Plastics in the Marine Environment

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Abstract

Plastics contamination in the marine environment was first reported nearly 50 years ago, less than two decades after the rise of commercial plastics production, when less than 50 million metric tons were produced per year. In 2014, global plastics production surpassed 300 million metric tons per year. Plastic debris has been detected worldwide in all major marine habitats, in sizes from microns to meters. In response, concerns about risks to marine wildlife upon exposure to the varied forms of plastic debris have increased, stimulating new research into the extent and consequences of plastics contamination in the marine environment. Here, I present a framework to evaluate the current understanding of the sources, distribution, fate, and impacts of marine plastics. Despite remaining knowledge gaps in mass budgeting and challenges in investigating ecological impacts, the increasing evidence of the ubiquity of plastics contamination in the marine environment, the continued rapid growth in plastics production, and the evidence—albeit limited—of demonstrated impacts to marine wildlife support immediate implementation of source-reducing measures to decrease the potential risks of plastics in the marine ecosystem.

1. INTRODUCTION

Plastic pollution in the ocean was first reported by scientists in the 1970s, yet in recent years it has drawn tremendous attention from the media, the public, and an increasing number of scientists spanning diverse fields, including polymer science, environmental engineering, ecology, toxicology, marine biology, and oceanography. The extremely visible nature of much of this contamination is easy to convey in shocking images of piles of trash on coastlines, marine mammals entangled in fishing nets, or seabird bellies filled with bottle caps, cigarette lighters, and colorful shards of plastic. Even without these images, anyone who has visited a beach has certainly encountered discarded cigarette butts, broken beach toys left behind, or pieces of fishing gear or buoys that have washed ashore. Whether as a result of the visceral response evoked by these experiences or the increasing awareness that plastics are ubiquitous and persistent in natural systems, this environmental concern is being addressed at the highest international levels (UNEP 2014, G7 2015). Ultimately, stakeholders and policymakers want to know how big the problem is, how widespread the harm is, and what the best prevention or mitigation strategies are. Scientific inquiry into these questions is not new, but systematic study of the sources, pathways, transformations, impacts, and sinks of plastics in the marine environment has rapidly accelerated only in the last decade (figure 1 in Browne et al. 2015a).

Supplemental Material

Here, I discuss the state of understanding of plastics contamination in the ocean, utilizing a framework that was initially conceived by the Marine Debris Working Group at the National Center for Ecological Analysis and Synthesis (see the **Supplemental Appendix**; follow the **Supplemental Materials link** from the Annual Reviews home page at **http://www.annualreviews.org**). Although my discussion is fundamentally based on the collaborative work of this group, the assessment here is my own and is limited in scope to plastic debris only.

1.1. Plastics and Marine Debris

Plastics are a class of synthetic organic polymers composed of long, chain-like molecules with a high average molecular weight. Many common classes of plastics are composed of hydrocarbons that are typically, but not always, derived from fossil fuel feedstocks (Am. Chem. Counc. 2015). During the conversion from resin to product, a wide variety of additives—including fillers, plasticizers, flame retardants, UV and thermal stabilizers, and antimicrobial and coloring agents—may be added to the resin to enhance the plastic's performance and appearance. The result is a class of materials that have highly versatile and desirable properties (including strength, durability, light weight, thermal and electrical insulation, and barrier capabilities) and can take many forms (such as adhesives, foams, fibers, and rigid or flexible solids, including films).

The first synthetic polymers were developed in the middle of the nineteenth century; rapid development of many new plastics then occurred in the early twentieth century, and commercial production accelerated during World War II (SPI 2015). Global plastics production has increased exponentially since 1950, with 311 million metric tons produced in 2014 (Plast. Eur. 2015). Today, seven commodity thermoplastics account for ~85% of total plastics demand for use in virtually all market sectors (Am. Chem. Counc. 2015) (**Supplemental Figure 1**). The largest market demand (35% in the United States) is for packaging materials (Am. Chem. Counc. 2015), which are designed for short-term use before disposal. Despite the substantial fraction of waste that results from consumer plastics use (12.8% of municipal solid waste by mass in the United States in 2013; US EPA 2016) and the relatively straightforward process of mechanical recycling of thermoplastics (grinding followed by remelting into resin pellets; Andrady 2015), only an estimated 8.8% of postconsumer plastics were recovered for recycling in the United States in 2012 (US EPA 2014).

Plastics recycling rates are higher in Europe but still reached only 30% in 2014 (Plast. Eur. 2015). Even in these highly developed countries with robust infrastructures, obstacles to recycling occur at every step from discard to fabrication of new products. Such obstacles include the unavailability of collection points, contamination of recycling feedstock, and the limited marketability of the recycled material (for a detailed discussion of end-of-life options for plastic waste, see Andrady 2015).

