should be represented in the future, but there is so far no estimate of global riverine inputs of macro- and microplastics into the ocean.

Well-designed studies are needed to quantify sea-based litter, particularly from aquaculture, including gear types, target species and impacts. Ideally, such studies would compare high and low aquaculture sites and would consider a number of variables, including infrastructure, local climate conditions, rates of production and consumption, and human attitudes.

Methods to monitor and estimate socioeconomic activities and resulting waste streams must be developed and implemented in targeted countries to allow a rigorous analysis of litter generation at the regional scale. Monitoring of coastal and urban centres could be particularly informative and may provide critical baseline data if and as infrastructure investment changes. Similarly to the work of Jambeck and colleagues (2015), further efforts are needed to assess the contribution of individual socioeconomic sectors at national and regional scales. This implies a need to build capacities in developing countries and to implement systematic and accurate monitoring of socioeconomic indicators.

There is a critical need for sampling the debris outflows from coastal regions using a statistically robust and repeatable sampling design that covers the coastal margin and the local offshore region. Value will be maximised if the sampling is statistically robust and coordinated across multiple sites.

The connection between the production, use and disposal of plastics on land and its loss into the ocean remains a key uncertainty. Critical questions in this area fall into two categories:

- Drivers for sources
  - How does government policy affect losses of plastics into the ocean?
  - What is the relative impact of facilities, incentives, social marketing and other types of interventions in relation to their cost?
  - Are there lessons from areas where plastics losses have been reduced, or have increased, that could be transferred to other regions?
  - How do drivers and dynamics differ across the range of economic development and income levels in different countries?
  - How can local changes in inputs be monitored cost-effectively to provide feedback to government and non-government actors on the scale of the problem and the impact of their actions?
- Source dynamics
  - What is the conceptual model for losses of plastic into the environment generally, and subsequently into marine systems?
  - What are the key processes determining the flows in that model?
  - How are the rates of input from land- and ocean-based sources changing with time?
  - Do changes in other variables, such as wealth, product design or economic activity, affect inputs of plastic into marine systems?
  - Are there predictable source dynamics that can help to identify hotspots for inputs or intervention?

In targeting research and investment in this area, it will be important to link activities to clear outcomes. For example, a potential outcome could be the design of cost-effective government policies to reduce inputs of plastic to marine systems. This target will determine the relative importance of the key uncertainties listed above, and may also suggest additional ones. Resolving the substantial mismatch between estimated litter inputs from land and the standing stock of plastics in the ocean requires a more nuanced understanding of the key processes and flows.

8.2 Fluxes

Understanding the relative magnitude of the various fluxes between oceanic compartments will be critical for estimating the load of plastics within compartments.

While transport processes in the open ocean, particularly in the surface layer, are relatively well understood, most of the other transport processes are not.

For example, large negatively buoyant items transported by extreme events have been well studied in some contexts, but the transport of debris less than 10 cm out from urban areas through waterways and into the ocean has not. There is a significant body of sediment transport research that could provide a basis for work in this topic, but little has been done to date.

A number of issues emerge from evaluating the uncertainty in our current estimates of the distribution of debris in marine systems. It will be critical to investigate further the three sources of uncertainty: data gaps, variation in sampling, and model parameterisation and mis-specification. Understanding the drivers of these sources of uncertainty is important, particularly as we attempt to translate sampling information into estimates of the sources, distribution and fate of debris.

An evaluation of the importance of coastal processes in determining debris sources, distribution and fate is clearly needed. The models that have been applied to predict debris distributions to date are largely focused on offshore processes, ignoring tides, freshwater outflows and coastal erosion and deposition. These omissions may lead to inaccuracies in the predictions and need to be evaluated, particularly as research moves towards investigating impacts on biota, which are primarily distributed in coastal habitats, and policy responses, which will necessarily be affected by coastal processes.

8.3 Changes in debris with time

Two aspects of plastic debris are understudied: its fragmentation and degradation, and the influence of those changes on its transport.

8.3.1 Fragmentation and degradation of plastic

Plastic debris slowly degrades and fragments, from large pieces into micro-sized and eventually nano-sized particles. Smaller particles have a larger surface area to volume ratio, decreasing the rate at which they sink. The surface area is also important for biofouling, which changes the density and hydrodynamics of plastic litter.

Fragmentation rates of marine plastic litter have been only roughly estimated, and there have been only rare attempts to determine loss of tensile strength or surface area (Andrady 2011; O'Brine & Thompson 2010). We do not know how long it takes under ambient marine environmental conditions for plastic to be broken down to nanoparticles and then mineralised into carbon dioxide and water. Recently, it was suggested that there appears to be a fast removal of plastic fragments smaller than 1 mm from surface water (Cozar *et al.* 2014). Hypothetical explanations for this observation include sampling and analytical artefacts, the selective ingestion of the size category by zooplankton, abrupt fragmentation of microplastics into nanoplastics, sinking due to biofouling, and high-speed mineralisation.

Model calculations from Deltares suggest that smaller plastic particles might indeed degrade and split into even smaller fragments at faster rates (Gerritse *et al.* 2015), but experimental evidence for this is needed. There is also some suggestion based on observational data that this fragmentation is driven by the beaching of larger material in the surf zone (Isobe *et al.* 2014).

New methods to measure plastics degradation and fragmentation have been tested in the European FP7 CleanSea Project (Gerritse *et al.* 2015). A marine mesocosm experiment in the laboratory containing a variety of conventional 'durable' and compostable plastic materials was set up to see whether electrical resistance measurements can be used to assess plastic degradation in seawater. First results indicate that the plastics in the mesocosm showed a decrease in electrical resistance over time, indicating polymer degradation, the absorption of seawater, or both. Further research is underway to determine whether such measurements can provide a simple, cheap and easy-to-use alternative method to determine degradation rates.

8.3.2 Influences of fragmentation on plastic litter movement in the marine environment

The quantities of small plastic fragments after fragmentation decrease exponentially into deeper layers (Kukulka *et al.* 2012; Reisser *et al.* 2015). The vertical distribution of the concentration (N) of such microplastics can be expressed as:

$N = N_0\, e^{wz/A0}$

where N_0 denotes the concentration of microplastics collected using a neuston net, w is the plastic rise velocity proportional to fragment sizes (⍰) as w [m/s] = 0.002×⍰⍰[mm] for hard plastics and sheets (Reisser *et al.* 2015), and z is the vertical axis looking upward from the sea surface. The dependency of the rise velocity on fragment sizes suggests that tiny microplastics with a large surface:volume ratio are likely to intrude into deeper layers by friction exerted on the plastic surface. The parameter A_0 is computed as:

$A_0 = 1.5\, u_*\, k\, H_S$,

where u_* represents the frictional velocity of water (=0.0012 W_{10}), k is the von Karman coefficient (0.4), Hs is the significant wave height, and W_{10} is the 10-m wind speed (Kukulka *et al.* 2012). Therefore, marine plastic debris after fragmentation is likely to sink into deeper layers as the fragmentation proceeds, especially under stormy (wavy) conditions.

The transport process in coastal waters favours the degradation of mesoplastics (Isobe *et al.* 2014). Field surveys, in conjunction with a numerical model, demonstrated the nearshore trapping of mesoplastics by a combination of Stokes drift onto beaches and high ascending velocities of relatively large fragments. The mesoplastics drifting close to the coast are likely to be washed ashore on beaches and easily returned to the ocean by tides and waves. This selective onshore transport of mesoplastics works persistently until they degrade on beaches into microplastics. Once mesoplastics degrade into fragments smaller than a few millimetres, they are free of nearshore trapping and thus able to spread offshore.

However, biofouling makes that motion more complicated. Biofouling can increase the density of small plastic fragments to the point where they sink. Buoyancy is related to volume, whereas fouling is related to surface area, so small items (which have high surface area to volume ratios) should start to sink sooner than large items (Fazey & Ryan 2015). In sampling small plastic fragments in the oceans, it was found that concentrations of tiny microplastics (<1 mm) decreased rapidly, while a similar rapid decrease in small-sized fragments never occurred for non-plastic particles (Cózar *et al.* 2014). This might suggest that a major destination of marine plastic debris is the deep ocean (Woodall *et al.* 2014; Ryan 2015).

Changes in particles over time, and particularly processes related to fragmentation and colonisation by marine life, are a key to understanding transport and deposition. If fragmentation processes could be understood, there is some possibility of using those processes in inverse modelling of sources and other key variables. Fragmentation is also very important for estimating potential impacts on marine ecosystems and wildlife.

There are currently several ongoing efforts to look at fragmentation processes, incorporating mechanical action, exposure to ultraviolet radiation and degradation by marine life (K. Law, pers. comm.; W. J. Shim pers. comm). So far, there have not been any publications in the marine debris literature focused on breakdown rates of materials, although there are a variety of observational data from debris surveys.

Key uncertainties about fragmentation include the rate of breakdown over time and the influence of exposure to oxidation, ultraviolet radiation and physical stresses in both open-water and coastal contexts, particularly when plastic is in contact with coastal sediments in the wave zone. These processes are almost certainly affected by polymer type, and may interact with colonisation by marine organisms.

It may be prudent to wait for preliminary results from current experimental studies before identifying priority actions in this area. One that can be identified at this point is to extend these experiments into field conditions, as the rates estimated in laboratory studies might not apply in the field.

8.4 Sinks

8.4.1 Coastlines

It is clear that there is a substantial load of plastic along the world's coastlines, but there has so far been little analysis of the role of the coast as a sink for plastic from the ocean. In particular, there is a significant challenge in parsing out terrestrial and marine sources of plastic along coastal margins. There is a clear pattern of fishing gear being more prevalent in remote locations, while consumer items and other non-fisheries waste are more common in more heavily populated coastal regions. While it is relatively clear that the fisheries component of this waste is marine in origin, the relative terrestrial and marine component of the non-fisheries waste is less clear.

The conclusion from the expert workshops was that the likely sink for the plastic missing from the input and standing stock budget is the coastline, and potentially coastal sediments. Understanding the deposition and resuspension process for plastic at coastal margins is probably the top priority for understanding and monitoring plastic in the ocean. In particular, it would be very useful to understand the flux rates and the interaction between debris characteristics (size, shape, density, buoyancy and so on) and deposition and resuspension. Similarly, it will be important to take into account the characteristics of the coastal region, including its geology, topography, vegetation, and other variables that will affect the balance between deposition and resuspension.

Tackling these uncertainties efficiently will require a mix of experimental work, inference from observational studies in the field and comparisons between model predictions and field observations. In order to control for the variation in these processes at local scales, it will be critical to implement studies at a large enough spatial scale and to use robust statistical designs and analyses to control for confounding factors.

8.4.2 Seabed and sediments

A recent review of the literature on plastics debris in the benthos attributed patterns of distribution to five major drivers: urban proximity, hydrology, geomorphology, vessel-based activities and river inputs (Corcoran 2015). While there are scattered reports of debris densities, there is so far no system-wide estimate of the plastic load in the benthos analogous to estimates for the ocean's surface waters. Plastic concentrations range widely, but can reach up to hundreds to thousands of items per square metre of benthic habitat.

Small-scale heterogeneity due to the five drivers listed above appears to produce much more complex patterns in the benthos than in surface waters and probably inhibits any reliable basin-scale estimates at this point.

Critically, many of the sampling methods used to date also introduce biases, due either to sampling location, size selectivity or response to material type, among other sources. Developing a synthetic dataset on plastics in the benthos will require not only robust sampling that can address the complexity of deposition rates and transport mechanisms, but also methods to address the bias in the various sampling methods used.

8.4.3 Water column

There has been relatively little sampling in the water column. However, the existing empirical studies document some plastic in the water column below the surface layer (Reisser *et al.* 2015; Desforges *et al.* 2014). A number of surface sampling efforts have noted the short-term effect of downward mixing by winds on plastic densities at the surface (e.g. Collignon *et al.* 2012). However, the most detailed survey of the water column to date (Reisser *et al.* 2015) suggests that the presence of debris in the water column is largely a result of downward mixing by wind and occurs during periods of strong wind. This implies that the water column is unlikely to be a major sink for plastic particles, as negative or positive buoyancy would be likely to drive the particles to the surface or benthos.

8.4.4 Biota

Marine organisms have been suggested as one of the potential reservoirs for plastics in the marine environment, particularly in the light of the mismatch between estimated inputs from land and the estimated standing stock of plastic in the ocean. While there have been a number of reviews of plastic interactions with species, most recently by Gall *et al.* (2015), there have been few efforts to document ingestion rates across a taxon, community or ecosystem.

A few recent studies have estimated debris loads globally for whole taxa, such as seabirds (Wilcox *et al.* 2015), marine turtles (Schuyler *et al.* 2015) and cetaceans (Baulch & Perry 2014). However, those studies were mainly focused on estimating exposure and potential ecological impacts, as opposed to the total load as a component of marine plastic. While the estimated loads in these taxa could be used as a component in an estimate of the total load of marine plastics in the biota, they cover only a tiny portion of the total number of taxa and the biomass that would need to be investigated in order to evaluate the role of marine biota as a sink for plastics.

8.5 Risks

The most critical research need in assessing the ecological risk from debris is to understand the demographic impacts of debris ingestion, and to a lesser extent entanglement. The information needed for estimating the rates of various types of interactions with plastic marine debris is generally available, at least to a first order. However, there is very little information on the demographic impact of those interactions, even for species with a long history of ingesting plastics.

8.6 Sampling

A number of regions are relatively poorly covered by the marine debris sampling effort. In particular, Asia and Africa are expected to make major contributions to the input of plastic into the ocean, but very little at-sea sampling is available from those regions (Jambeck *et al.* 2015; van Sebille *et al.* 2015). It will also be important to extend sampling into coastal regions. While most plastic debris is estimated to come from land, the land–sea interface remains substantially undersampled. This could readily be addressed, as coastal sampling is relatively low cost and logistically straightforward in comparison with sampling in offshore regions. However, to be effective it will require coordination among representatives from a number of participating countries.

The existence of fine-scale variations in debris density suggests that sampling will need to include additional information if at-sea samples are to provide useful estimates of the distribution of debris in the ocean. In particular, accounting for fine-scale wind-driven mixing will be likely to be critical. This can potentially be done using replicate sampling during at-sea surveys, matching replicates as closely in time and space as possible. This will allow the estimation of local variation, which will assist in separating true differences in debris densities in space and time from sampling variation driven by wind mixing and sampling conditions.

Sampling methods should be standardised, and should include coverage of affected and unaffected sites. If possible, sampling could be extended to measurements in the atmosphere, rivers, sandy beaches (surface and deep sampling) and sediments.

Further recommendations for sampling, as it relates directly to modelling, are in Section 6.4.

8.7 Monitoring

Workshop participants made specific recommendations to improve marine litter monitoring:

- Repeated sampling in consistent areas over a large geographical expanse, particularly in undersampled areas outside the North Pacific and North Atlantic, would significantly increase our understanding. While there are some large datasets on floating marine litter, for most regions there are no data for longer timeframes (such as 30 years or more).
- Watershed, coastal, sea-surface and sediment sampling could be set up as co-occurring or concurrent monitoring programs, which would ideally be carried out in different countries simultaneously and with shared methodologies.
- At the local scale, rubbish traps could be set up on waterways (up and downstream) to help identify sources of litter. Coupling such data collection strategies also has tremendous opportunities for outreach and community engagement activities. One recent example is the Baltimore Water Wheel, which also supports community outreach and engagement, although simpler approaches may be just as effective (WPB, n.d.).
- Citizen scientists can play an integral part in sampling and monitoring. They could include beachgoers, recreational sailors, scuba divers, school groups, corporate groups and other interested members of the public. Even something as simple as asking people to weigh or count litter collected from cleaning activities or fishing-for-litter programs would help to fill a critical knowledge gap.

8.8 Modelling

Section 6.4 describes uncertainties in modelling in technical detail and suggests ways of overcoming or accounting for them.

In addition, the two expert modelling workshops concurred on other suggestions:

- Improvements are needed in the representation of plastic biogeochemical processes in models.
- We should standardise modelling techniques, including for time and space resolutions, perhaps using particular sites with detailed information to inform particular models.
- Models need calibration and model results need validation.
- Inverse Lagrangian models can detect potential sources of plastics by using hindcasting to see where things come from. To appreciate stochastic processes, a main point of consideration is not to be deterministic.
- Modellers should integrate the expertise of several other scientific areas (such as the domains of ecologists, clinicians, chemists, ecotoxicologists and others).
- Using proxies for areas in which data are lacking can improve model solutions but has not been done widely. Further exploration of the use of proxies in combination with statistical and process models (particularly in considering missing data) will undoubtedly prove useful.

8.9 Recording and communicating

It would be useful to catalogue historical, recent and future releases of plastic to the environment and other relevant data in a single database that is mirrored in multiple sites. The database could hold a comprehensive dataset on plastics at the coastal margin on land and at sea near and away from urban centres for countries identified in recent literature (Jambeck *et al.* 2015) as making significant inputs to the marine environment.

To date, there has been a lack of standardised reporting. Consistency could be achieved via a website with freely available open-source methodology and datasheets. Links to other research projects applying particular methodologies would also improve communication. Better reporting would increase our ability to compare types, sources and quantities around the globe.

Empirical data and statistical models could be combined to produce maps of plastic plumes generated from urban centres (including covariates such as population density, infrastructure and other potentially important correlates). This material would be shared publicly through visual media, reportage and social media streams.

A global library or repository could publicise contributors, initiatives, contacts, sources, data holders and results. It would not necessarily hold all the data, but would be a place where information, approaches, methodologies and other information would be shared on an open-source platform. It could also include upcoming events, conferences and information-sharing opportunities. Ideally, would be hosted by UNEP to enhance its authority and to engage more people to contribute. Sustained commitment to such a resource would be critical.

At the regional level, we should identify charismatic or iconic fauna to be state-of-the-environment or community target taxa. This may include threatened and endangered species.

8.10 Other knowledge gaps and opportunities

Our research and the participants in the modelling workshops suggested a number of other areas of relevant research, which are described in this section.

8.10.1 Climate change

Few studies have considered the interaction between climate change and plastics. Ocean currents, migration routes and species distributions are changing, so understanding the interaction between climate and plastics may be particularly relevant for understanding impacts to biodiversity. Modelling that specifically addresses plastics movement between compartments as temperatures and associated processes change will improve our ability to predict risks to wildlife. Would more buoyant plastic polymers occur at the surface due to vertical differentiation? If so, would this result in differential availability of plastic to surface-feeding species?

8.10.2 The food chain

Evaluating the effects of plastic contamination on the marine food chain and environment is difficult but necessary. A combination of modelling and experimental approaches (including meso-or microcosm experiments) could provide needed data on growth, mortality and reproductive output. Dynamic energy budget modelling can be used to look at the effect of productivity on trophic levels of the food chain.

8.10.3 Other pollutants

We still know relatively little about the impacts of pollutant concentrations in and on plastics and their effects on marine biota. Plastics can contain, accumulate and carry pollutants, inserted as additives or absorbed from the environment, that may act as soon as they are delivered to organisms. Additional experiments to evaluate pollutant assimilation, accumulation and transport between tissues are needed to more fully quantify ecological risk at the individual, population and species levels.

8.10.4 Economics

Economic losses from plastics pollution to tourism, fishing, diving and other industries could be quantified in projects at paired sites. This would also provide opportunities for outreach and campaigns to increase awareness of the local economic costs of littering.

8.10.5 Creative uses of data, events and experiments

Environmental accidents and extreme weather can be used to train or improve our models, as they provide opportunities for large-scale ‘natural’ experiments, and creative thinking suggests other possibilities. For example, combining oceanic plastics movement models with shipping data and fishing effort data could allow us to better estimate or quantify at-sea losses into the ocean. Community-level surveys on waste management, flows and loss rates from coastal communities can be applied to tune models.

Our research would also benefit from:

- laboratory experiments, particularly experiments that focus on plastics fragmentation rates
- laboratory-based or *in situ* experiments to look at plastics sinking rates
- field particle-tracking experiments to improve model fits for geostrophic currents, Stokes drift, wind waves, windage and water drag
- toxicological impacts experiments to evaluate risks and impacts to biota
- experiments to quantify ingestion, filtering and transport from biota to the other oceanic compartments
- field experiments to document atmospheric deposition
- experiments using biomarkers as indicators of toxic effects
- experiments to measure contamination on sandy beaches by persistent organic pollutants and heavy metals due to plastic dispersion.