The prevalence of and dependence on plastics in everyday life are reflected in its ubiquitous presence as litter in the environment. Marine debris (or marine litter) consists of any manufactured or processed solid material that was discarded or transported into the marine environment, including glass, metals, paper, textiles, wood, rubber, and plastics. Some of these materials may be readily biodegradable (e.g., paper, wood, or natural fibers), whereas others are long lived in the marine environment. Persistent, nonplastic marine debris has existed for centuries in the form of (for example) sunken wooden vessels that contain ceramic artifacts (Schleicher et al. 2008). However, plastics are unique in that they are both persistent (resistant to biodegradation) and—because of their light weight—readily transportable by wind and water.

With the exception of investigations into item-specific debris, such as derelict fishing gear or lost or abandoned vessels, plastics have become the primary focus of recent marine debris research. Plastics are the most abundant material collected in studies of marine debris floating on the ocean surface (e.g., Law et al. 2010) and collected in beach surveys and beach cleanups (e.g., Thiel et al. 2013, Ocean Conserv. 2014), and they are commonly observed on the seafloor (e.g., Galgani et al. 2000). In addition, some of the earliest publications on marine debris documented risks of plastic debris to wildlife (for a brief history of this research, see Ryan 2015). With the continued growth of plastics production worldwide, the abundance and risks of plastics in the marine environment warrant concern and motivate research not only to quantify plastics contamination and its biological, ecological, social, and economic impacts, but also to inform solutions.

1.2. Framework for Study

The proposed framework to study plastic debris in the marine environment addresses three fundamental questions:

1. How much plastic is in the marine environment?
2. What are the impacts of plastics in the marine environment?
3. What is the risk to a particular cohort (organism, species, assemblage, etc.) from a particular type of plastic debris (item, material, size, form, function, etc.)?

The first question amounts to a mass balance exercise (**Figure 1**), akin to the carbon budgeting carried out since the 1990s to uncover the "missing sink" of anthropogenic carbon dioxide (Keeling et al. 1989). The mass balance can be evaluated using two approaches: (*a*) assessing the plastic inputs into and outputs from the marine environment as a whole and (*b*) quantifying the standing stock of plastics in major marine reservoirs. Of course, reliance on state variables alone is a gross oversimplification of time-dependent processes, ignoring the flux of plastics between reservoirs as well as their transformation within those reservoirs. In addition, the term plastics refers to a broad collection of synthetic materials (**Supplemental Table 1**) that is further diversified by innumerable combinations of chemical additives; thus, their behavior upon entering the marine environment is not easily generalized. However, the simple box model shown in **Figure 1** provides a useful starting point to evaluate available information and to highlight major gaps in data or understanding.

Supplemental Material

The second question seeks to quantify the impacts (negative or positive) that result from an encounter with plastic marine debris. Potential impacts include those that affect marine organisms,

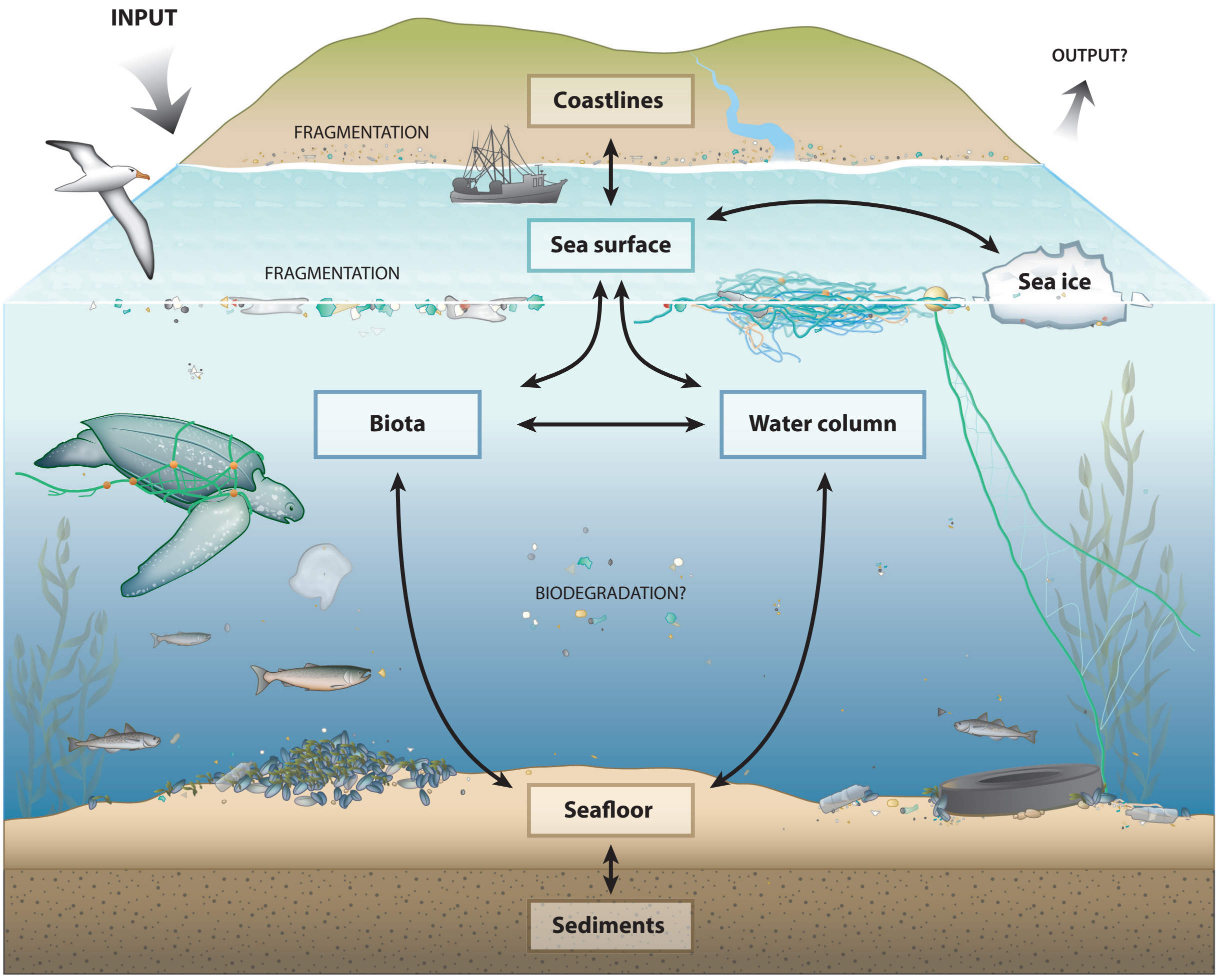

Figure 1

The mass balance of plastics in the marine environment. The large gray arrows indicate fluxes into and out of the marine environment, including potential biodegradation of plastics. The boxes indicate reservoirs of plastic debris, and the black arrows indicate potential pathways of plastics between reservoirs. Fragmentation of plastics caused by weathering and biological processes can occur in all reservoirs, especially when exposed to sunlight (at the sea surface and along coastlines).

habitats, ecosystems, and perhaps even biogeochemical cycling, as well as those that affect human activities, economics, and human health. The most commonly reported interactions between plastic debris and wildlife are entanglement and ingestion, whereas people commonly encounter litter on beaches and large debris as hazards to navigation. Impacts upon encounter with debris are dependent on the particular characteristics of the debris, such as its size, shape, form, and chemical composition. For example, both a large derelict fishing net and a millimeter-sized plastic particle drifting at the sea surface could transport rafting organisms; however, unlike the net, the particle does not pose a hazard to navigation but could be easily ingested. Evidence of impacts might come from observational data (such as surveys of wildlife or habitats), laboratory experiments, or field experiments. Especially for observational data, care must be taken to distinguish evidence of contamination (i.e., the presence of debris) from evidence of impact, or a response to the debris

(Rochman et al. 2016). On the other hand, laboratory and field experiments must ultimately ensure a robust experimental design that reflects environmentally relevant conditions (Rochman & Boxall 2014, Phuong et al. 2016).

To quantitatively assess the consequences of plastic debris and its interactions with constituents of the marine environment, one useful approach is a probabilistic risk assessment framework. The US Environmental Protection Agency (US EPA), for example, commonly uses risk assessments to evaluate the consequences of exposure to environmental stressors on ecosystems (US EPA 1998). Risk assessment frameworks can provide a robust scientific basis for recommendations of remediation or mitigation activities. They can also be used to evaluate uncertainties in the analysis, which are useful to inform the design of future research efforts, particularly if a goal is to inform management decisions (US EPA 1998). Because of the heterogeneous nature of marine plastics, a risk assessment must necessarily target a particular type of debris and/or a cohort that is potentially at risk (Koelmans et al. 2014b).

Although not discussed in this review, social science research is also under way to understand behavioral, societal, and economic drivers of marine debris that might be altered as strategies for reduction (e.g., Ritch et al. 2009, Butler et al. 2013, Newman et al. 2015).