8.11 Cross-cutting issues

This work identified a number of questions that cross disciplinary and epistemological boundaries. They are discussed in this section.

8.11.1 The coast–ocean–coast flux

The largest flux identified in our expert workshops was the coast–ocean–coast (C-O-C) flux from the nearshore environment into the ocean and back. The C-O-C flux is one of the most important issues in understanding the sources, dynamics and impacts to the marine debris issue and devising policy responses.

The mechanisms of this flux are complicated by dynamic forcing from winds, tides and wave action, along with heterogeneity in coastal landforms, currents and plastic inputs. This is one of the areas where our uncertainty is substantial. The oceanographic models used for estimating the distribution of debris on the ocean surface are relatively inaccurate in coastal regions, as they do not generally incorporate phenomena such as tides, coastal geological features and other factors that critically influence deposition and suspension in those regions.

This is a key research area and should be prioritised above the other fluxes and transport processes. We could use existing transport models to predict C-O-C fluxes and then compare the predictions against empirical data on at-sea distributions of plastic and plastic densities on the coastline.

8.11.2 Study design

For further sound research on marine litter, we need standardised methods for sampling the various oceanic compartments, analytical tools for investigating patterns in the samples, and models for projecting dynamics of plastics in the environment and their impacts on wildlife. A number of studies that have tried to compile data to draw inferences at the global scale have noted the problems arising from poor study designs, studies that measure dissimilar responses or work that is incompletely reported.

While inference is still possible in these cases, it is harder when variables available in high-quality studies have to be disregarded to allow data to be combined with data from less well structured or less detailed studies.

Investment in the development of standardised coastal survey methods, supported by a freely available database structure and statistical tools, could substantially improve the quality of data available at the global scale for drawing inferences about loads, trends and dynamics. Our use of standardised methods developed by plankton researchers over many years is one of the main reasons why our understanding of the standing stock of debris in the ocean surface is so much better than our knowledge of the stock in the other oceanic compartments.

8.11.3 Integrating sources, transport and deposition in coastal regions

We need a better understanding of the dynamics in the coastal region, particularly the link between coastal litter sources, transport and deposition. Those processes govern the balance between inputs and deposition in what is suspected to be the primary sink for plastic. In addition, debris in coastal and continental shelf regions is likely to have the greatest ecological impact because of those regions' high biodiversity, and the greatest economic impact because of interactions with tourism, fisheries, transport and solid-waste management industries.

The most efficient approach is likely to be tackling this system in an integrated fashion, using focal areas to understand sources and drivers, locally relevant transport models to represent movement of plastics, and statistically robust sampling in coastal regions to look at deposition and resuspension. There have been some efforts in this area, including recent work by Isobe *et al.* (2015) along the Japanese coastline and Hardesty *et al.* (in press) in Australia. However, there remains a need for a study that can link land-based sources, transport to the marine system and along continental shelves and coastal margins, and deposition and resuspension from coastlines.

This study would need to integrate:

- land-based sampling of wastes and loss rates to the environment
- flow modelling for wind, water and human transport to the marine system
- modelling of transport processes in the marine system, particularly at coastal margins and including deposition and resuspension
- robust sampling and analysis of debris in the coastal zone.

This effort would require higher resolution numerical models than have been used for transport to date, along with investigations of transport processes on land, and the connection of both sets of transport processes to system-scale sampling in the terrestrial system. It is likely to be most productive if conducted using a limited number of case study regions, covering different socioeconomic aspects and physical systems.

Appendix 1: Ocean circulation models and oceanographic datasets used for marine debris modelling and particle tracking

Table 5. Available datasets containing environmental drivers for off-line trajectory models.

Dataset	Description	Type[1]	Environmental drivers	Spatial resolution / domain Temporal resolution / range	Limitations	Availability
OSCAR	Based on satellite sea level and surface winds	EE	Surface currents	1/3°, global 5-day, 1992–present	Satellite-based; limited near coastlines	NOAA, NASA PO.DAAC
Argo	Autonomous profiling floats that drift in the ocean	O/EE	Surface currents	1°, global Monthly, 2003–present	Based on surface drift; marginal seas/southern ocean gaps; nearshore limitations	APDRC IFremer
Surface Drifters	NOAA's Global Drifter Program maintains about 1,000 surface drifters throughout the globe	O/EE	Surface currents	½°, global Monthly climatology	Based on drouged drifting buoys; somewhat sparse coverage	NOAA
AVISO	Satellite-measured sea surface height	EE	Surface currents	¼°, global Daily, 1993–2014		
SCUD	Diagnostic model based on satellite winds and sea level	EE	Surface currents	¼°, global/Pacific Daily, 1999–2009 (global) 1999-present (Pacific)		APDRC
SODA	Long-integration of data-assimilating model	M	Surface currents, surface winds	½°, global Monthly 1871–2010		APDRC SODA/TAMU SODA/UMD
ECMWF	Ocean reanalysis (ORA-S3, ORA-S4)	M	Surface currents, winds	1°, global Monthly 1958–2014		APDRC ECMWF
WW3	Operational wave forecast	M	Surface winds, surface waves	1°, global Hourly, weekly (hindcast and forecast)		NCEP
SWAN	Operational wave forecast	M	Surface waves	See website		DELFT
GFS	Operational weather forecast	M	Surface winds	¼°, global Hourly for past month		NCEP
NCOM	Operational data-assimilating ocean model	M	Surface currents, surface winds	1/8°, global Daily, 2003–2013		

NLOM	Operational data-assimilating model	M	Surface currents, surface winds	1/16°, global Daily 2002–2006 1/32°, global Daily 2005–2013		
HYCOM	Operational data-assimilating ocean model	M	Surface currents, surface winds	1/12°, global Daily, 2009–present		
BlueLink	Operational data-assimilating ocean model	M	Surface currents, surface winds	1/10° (variable), regional (90–180, to 20 N) Daily, weekly forecast		
OFES	Hindcast ocean model	M	Surface currents, surface winds	1/10°, global Daily/monthly, 1950–2011		
IPCC	Coupled climate models	M	Surface currents, surface winds	Typically 1° global Monthly output from decadal runs		
ECCO	Data-assimilating GODAE-era model	M	Surface currents, surface winds			
ROMS	High-resolution regional operational model	M	Surface currents, surface winds	Variable resolution, different regional implementation (e.g. IOOS)		
SLIM		M				
Delft-3D FLOW	Hydrodynamic nearshore model	M	Produces 2D or 3D dynamic flow fields	Is generic software; spatial and temporal resolution depends on specific implementation; can be very flexible		Deltares, open source http://oss.deltares.nl/web/delft3d
Delft-3D WAQ	Sediment transport and water quality software	M	Sediment and water quality substances	Is generic software; spatial and temporal resolution depends on specific implementation; can be very flexible		Deltares, open source http://oss.deltares.nl/web/delft3d
Delft-3D BLOOM	Phytoplankton model	M	Algae concentrations, limiting factors (nutrients, light)	Is generic software; spatial and temporal resolution depends on specific implementation; very flexible		Deltares, open source http://oss.deltares.nl/web/delft3d
Delft-3D PART	Particle tracking software	M	Trajectories of particles	Is generic software; spatial and temporal resolution depends on specific implementation; can be very flexible		Deltares, open source http://oss.deltares.nl/web/delft3d

SIMONA	2D/3D hydrodynamic software	M	Produces 2D or 3D dynamic flow fields	Is generic software; spatial and temporal resolution depends on specific implementation; can be very flexible	http://simona.deltares.nl/
WFLOW	Distributed hydrological (catchment) software	M	1D flow velocity, currents, discharge	Is generic software; spatial and temporal resolution depends on specific implementation; can be very flexible	Deltares https://publicwiki.deltares.nl/display/OpenS/WFlow+rainfall-runoff+model
SOBEK	1D hydrology software	M	1D flow velocity, currents, discharge	Is generic software; spatial and temporal resolution depends on specific implementation; can be very flexible	Deltares https://www.deltares.nl/en/software/sobek/
DFLOW-Flexible MESH	1D/2D/3D hydrodynamic software	M	Produces 1D/2D/3D dynamic flow fields	Is generic software; spatial and temporal resolution depends on specific implementation; can be very flexible	http://oss.deltares.nl/web/delft3d/d-flow-flexible-mesh

a Can be observation (O), empirical estimate based on observations (EE) or dynamical simulation (M).

Table 6. Some of the available particle tracking models, with information on their capability and limitations

Trajectory code	Description	Highlights	Limitations	Availability
Arianne				
trackmass	U. Stockholm			
CMS	Particle tracking code	Written in FORTRAN; source code available; memory efficient; relatively fast; includes forward and backward trajectories; sources/sinks can be digitally available; can read via OPeNDAP		
GNOME	NOAA oil spill response and restoration model; users can input source/sink and forcing fields and see advection in real time	Easy to use; includes terms for uncertainties, weathering, amount of spill; can specify forcing; forward/backward tracking	Digital output (e.g. data files) are difficult (screen output only); source code not available; simulations can be limited by system memory (need to load entire forcing fields)	NOAA ORR
Pol3DD				
OSCURS	Old NOAA web application	Easy to use	Very limited for long integrations; web application, so digital I/O not possible	
PELLET-2D				
Connie2		User friendly	Updated through 2014	Open source: http://www.csiro.au/connie2/

Adrift.org.au	User friendly; educational tool	Large-scale resolution	Open source: www.adrift.org.au

Appendix 2: Participants in modelling workshops

Table 7. Participants in first modelling workshop, April 2015

Name and title	Affiliation	e-mail
Dr Alexander Turra	Oceanographic Institute, São Paulo University	turra@usp.br
Dr Denise Hardesty	CSIRO	Denise.hardesty@csiro.au
Dr Erik van Sebille	Imperial College London	E.van-Sebille@imperial.ac.uk
Dr James Potemra	University of Hawaii	jimp@hawaii.edu
Dr Peter Kershaw	Independent adviser—marine environmental protection	peter@pjkershaw.com
Mr Laurent Lebreton	Dumpark Ltd	laurent@dumpark.com
Prof Dick Vethaak	Deltares and VU University Amsterdam	dick.vethaak@deltares.nl
Mr Luis Valdes	IOC–UNESCO	Jl.valdes@unesco.org
Ms Heidi Savelli	UNEP	Heidi.savelli@unep.org

Table 8. Participants in second modelling workshop, August–September 2015

Name and title	Affiliation	e-mail
Dr Atsuhiko Isobe	Research Institute for Applied Mechanics, Kyushu University	isobeatsuhiko@icloudcom
Dr Joseph Harari	Oceanographic Institute, São Paulo University	joharari@uspbr
Dr Denise Hardesty	CSIRO	Denisehardesty@csiroau
Dr Kara Lavender-Law[a]	Sea Education Association	klavender@seaedu
Mr Laurent Lebreton	Dumpark Ltd	laurent@dumparkcom
Dr Nikolai Maximenko	University of Hawaii	maximenk@hawaiiedu
Dr James Potemra	University of Hawaii	jimp@hawaiiedu
Dr Erik van Sebille	Imperial College London	Evan-Sebille@imperialacuk
Prof Dick Vethaak	Deltares and VU University Amsterdam	dickvethaak@deltaresnl
Dr Chris Wilcox	CSIRO	chriswilcox@csiroau
Ms Heidi Savelli	UNEP	Heidisavelli@uneporg

a Remote participation via Skype.

References

Acampora H., Schuyler Q., Townsend K., Hardesty B. D. (2013). Comparing plastic ingestion between juvenile and adult stranded short-tailed shearwaters (*Puffinus tenuirostris*) in Eastern Australia, *Marine Pollution Bulletin*, http://doi.org/10.1016/j.marpolbul.2013.11.009

Ainley D. G., Spear L. B. Ribic, C. A. (1990). The incidence of plastic in the diets of pelagic seabirds in the eastern equatorial Pacific region, in *Proceedings of the second international conference on marine debris*, 653–664.

Allen D., Arthur S., Wallerstien N. *et al.* (2015). Provision, transport and deposition of debris in urban waterways, *International Journal of Sediment Research*, *30*(2), 142–149.

Andrady A. L. (2011). Microplastics in the marine environment, *Marine Pollution Bulletin*, *62*, 1596–1605. http://dx.doi.org/10.1016/j.marpolbul.2011.05.030.

Angiolillo M., di Lorenzo B., Farcomeni A., Bo M., Bavestrello G., Santangelo G., Cau A., Mastascusa V., Cau A., Sacco F., Canese S. (2015). Distribution and assessment of marine debris in the deep Tyrrhenian Sea (NW Mediterranean Sea, Italy), *Marine Pollution Bulletin*, *92*, 149–159.

Armitage N., Rooseboom A. (2000). The removal of urban litter from stormwater conduits and streams: Paper 1–The quantities involved and catchment litter management options, *Water SA*, *26*(2), 181–187.

Ayaz, A., Ünal, V., Acarli, D., Altinagac, U. (2010). Fishing gear losses in the Gökova Special Environmental Protection Area (SEPA), eastern Mediterranean, Turkey. *Journal of Applied Ichthyology*, *26*(3), 416-419.

Bagulayan A., Bartlett-Roa J., Carter A., Inman B., Keen E., Orenstein E., Patin N., Sato K., Sibert E., Simonis A., Van Cise A., Franks P. (2012). Journey to the center of the gyre: the fate of the Tohoku tsunami debris field, *Oceanography–Oceanography Society*, *25*, 200–207

Baird R. W., Hooker S. K. (2000). Ingestion of plastic and unusual prey by a juvenile harbour porpoise, *Marine Pollution Bulletin*, *40*(8), 719–720.

Baird J., Curry R., Cruz P. (2014). An overview of waste crime, its characteristics, and the vulnerability of the EU waste sector, *Waste Management & Research*, *32*(2), 97–105.

Barnes D. (2002). Biodiversity: invasions by marine life on plastic debris, *Nature*, *416*, 808–809.

Barnes D. K., Galgani F., Thompson R. C., Barlaz M. (2009). Accumulation and fragmentation of plastic debris in global environments, *Philosophical Transactions of the Royal Society B: Biological Sciences*, *364*(1526), 1985–1998.

Barnes, D. K., Walters, A., & Gonçalves, L. (2010). Macroplastics at sea around Antarctica. *Marine Environmental Research*, *70*(2), 250-252.

Barrett, R. T., Camphuysen, K. C., Anker-Nilssen, T., Chardine, J. W., Furness, R. W., Garthe, S., ... & Veit, R. R. (2007). Diet studies of seabirds: a review and recommendations. *ICES Journal of Marine Science: Journal du Conseil*, *64*(9), 1675-1691.

Baulch, S., & Perry, C. (2014). Evaluating the impacts of marine debris on cetaceans. *Marine pollution bulletin*, *80*(1), 210-221.

Bergmann M., Klages M. (2012). Increase of litter at the Arctic deep-sea observatory Hausgarten, *Marine Pollution Bulletin*, *64* ,2734–2741, doi:10.1016/j.marpolbul.2012.09.018.

Boerger, C. M., Lattin, G. L., Moore, S. L., & Moore, C. J. (2010). Plastic ingestion by planktivorous fishes in the North Pacific Central Gyre. *Marine pollution bulletin*, *60*(12), 2275-2278.

Browne M. A., Crump P., Niven S. J., Teuten E., Tonkin A., Galloway T., Thompson R. (2011). Accumulation of microplastic on shorelines worldwide: sources and sinks, *Environmental Science & Technology*, *45*, 9175–9179.

Brunner, K., Kukulka, T., Proskurowski, G., & Law, K. L. (2015). Passive buoyant tracers in the ocean surface boundary layer: 2. Observations and simulations of microplastic marine debris. *Journal of Geophysical Research: Oceans*, *120*(11), 7559-7573.

Burnham K. P., Anderson D. R. (2002). *Model selection and multimodel inference: a practical information-theoretic approach*, 2nd edition, Springer–Verlag.

G.C. Cadée (2002). Seabirds and floating plastic debris. Marine Pollution Bulletin, 44 (2002), pp. 1294–1295.

Calder D., Choong H., Carlton J., Chapman J., Miller J., Geller J. (2014). Hydroids (Cnidaria: Hydrozoa) from Japanese tsunami marine debris washing ashore in the northwestern United States, *Aquatic Invasions*, *9*, 425–440.

Carey M. J. (2011). Intergenerational transfer of plastic debris by short-tailed shearwaters (*Ardenna tenuirostris*), *Emu*, *111*(3), 229–234.

Carson H. S. (2013). The incidence of plastic ingestion by fishes: from the prey's perspective, *Marine Pollution Bulletin*, *74*, 170–174, http://dx.doi.org/10.1016/j.marpolbul.2013.07.008.

Carson H. S., Lamson M. R., Nakashima D. *et al.* (2013). Tracking the sources and sinks of local marine debris in Hawaii, *Marine Environmental Research*, *84*, 76–83.

Cho D.-O. (2011). Removing derelict fishing gear from the deep seabed of the East Sea, *Marine Policy*, *35*(5), 610–614.

Claessens M., De Meester S., Van Landuyt L., De Clerck K., Janssen C.R. (2011). Occurrence and distribution of microplastics in marine sediments along the Belgian coast, *Marine Pollution Bulletin*, *62*, 2199–2204, http://dx.doi.org/10.1016/ j.marpolbul.2011.06.030.

Cole M., Lindeque P., Halsband C., & Galloway T. S. (2011). Microplastics as contaminants in the marine environment: a review, *Marine Pollution Bulletin*, *62*(12), 2588–2597.

Collignon A., Hecq J. H., Glagani F., Voisin P., Collard F., & Goffart A. (2012). Neustonic microplastic and zooplankton in the north western Mediterranean Sea, *Marine Pollution Bulletin*, *64*(4), 861–864.

Convey P., Barnes D., Morton A., (2002). Debris accumulation on oceanic island shores of the Scotia Arc, Antarctica, *Polar Biology*, 25, 612–617.

Corcoran P. L. (2015). Benthic plastic debris in marine and freshwater environments, *Environmental Science. Processes & Impacts*, *17*(8), 1363–1369.

Costa, M. F., Silva-Cavalcanti, J. S., Barbosa, C. C., Portugal, J. L., & Barletta, M. (2011). Plastics buried in the inter-tidal plain of a tropical estuarine ecosystem. *Journal of Coastal Research*, (64), 339.

Cozar A., Echevarria F., Gonzalez-Gordillo J. I., Irigoien X., Úbeda B., Hernandez- Leon S., Palma A. T., Navarro S., García-de-Lomas J., Ruiz A., Fernandez-de-Puelles M. L., Duarte C. M. (2014). Plastic debris in the open ocean, *Proceedings of the National Academy of Sciences of the United States of America*, *111*, 10239e10244. http://dx.doi.org/10.1073/pnas. 1314705111.

Cummings J.A. (2005). Operational multivariate ocean data assimilation, *Quarterly Journal of the Royal Meteorological Society*, Part C, *131*(613), 3583–3604.

Cummings J. A., Smedstad O. M. (2013). Variational data assimilation for the global ocean, in Park S. K., Xu L. (eds), *Data assimilation for atmospheric, oceanic and hydrologic applications*, vol. II, chapter 13, Springer–Verlag, 303–343.

Dangi, M. B., Urynowicz, M. a., & Belbase, S. (2013). Characterization, generation, and management of household solid waste in Tulsipur, Nepal. *Habitat International*, *40*, 65–72. http://doi.org/10.1016/j.habitatint.2013.02.005

de Stephanis R., Giménez J., Carpinelli E., Gutierrez-Exposito C., Cañadas A. (2013). As main meal for sperm whales: plastics debris, *Marine Pollution Bulletin*, *69*(1), 206–214.

DEE (Department of the Environment and Energy) 2009. *Threat abatement plan for the impacts of marine debris on vertebrate marine life*, DEE, Canberra, https://www.environment.gov.au/marine/publications/threat-abatement-plan-impacts-marine-debris-vertebrate-marine-life.

Dee D. P., Uppala S. M., Simmons A. J., Berrisford P., Poli P., Kobayashi S., Andrae U., Balmaseda M.A., Balsamo G., Bauer P., Bechtold P., Beljaars A. C. M., van de Berg L., Bidlot J., Bormann N., Delsol C., Dragani R., Fuentes M., Geer A. J., Haimberger L., Healy S. B., Hersbach H., Holm E. V., Isaksen L., Kallberg P., Koehler M., Matricardi M., McNally A. P., Monge-Sanz B. M., Morcrette J. J., Park B. K., Peubey C., de Rosnay P., Tavolato C., Thepaut J. N., Vitart F. (2011). The ERA-Interim reanalysis: configuration and performance of the data assimilation system, *Quarterly Journal of the Royal Meteorological Society*, *137*, 553–597.

Derraik J. G. (2002). The pollution of the marine environment by plastic debris: a review, *Marine Pollution Bulletin*, *44*(9), 842–852.

Desforges J.-P., Galbraith M., Ross P. S. (2015). Ingestion of microplastics by zooplankton in the northeast Pacific Ocean, *Archives of Environmental Contamination and Toxicology*, 10.1007/s00244-015-0172-5.

Dixon, T.R., Dixon, T.J. (1981). Marine litter surveillance, Marine Pollution Bulletin, 9, 289-295.

EC (European Commission) (2008). *Marine Strategy Framework Directive*, EC, Brussels, http://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:32008L0056.

Eriksson C., Burton H. (2003). Origins and biological accumulation of small plastic particles in fur seals from Macquarie Island, *A Journal of the Human Environment*, *32*, 380–384.

Eriksen M., Maximenko N., Thiel M., Cummins A., Lattin G., Wilson S., Rifman S. (2013a). Plastic pollution in the South Pacific subtropical gyre, *Marine Pollution Bulletin*, *68*(1), 71–76.

Erikssen M., Mason S., Wilson S., Box C., Zellers A., Edwards W., Amato S. (2013b). Microplastic pollution in the surface waters of the Laurentian Great Lakes, *Marine Pollution Bulletin*, *77*(1), 177–182.

Erikssen M., Lebreton L. C. M., Carson H. S., Thiel M., Moore C. J., Borerro J. C., Galgani F., Ryan P. G., Reisser J. (2014). Plastic pollution in the world's oceans: more than 5 trillion plastic pieces weighing over 250,000 tons afloat at sea, *PLoS ONE*, *9*(12): e111913, doi:10.1371/journal.pone.0111913.

Fazey, F. M., & Ryan, P. G. (2016). Biofouling on buoyant marine plastics: An experimental study into the effect of size on surface longevity. *Environmental Pollution*, *210*, 354-360.

Fischer, V., Elsner, N. O., Brenke, N., Schwabe, E., & Brandt, A. (2015). Plastic pollution of the Kuril–Kamchatka Trench area (NW pacific). *Deep Sea Research Part II: Topical Studies in Oceanography*, *111*, 399-405.

Fossi M. C., Panti C., Guerranti C., Coppola D., Giannetti M., Marsili L., Minutoli R. (2012). Are baleen whales exposed to the threat of microplastics? A case study of the Mediterranean fin whale (*Balaenoptera physalus*), *Marine Pollution Bulletin*, *64*(11), 2374–2379.

Fox D. N., Teague W. J., Barron C. N., Carnes M. R., Lee C. M. (2002). The Modular Ocean Data Assimilation System (MODAS), *Journal of Atmospheric and Oceanic Technology*, *19*, 240–252.

Frost A., Cullen M. (1997). Marine debris on northern New South Wales beaches (Australia): sources and the role of beach usage, *Marine Pollution Bulletin*, *34*(5), 348–352.

Gabrielides G. P., Golik A., Loizides L., Marino M. G., Bingel F., Torregrossa M. V. (1991). Man-made garbage pollution on the Mediterranean coastline, *Marine Pollution Bulletin*, *23*, 437–441.

Galgani F. (2015). Marine litter: future prospects for research, *Frontiers in Marine Science*, 26 October, http://journal.frontiersin.org/article/10.3389/fmars.2015.00087/full

Galgani, F., Hanke, G., & Maes, T. (2015). Global distribution, composition and abundance of marine litter. In *Marine anthropogenic litter* (pp. 29-56). Springer International Publishing.

Galgani F., Andral B. (1998). Methods for evaluating debris on the deep sea floor, *Oceans 1998 Conference Proceedings*.

Galgani F., Lecornu F. (2004). Debris on the sea floor at 'Hausgarten': in the expedition ARKTIS XIX/3 of the research vessel POLARSTERN in 2003, *Berichte Polar Meeresforsch*, *488*, 260–262.

Galgani F., Jaunet S., Campillo A., Guenegen X., His E. (1995). Distribution and abundance of debris on the continental-shelf of the north-western Mediterranean-sea, *Marine Pollution Bulletin*, *30*, 713–717.

Galgani F., Souplet A., Cadiou Y. (1996). Accumulation of debris on the deep sea floor off the French Mediterranean coast, *Marine Ecology Progress Series*, *142*, 225–234.

Galgani F., Leaute J. P., Moguedet P., Souplet A., Verin Y., Carpentier A., and Mahe J. C. (2000). Litter on the sea floor along European coasts, *Marine Pollution Bulletin*, *40*(6), 516–527.

Gall S. C., Thompson R. C. (2015). The impact of debris on marine life, *Marine Pollution Bulletin*, *92*, 170–179, doi: 10.1016/j.marpolbul.2014.12.041.

Gerritse J., Leslie H., Vethaak D. (2015). *Fragmentation of plastic litter in the marine environment: our plastic-littered seas and how they transition from 'extra chunky' soup to a plastic 'bouillon'*, CleanSea Project special newsletter, in press.

Gilardi K. V. K., Carlson-Bremer D., June J. A., Antonelis K., Broadhurst G., Cowan T. (2010). Marine species mortality in derelict fishing nets in Puget Sound, WA and the cost/benefits of derelict net removal, *Marine Pollution Bulletin*, *60*, 376–382.

Gilman, E. (2015). Status of international monitoring and management of abandoned, lost and discarded fishing gear and ghost fishing. *Marine Policy*, *60*, 225-239.

Glanville K., Chang H.-C. (2015). Mapping illegal domestic waste disposal potential to support waste management efforts in Queensland, Australia, *International Journal of Geographical Information Science*, *29*(6), 1042–1058.

Goldstein, M. C., Rosenberg, M., & Cheng, L. (2012). Increased oceanic microplastic debris enhances oviposition in an endemic pelagic insect. *Biology letters*, *8*(5), 817-820.

Graham, E. R., & Thompson, J. T. (2009). Deposit-and suspension-feeding sea cucumbers (Echinodermata) ingest plastic fragments. *Journal of Experimental Marine Biology and Ecology*, *368*(1), 22-29.

Guneroglu A. (2010). Marine litter transportation and composition in the coastal southern Black Sea region, *Scientific Research and Essays*, *5*(3), 296–303.

Hammer J., Kraak M. H., Parsons J. R. (2012). Plastics in the marine environment: the dark side of a modern gift, in de Voogt P., *Reviews of environmental contamination and toxicology* (pp. 1–44), Springer, New York, 1–44.

Hardesty B. D., Good T. P., Wilcox C. (2015). Novel methods, new results and science-based solutions to tackle marine debris impacts on wildlife, *Ocean & Coastal Management*, *115*, 4–9.

Hardesty, B. D., Holdsworth, D., Revill, A. T., and Wilcox, C. (2015). A biochemical approach for identifying plastics exposure in live wildlife. *Methods in Ecology and Evolution*, *6*(1), 92-98.

Hardesty BD, C Wilcox, TJ Lawson, M Lansdell and T van der Velde. 2014. Understanding the effects of marine debris on wildlife. A Final report for Earthwatch Australia. http://www.csiro.au/Organisation-Structure/Flagships/Wealth-from-Oceans-Flagship/marine-debris.aspx

Hardesty BD, TJ Lawson, T van der Velde, M Lansdell, G Perkins and C Wilcox. 2016. Estimating quantities and sources of marine debris at a continental scale. *Accepted* Frontiers in Ecology and the Environment.

Hettiarachchi M., Anurangi J., Alwis A. de. (2011). Characterisation and description of surface water quality in the threatened urban wetlands around the city of Colombo, *Journal of Wetlands Ecology*, *5*, 10–19.

Hidalgo-Ruz V., Thiel M. (2013). Distribution and abundance of small plastic debris on beaches in the SE Pacific (Chile): a study supported by a citizen science project, *Marine Environmental Research*, *87*, 12–18.

Hinojosa I., Thiel M. (2009) Floating marine debris in fjords, gulfs and channels of southern Chile, *Marine Pollution Bulletin*, *58*, 341–350, doi: 10.1016/j.marpolbul.2008.10.020.

Hong S. (2013). Assessment of marine debris pollution and emergency evaluation of its management measures in Korea, PhD dissertation, Graduate School, Pukyong National University, 155.

Hong S., Lee J., Kang D., Choi H.W., Ko S.H., (2014). Quantities, composition, and sources of beach debris in Korea from the results of nationwide monitoring, *Marine Pollution Bulletin*, *84*, 27–34.

Hu L. (2010). Promotion of marine litter monitoring and related activities in China, 2010 NOWPAP International Coastal Cleanup and Workshop on Marine Litter, Jeju, Korea, 1–2 October.

Hutton I., Carlile N., Priddel D. (2008). Plastic ingestion by flesh-footed shearwaters, *Puffinus carneipes*, and wedge-tailed shearwaters, *Puffinus pacificus*, *Papers and Proceedings of the Royal Society of Tasmania*, *142*(1), 67–72.

Isobe A., Kubo K., Tamura Y., Kako S., Nakashima E., Fujii N. (2014). Selective transport of microplastics and mesoplastics by drifting in coastal waters, *Marine Pollution Bulletin*, *89*, 324–330.

Isobe, A., Uchida, K., Tokai, T., Iwasaki, S. (2015). East Asian seas: A hot spot of pelagic microplastics. *Marine pollution bulletin*, *101*(2), 618-623.

Jacobsen J. K., Massey L., Gulland F. (2010). Fatal ingestion of floating net debris by two sperm whales (*Physeter macrocephalus*), *Marine Pollution Bulletin*, *60*(5), 765–767.

Jambeck J. R, Johnsen K. (2015). Marine debris tracker: citizen-based litter and marine debris data collection and mapping, *Computing in Science & Engineering*, *17*, 20–26.

Jambeck J. R., Geyer R., Wilcox C., Siegler T. R., Perryman M., Andrady A., Narayan R., Law K. L. (2015). Plastic waste inputs from land into the ocean, *Science*, *347*(6223), 768–771.

Jang Y. C., Hong S., Lee J., Lee M. J., Shim W. J. (2014). Estimation of lost tourism revenue in Geoje Island from the 2011 marine debris pollution event in South Korea, *Marine Pollution Bulletin*, *81*(1), 49–54.

Jang S. W., Lee S. K., Kim D. H., Chung Y. H., Yoon J. J. (2015). Application of remote environmental monitoring technique to efficient management of beach litter, *International Journal of u- and e-Service*, *8*, 357–368.

Kako S., Isobe A., Magome S. (2012). Low altitude remote-sensing method to monitor marine and beach litter of various colors using a balloon equipped with a digital camera, *Marine Pollution Bulletin*, *64*, 1156–62.

Kako, S. I., Isobe, A., Kataoka, T., & Hinata, H. (2014). A decadal prediction of the quantity of plastic marine debris littered on beaches of the East Asian marginal seas. *Marine pollution bulletin*, *81*(1), 174-184.

Kalnay, E., Kanamitsu, M., Kistler, R., Collins, W., Deaven, D., Gandin, L., ... & Zhu, Y. (1996). The NCEP/NCAR 40-year reanalysis project. *Bulletin of the American meteorological Society*, *77*(3), 437-471.

Kalnay E., Kanamitsu M., Kistler R., Collins W., Deaven D., Gandin L., Iredell M., Saha S., White G., Woollen J., Zhu Y., Leetmaa A., Reynolds R., Chelliah M., Ebisuzaki W., Higgins W., Kuo F. J., Huang H. W. (2014). Strategy for mitigation of marine debris: analysis of sources and composition of marine debris in northern Taiwan, *Marine Pollution Bulletin*, *83*, 70–78.

Karnovsky, N. J., Hobson, K. A., & Iverson, S. J. (2012). From lavage to lipids: estimating diets of seabirds. *Marine Ecology Progress Series*, *451*, 263-284.

Katsanevakis, S., Katsarou, A. (2004). Influences on the distribution of marine debris on the seafloor of shallow coastal areas in Greece (Eastern Mediterranean). *Water, Air, and Soil Pollution*, *159*(1), 325-337.

Katsumata K., Yoshinari H. (2010). Uncertainties in global mapping of Argo drift data at the parking level, *Journal of Oceanography*, *66*, 553–569.

Keller A. A., Fruh E. L., Johnson M. M., Simon V., McGourty C. (2010). Distribution and abundance of anthropogenic marine debris along the shelf and slope of the US West Coast, *Marine Pollution Bulletin*, *60*, 692–700.

Kim M., Hyun S., Kwon J.-H. (2015). Estimation of the environmental load of high- and low-density polyethylene from South Korea using a mass balance approach, *Archives of Environmental Contamination and Toxicology*, *69*(3), 367–373.

KIMO (2010) Fishing for Litter, Kommunenes Internasjonale Miljoorganisasjon, Local Authorities International Environmental Organisation

Kukulka T., Brunner K. (2015). Passive buoyant tracers in the ocean surface boundary layer: influence of equilibrium wind-waves on vertical distributions, *Journal of Geophysical Research—Oceans*, *120*, 3837–3858.

Kukulka T., Proskurowski G., Moret-Ferguson S., Meyer D. W., Law K. L. (2012). The effect of wind mixing on the vertical distribution of buoyant plastic debris, *Geophysical Research Letters*, *39*.

Kuo L.-J., Lee C.-L., Louchouarn P. *et al.* (2014). A centennial record of anthropogenic impacts and extreme weather events in southwestern Taiwan: evidence from sedimentary molecular markers in coastal margin, *Marine Pollution Bulletin*, *86*(1–2), 244–253.

Kuo, F. J., & Huang, H. W. (2014). Strategy for mitigation of marine debris: analysis of sources and composition of marine debris in northern Taiwan. *Marine pollution bulletin*, *83*(1), 70-78.

Lattin G. L., Moore C. J., Zellers A. F., Moore S. L., Weisberg S. B. (2004). A comparison of neustonic plastic and zooplankton at different depths near the southern California shore, *Marine Pollution Bulletin*, *49*, 291–294.

Law K. L., Morét-Ferguson S., Maximenko N. A., Proskurowski G., Peacock E. E., Hafner J., Reddy C. M. (2010). Plastic accumulation in the North Atlantic subtropical gyre, *Science*, *329*(5996), 1185–1188.

Law K. L., Morét-Ferguson S. E., Goodwin D. S., Zettler E. R., DeForce E., Kukulka T., Proskurowski G. (2014). Distribution of surface plastic debris in the eastern Pacific Ocean from an 11-year data set, *Environmental Science & Technology*, *48*(9), 4732–4738.

Law, K. L., Thompson, R. C. (2014). Microplastics in the seas. *Science*, *345*(6193), 144-145.

Lebreton L., Borrero J. (2013). Modeling the transport and accumulation of floating debris generated by the 11 March 2011 Tohoku tsunami, *Marine Pollution Bulletin*, 66, 53–58.

Lebreton L. M., Greer S. D., Borrero J. C. (2012). Numerical modelling of floating debris in the world's oceans, *Marine Pollution Bulletin*, *64*(3), 653–661.

Liu T. K., Wang M. W., Chen P. (2013). Influence of waste management policy on the characteristics of beach litter in Kaohsiung, Taiwan, *Marine Pollution Bulletin*, *72*, 99–106.

Lobelle D., Cunliffe M. (2011). Early microbial biofilm formation on marine plastic debris, *Marine Pollution Bulletin*, *62*, 197–200, http://dx.doi.org/10.1016/j.marpolbul.2010.10.013.

Long M., Moriceau B., Gallinari M., Lambert C., Huvet A., Raffray J., Soudant P. (2015). Interactions between microplastics and phytoplankton aggregates: impact on their respective fates, *Marine Chemistry*, *175*, 39–46.

Lusher A. (2015). Microplastics in the marine environment: distribution, interactions and effects, in Bergmann M., Gutow L., Klages M. (eds), *Marine anthropogenic litter* (pp. 245–307), Springer International Publishing.

Lusher A. L., Burke A., O'Connor I., Officer R. (2014). Microplastic pollution in the northeast Atlantic Ocean: validated and opportunistic sampling, *Marine Pollution Bulletin*, *88*(1), 325–333.

Lusher A. L., Tirelli V., O'Connor I., Officer R. (2015). Microplastics in Arctic polar waters: the first reported values of particles in surface and sub-surface samples, *Scientific Reports*, *5*.

Macfadyen G, Huntington T, Cappell R (2009) Abandoned, lost or otherwise discarded fishing gear. Tech Pap 2009, No. 523, FAO Fisheries and Aquaculture, Rome.

Madzena A., Lasiak T. (1997). Spatial and temporal variations in beach litter on the Transkei coast of South Africa, *Marine Pollution Bulletin*, *34*, 900–907.

Mansui J., Molcard A., Ourmières Y. (2015). Modelling the transport and accumulation of floating marine debris in the Mediterranean basin, *Marine Pollution Bulletin*, *91*(1), 249–57.

Marais M., Armitage N., Wise C. (2004). The measurement and reduction of urban litter entering stormwater drainage systems: Paper 1— Quantifying the problem using the city of Cape Town as a case study, *Water SA*, *30*(4), 469–482.

Maselko, J., Bishop, G., Murphy, P. (2013). Ghost fishing in the Southeast Alaska commercial Dungeness crab fishery. *North American Journal of Fisheries Management*, *33*(2), 422-431.

Masura J., Baker J., Foster G., Arthur C. (2015). *Laboratory methods for the analysis of microplastics in the marine environment: recommendations for quantifying synthetic particles in waters and sediments*, NOAA technical memorandum NOS-OR&R-48.

Maximenko N., Hafner J., Niiler P. (2012). Pathways of marine debris derived from trajectories of Lagrangian drifters, *Marine Pollution Bulletin*, *65*, 51–62.

McDermid K. J., McMullen T. L. (2004). Quantitative analysis of small-plastic debris on beaches in the Hawaiian archipelago, *Marine Pollution Bulletin*, *48*, 790–794.

Merrell, T.R., Jr. (1980). Fishnets and other plastic litter on Alaska Beaches. In *Proceedings of the workshop on the fate and impat of marine debris, 27-29 November 1984.* Honolulu, HI. R. S. Shomura and H.O. Yoshida, eds. Pp. 260-182.

Moret-Ferguson S., Law K. L., Proskurowski G., Murphy E. K., Peacock E. E., Reddy C. M. (2010). The size, mass, and composition of plastic debris in the western North Atlantic Ocean, *Marine Pollution Bulletin*, *60*, 1873–1878, http://dx.doi.org/10.1016/j.marpolbul.2010.07.020.

Neves, V. C., Bolton, M., & Monteiro, L. R. (2006). Validation of the water offloading technique for diet assessment: an experimental study with Cory's shearwaters (Calonectris diomedea). *Journal of Ornithology*, *147*(3), 474-478.

Nilsen F., Hyrenbach K. D., Fang J., Jensen B. (2014). Use of indicator chemicals to characterize the plastic fragments ingested by Laysan albatross, *Marine Pollution Bulletin*, *87*(1), 230–236.

Nixon Z., N. Barnea. (2010). *Development of the Gulf of Mexico Marine Debris Model*, NOAA technical memorandum NOS-OR&R-35.

Njue C. N., Cundy A. B., Smith M. *et al.* (2012). Assessing the impact of historical coastal landfill sites on sensitive ecosystems: a case study from Dorset, Southern England, *Estuarine Coastal and Shelf Science*, *114*(SI), 166–174.

NOAA (National Oceanic and Atmospheric Administration) (no date). *Monitoring and Assessment Project: Get started toolbox*, NOAA, Washington DC, https://marinedebris.noaa.gov/research/monitoring-toolbox.

NOAA (National Oceanic and Atmospheric Administration) (2016). *The Global Drifter Program*, NOAA, Washington DC, http://www.aoml.noaa.gov/phod/dac/index.php.

Nor, N. H. M., & Obbard, J. P. (2014). Microplastics in Singapore's coastal mangrove ecosystems. *Marine pollution bulletin*, *79*(1), 278-283.

Norén, F., 2007. Small plastic particles in Coastal Swedish waters. KIMO Sweden.

NOWPAP CEARAC, *Current situation on marine litter in the NOWPAP region*, Northwest Pacific Action Plan and Special Monitoring & Coastal Environmental Assessment Regional Activity Centre, http://cearac.nowpap.org.

Obbard R. W., Sadri S., Wong Y. Q., Khitun A. A., Baker I., Thompson R. C. (2014). Global warming releases microplastic legacy frozen in Arctic Sea ice, *Earth's Future*, *2*(6), 315–320.

Oberbeckmann, S., Löder, M. G., & Labrenz, M. (2015). Marine microplastic-associated biofilms–a review. *Environmental Chemistry*, *12*(5), 551-562.

O'Brine, T., & Thompson, R. C. (2010). Degradation of plastic carrier bags in the marine environment. *Marine Pollution Bulletin*, *60*(12), 2279-2283.

Ofiara D. D., Brown B. (1999). Assessment of economic losses to recreational activities from 1988 marine pollution events and assessment of economic losses from long-term contamination of fish within the New York Bight to New Jersey, *Marine Pollution Bulletin*, *38*(11), 990–1004.

Øhlenschlæger, J. P., Newman, S., Farmer, A. (2013). Reducing ship generated marine litter—Recommendations to improve the EU port reception facilities directive. Report produced for Seas At Risk. *Institute for European Environmental Policy, London.*

Osborn T. R., Danley H., Martin E. L. (2008). Development and implementation of a large-scale debris survey and mapping project after the Gulf Coast hurricanes of 2005, in McLaughlin K. D. (ed.), *Mitigating Impacts of Natural Hazards on Fishery Ecosystems*, American Fisheries Society Symposium 64, San Francisco, 5–6 September 2007, 95–99.

OSPAR (2007). *Monitoring of marine litter on beaches in the OSPAR region*, OSPAR Commission, London.

OSPAR (2014). *Marine Litter Regional Action Plan*, OSPAR Commission, London.

Otley H., Ingham R. (2003). Marine debris surveys at Volunteer Beach, Falkland Islands, during the summer of 2001/02, *Marine Pollution Bulletin*, *46*, 1534–1539.

Park S., Yoon, H.-S. (2007). Distribution of marine debris collected from the sandbar coastline of Nakdong River estuary after the typhoons' passage, *Journal of the Korean Society of Marine Environment & Safety*, *13*(4), 1–7.

Pawson M. G. (2003). The catching capacity of lost static fishing gears: introduction, *Fisheries Research*, *64*, 101–105.

Pham C. K., Ramirez-Llodra E., Alt C. H. S., Amaro T., Bergmann M., Canals M., Company J. B., Davies J., Duineveld G., Galgani F., Howell K. L., Huvenne V. A. I., Isidro E., Jones D. O. B., Lastras G., Morato T., Gomes-Pereira J., Purser A., Stewart H., Tojeira I., Tubau X., Van Rooij D., Tyler P. A. (2014). Marine litter distribution and density in European seas, from the shelves to deep basins, *PLoS ONE*, *9*(4), e95839, doi:10.1371/journal.pone.0095839.

Plastics Europe (2013) Plastics— The Facts 2013: An Analysis of European Latest Plastics Production, Demand, and Waste Data (Plastics Europe, Brussels).

Poulain P. M., Menna M., Mauri E. (2012). Surface geostrophic circulation of the Mediterranean Sea derived from drifter and satellite altimeter data, *Journal of Physical Oceanography*, *42*(6), 973–990.

Reddy, M. S., Basha, S., Adimurthy, S., & Ramachandraiah, G. (2006). Description of the small plastics fragments in marine sediments along the Alang-Sosiya ship-breaking yard, India. *Estuarine, Coastal and Shelf Science*, *68*(3), 656-660.

Reisser J., Shaw J., Wilcox C., Hardesty B. D., Proietti M., Thums M., Pattiaratchi C. (2013). Marine plastic pollution in waters around Australia: characteristics, concentrations, and pathways, *PloS ONE*, *8*, doi:10.1371/journal.pone.0080466.

Reisser J., Shaw J., Hallegraeff G., Proietti M., Barnes D. K., Thums M., Pattiaratchi C. (2014). Millimeter-sized marine plastics: a new pelagic habitat for microorganisms and invertebrates, *PLoS ONE*, *9*, e100289, DOI: 10.1371/journal.pone.0100289.

Reisser J., Slat B., Noble K., du Plessis K., Epp M., Proietti M., de Sonneville J., Becker T., Pattiaratchi C. (2015). The vertical distribution of buoyant plastics at sea: an observational study in the North Atlantic Gyre, *Biogeosciences*, *12*, 1249–1256.

Ribic, C. A., Dixon, T. R., & Vining, I. (1992). *Marine debris survey manual* (p. 92). US Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

Richardson K., Talouli A., Donoghue M., Haynes D. (2015). Marine pollution originating from purse seine fishing vessel operations in the Western and Central Pacific region, 2004–2014, WCPFC-SC11-2015 ST-IP-05, Scientific Committee 11th Regular Session, Pohnpei, Federated States of Micronesia, 5–13 August 2015.

Rochman C. M. (2015a). The complex mixture, fate and toxicity of chemicals associated with plastic debris in the marine environment, in Bergmann M., Gutow L., Klages M. (eds), *Marine anthropogenic litter* (pp. 117–140), Springer International Publishing.

Rochman, C. M., Tahir, A., Williams, S. L., Baxa, D. V., Lam, R., Miller, J. T., ... & Teh, S. J. (2015b). Anthropogenic debris in seafood: Plastic debris and fibers from textiles in fish and bivalves sold for human consumption. *Scientific reports*, *5*.

Rochman C. M., Browne M. A., Underwood A. J., van Franeker J., Thompson R. C., Amaral-Zettler L. A. (2016). The ecological impacts of marine debris: unravelling the demonstrated evidence from what is perceived, *Ecology*, http://dx.doi.org/10.1890/14-2070.1.

Ryan P. G. (1987). The effects of ingested plastic on seabirds: correlations between plastic load and body condition, *Environmental Pollution*, *46*(2), 119–125.

Ryan P. G. (1988). The characteristics and distribution of plastic particles at the sea-surface off the southwestern Cape Province, South Africa, *Marine Environmental Research*, *25*(4), 249–273.

Ryan P. G. (2008). Seabirds indicate changes in the composition of plastic litter in the Atlantic and south-western Indian oceans, *Marine Pollution Bulletin*, *56*(8), 1406–1409.

Ryan P.G. (2013). Simple technique for counting marine debris at sea reveals steep litter gradients between the Straits of Malacca and the Bay of Bengal, *Marine Pollution Bulletin*, 69(1–2), 128–136.

Ryan, P. G. (2015). A brief history of marine litter research. In *Marine anthropogenic litter* (pp. 1-25). Springer International Publishing.

Ryan P. G., Moore C. J., van Franeker J. A., Moloney C. L. (2009). Monitoring the abundance of plastic debris in the marine environment, *Philosophical Transactions of the Royal Society B: Biological Sciences*, *364*(1526), 1999–2012.

Santos I. R., Friedrich A. C., Do Sul J. A. I. (2009). Marine debris contamination along undeveloped tropical beaches from northeast Brazil, *Environmental Monitoring and Assessment*, *148*(1–4), 455–462.

Schuyler Q., Townsend K., Wilcox C., Hardesty B. D., Marshall J. (2014a). Marine debris through a turtle-eyed view, *BMC Ecology*, http://www.biomedcentral.com/1472-6785/14/14.

Schuyler Q., Hardesty B. D., Wilcox C., Townsend K. (2014b). Global analysis of anthropogenic debris ingestion by sea turtles, *Conservation Biology*, *28*, 129–139.

Schuyler, Q. A., Wilcox, C., Townsend, K. A., Wedemeyer-Strombel, K. R., Balazs, G., Sebille, E., & Hardesty, B. D. (2016). Risk analysis reveals global hotspots for marine debris ingestion by sea turtles. *Global Change Biology*, *22*(2), 567-576.

Secchi, E. R., & Zarzur, S. (1999). Plastic debris ingested by a Blainville's beaked whale, Mesoplodon densirostris, washed ashore in Brazil. *Aquatic Mammals*, *25*(1), 21-24.

Shimizu T., Nakai J., Nakajima K., Kozai N., Takahashi G., Matsumoto M., Kikui J., (2008). Seasonal variations in coastal debris on Awaji Island, Japan, *Marine Pollution Bulletin*, *57*, 182–186.

Sievert, P. R., & Sileo, L. (1993). The effects of ingested plastic on growth and survival of albatross chicks. *The status, ecology, and conservation of marine birds of the North Pacific. Otawa: Canadian Wildlife Service Special Publication*, 212-217.

Song Y., Andrady A. L. (1991). Fouling of floating plastic debris under Biscayne Bay exposure conditions, *Marine Pollution Bulletin*, *22*, 608–613.

Spear L. B., Ainley D. G., Ribic C. A. (1995). Incidence of plastic in seabirds from the tropical Pacific, 1984–1991: relation with distribution of species, sex, age, season, year and body weight, *Marine Environmental Research*, *40*(2), 123–146.

Spengler, A., Costa, M. F. (2008). Methods applied in studies of benthic marine debris. *Marine Pollution Bulletin*, *56*(2), 226-230.

Stefatos A., Charalampakis M., Papatheodorou G., Ferentinos G. (1999). Marine debris on the seafloor of the Mediterranean Sea: examples from two enclosed gulfs in western Greece, *Marine Pollution Bulletin*, *38*(5), 389–393.

Storrier K. L., McGlashan D. J. (2006). Development and management of a coastal litter campaign: the voluntary coastal partnership approach, *Marine Policy*, *30*(2), 189–196.

Suaria G., Avio C., Lattin G., Regoli F., Aliani S. (2015). Neustonic microplastics in the Southern Adriatic Sea, preliminary results from Micro 2015 seminar of the Delfishgear Project, abstract book, Piran, 4–6 May 2015, 42.

Team R. C. (2014). *R: a language and environment for statistical computing*, Foundation for Statistical Computing, Vienna, Austria.

Thiel M., Hinojosa I., Vásquez N., Macaya E. (2003). Floating marine debris in coastal waters of the SE-Pacific (Chile), *Marine Pollution Bulletin*, *46*(2), 224–231.

Thompson R. C., Olsen Y., Mitchell R. P., Davis A., Rowland S. J., John A. W. G., McGonigle D., Russell A. E. (2004). Lost at sea: where is all the plastic?, *Science*, *304*, http://dx.doi.org/10.1126/science.1094559, 838e838.

Tolman H. L. (1997). *User manual and system documentation of WAVEWATCH-III version 1.15*, NOAA/NWS/NCEP/OMB technical note 151.

Troost T., Leslie H., Vethaak A.D. (2015). Impact of microplastics on North Sea marine ecosystems productivity, special newsletter FP7, CleanSea Project.

Turra A., Manzano A. B., Dias R. J. S., Mahiques M. M., Barbosa L., Balthazar-Silva D., Moreira F. T. (2014). Three-dimensional distribution of plastic pellets in sandy beaches: shifting paradigms, *Scientific Reports*, *4*.

UNEP (United Nations Environment Programme) (accessed September 2015). Distribution of marine litter, UNEP, Nairobi, http://www.unep.org/regionalseas/marinelitter/about/distribution/.

UNEP (2009). Marine Litter: A Global Challenge. Nairobi: UNEP. 232 p.

Van Cauwenberghe L., Claessens M., Vandegehuchte M. B., Mees J., Janssen C. R. (2013a). Assessment of marine debris on the Belgian continental shelf, *Marine Pollution Bulletin*, *73*, 161–169, http://dx.doi.org/10.1016/j.marpolbul.2013.05.026.

Van Cauwenberghe L., Vanreusel A., Mees J., Janssen C. R. (2013b). Microplastic pollution in deep-sea sediments, *Environmental Pollution*, *182*, 495–499, http://dx.doi.org/10.1016/j.envpol.2013.08.013.

Van Cauwenberghe L., Devriese L., Galgani F., Robbens J., Janssen C. R. (2015). Microplastics in sediments: a review of techniques, occurrence and effects, *Marine Environmental Research*, *111*, 5–17.

van Franeker J. A. (2010). *Fulmar litter EcoQO monitoring in the Netherlands 1979*, Wageningen Marine Research.

van Franeker J. A. (2011). *Reshape and relocate: seabirds as transformers and transporters of microplastics*, NOAA technical memorandum, NOS-OR&R-*38*, 278–280, http://5imdc.files.wordpress.com/2011/03/5imdc-proceedings-final1.pdf.

van Franeker J. A., Blaize C., Danielsen J., Fairclough K., Gollan J., Guse N., Hansen P. L., Heubeck M., Jensen J.-K., Le Guillou G., Olsen B., Olsen K. O., Pedersen J., Stienen E. W. M., Turner D. M. (2011). Monitoring plastic ingestion by the northern fulmar *Fulmarus glacialis* in the North Sea, *Environmental Pollution*, *159*, 2609–2615 http://dx.doi.org/10.1016/j.envpol.2011.06.008.

van Franeker, J. A., Law, K. L. (2015). Seabirds, gyres and global trends in plastic pollution. *Environmental Pollution*, *203*, 89-96.

van Sebille, E. (2014). Adrift. org. au—a free, quick and easy tool to quantitatively study planktonic surface drift in the global ocean. *Journal of Experimental Marine Biology and Ecology*, *461*, 317-322.

van Sebille E. (2015). The oceans' accumulating plastic garbage, *Physics Today*, *68*, 60–61, http://doi.org/10.1063/PT.3.2697.

van Sebille E., England M. H., Froyland G. (2012). Origin, dynamics and evolution of ocean garbage patches from observed surface drifters, *Environmental Research Letters*, *7*.

van Sebille, E., Wilcox, C., Lebreton, L., Maximenko, N., Hardesty, B. D., Van Franeker, J. A., ... & Law, K. L. (2015). A global inventory of small floating plastic debris. *Environmental Research Letters*, *10*(12), 124006.

Vauk G. J., Schrey E. (1987). Litter pollution from ships in the German Bight, *Marine Pollution Bulletin*, *18*(6), 316–319.

Vianello A., Boldrin A., Guerriero P., Moschino V., Rella R., Sturaro A., Da Ros L. (2013). Microplastic particles in sediments of lagoon of Venice, Italy: first observations on occurrence, spatial patterns and identification, *Estuarine, Coastal and Shelf Science*, *130*, 54–61, http://dx.doi.org/10.1016/j.ecss.2013.03.022.

Vieira B. P., Dias D., Nakamura E. M. *et al.* (2013). Is there temporal variation on solid waste stranding in mangroves? A case study in Ratones mangrove, Florianopolis, Brazil, *Biotemas*, *26*(1), 78–86.

Walker T. R., Reid K., Arnould J. P. Y., Croxall J. P. (1997). Marine debris surveys at Bird Island, South Georgia 1990–1995, *Marine Pollution Bulletin*, *34*, 61–65.

Walker T.R., Grant J., Archambault M.-C. (2006). Accumulation of marine debris on an intertidal beach in an urban park (Halifax Harbour, Nova Scotia), *Water Quality Research Journal of Canada*, *41*, 256–262.

Wang, J., Tan, Z., Peng, J., Qiu, Q., & Li, M. (2016). The behaviors of microplastics in the marine environment. *Marine environmental research*, *113*, 7-17.

Watkins E., ten Brink P., Withana S., Mutafoglu K., Schweitzer J.-P., Russi D., Kettunen M. (2015). *Marine litter: socio-economic study: scoping report*, Institute for European Environmental Policy, London, Brussels.

Wilcox C., van Sebille E., Hardesty B. D. (2015). Threat of plastic pollution to seabirds is global, pervasive, and increasing, *Proceedings of the National Academy of Sciences of the United States of America*, *112*, 11899–11904.

Wilcox C., Hardesty B. D., Sharples R., Griffin D. A., Lawson T. J., Gunn R. (2013). Ghost net impacts on globally threatened turtles: a spatial risk analysis for northern Australia, *Conservation Letters*, DOI: 10.1111/conl.12001.

Wilcox C., Heathcote G., Goldberg J., Gunn R., Peel D., Hardesty B. D. (2014). Understanding the sources, drivers and impacts of abandoned, lost and discarded fishing gear in northern Australia, *Conservation Biology*, *29*(1), DOI: 10.1111/cobi.12355

Wilcox C., Mallos N., Leonard G. H., Rodriguez A., Hardesty B. D. (2016). Using expert elicitation to estimating the impacts of plastic pollution on marine wildlife, *Marine Policy*, 65, 107–114.

Wood S. (2008). *Package 'mgcv'*, http://cran.r-project.org/web/packages/mgcv/mgcv.pdf.

Woodall L. C., Sanchez-Vidal A., Canals M., Paterson G. L. J., Coppock R., Sleight V., Calafat A., Rogers A. D., Narayanaswamy B. E., Thompson R. C. (2014). The deep sea is a major sink for microplastic debris, *Royal Society Open Science*, 1 140317.

Woodall L. C., Robinson L. F., Rogers A. D., Narayanaswamy B. E., Paterson G. L. (2015a). Deep-sea litter: a comparison of seamounts, banks and a ridge in the Atlantic and Indian oceans reveals both environmental and anthropogenic factors impact accumulation and composition, *Frontiers in Marine Science*, *2*, 3.

Woodall L. C., Gwinnett C., Packer M., Thompson R. C., Robinson L. F., Paterson G. L. (2015b). Using a forensic science approach to minimize environmental contamination and to identify microfibres in marine sediments, *Marine Pollution Bulletin*, *95*(1), 40–46.

WPB (Waterfront Partnership of Baltimore), no date. *Mr Trash Wheel*, WPB, Baltimore, http://baltimorewaterfront.com/healthy-harbor/water-wheel/.

Yetman G., Gaffin S. R., Xing X. (2004). *Global 15 x 15 minute grids of the downscaled population based on the SRES B2 scenario, 1990 and 2025*, NASA Socioeconomic Data and Applications Center, Palisades, New York, http://dx.doi.org/10.7927/H4HQ3WTH.

Zettler E. R., Mincer T. J., Amaral-Zettler L. A. (2013). Life in the 'plastisphere': microbial communities on plastic marine debris, *Environmental Science & Technology*, *47*, 7137–7146, http://dx.doi.org/10.1021/es401288x.

Amaral-Zettler, L.A., Zettler, E.R., Slikas, B., Boyd, G.D., Melvin, D.W., Morrall, C.E., Proskurowski, G. and Mincer, T.J., 2015. The biogeography of the Plastisphere: implications for policy. *Frontiers in Ecology and the Environment*, *13*(10), pp.541-546.

Zhou P., Huang C., Fang H., Cai W., Li D., Li X., Yu H., (2011). The abundance, composition and sources of marine debris in coastal seawaters or beaches around the northern South China Sea (China), *Marine Pollution Bulletin*, *62*, 1998–2007.

Acronyms and abbreviations

C-O-C	coast–ocean–coast
CSIRO	Commonwealth Scientific and Industrial Research Organisation (Australia)
EcoQOs	ecological quality objectives
GESAMP	Joint Group of Experts on the Scientific Aspects of Marine Environment Protection
GPA	Global Programme of Action for the Protection of the Marine Environment from Land-Based Activities
GPML	Global Partnership for Marine Litter
NOAA	National Oceanic and Atmospheric Administration (United States)
NOWPAP	Northwest Pacific Action Plan
SEA	Sea Education Association
UNEP	United Nations Environment Programme