2. MASS BALANCE OF PLASTICS IN THE MARINE ENVIRONMENT

Quantifying the amount of plastic in the marine environment is, in many respects, an accounting exercise, but understanding its sources (rates, locations, and debris forms) and its pathways and transformations after it enters the marine environment is essential to determining the risks and impacts of plastics contamination discussed in Section 3. Without knowledge of exposure, one cannot determine risk.

2.1. Inputs of Plastics

Figure 2 shows a proposed framework for capturing the pathways of plastics into the marine environment, from resin production through loss or discard. The first point of loss is the spillage or mishandling of industrial resin pellets, millimeter-sized quasi-spherical beads that constitute plastic feedstock. Spilled pellets may directly enter waterways or be washed into wastewater or storm-water drains (US EPA 1993). Resin pellets were among the first plastic debris items reported in the ocean (Carpenter & Smith 1972), and they have been detected at sea and on beaches worldwide (Hirai et al. 2011). The abundance of both pellets floating in the North Atlantic and those ingested by northern fulmars in the North Sea has steeply declined since the 1980s (van Franeker & Law 2015), which is hypothesized to reflect a decrease in input after pellet loss prevention measures were recommended to the plastics industries (US EPA 1993). However, an alternative explanation, that a major shift in the geographic location of resin producers or processors resulted in the observed decrease, has not been ruled out.

Once resin is converted into plastic products, those products can enter the environment either unintentionally during use or upon disposal as waste. In this framework, properly managed waste is collected and contained in a robust waste management infrastructure designed to minimize loss to the environment. By contrast, improper management includes open dumping, disposal in open (uncontained) landfills, and littering. By this definition, wastewater discharge is considered proper management; however, plastic microbeads used as abrasives in many personal care products as well as fibers released from synthetic clothing upon washing (Browne et al. 2011) can enter household wastewater. The capture of these particles in wastewater treatment plants (i.e., before the effluent is discharged to the environment) depends on the particular treatment process. Studies

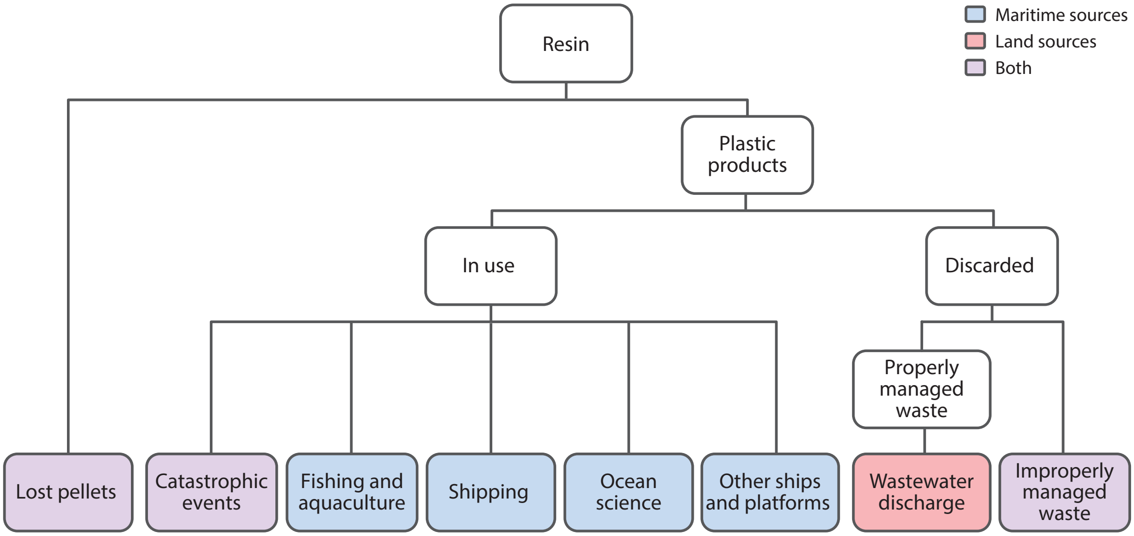

Figure 2

Flow chart describing inputs of plastics into the marine environment, beginning with the manufacture of common plastic resins in the form of industrial pellets. The lowest level shows direct sources to the marine environment; blue shading indicates sources from maritime activities, red indicates sources from land activities, and purple indicates sources from either maritime or land activities.

of wastewater treatment plants in Sweden, Russia, and the United States found extremely high capture rates (> 95%) of plastic particles (Magnusson & Norén 2014, Talvitie & Heinonen 2014, Carr et al. 2016). However, given the immense volume of influent processed through such facilities every day, even low loss rates could result in detectable concentrations of these plastic particles in the environment (Browne et al. 2011, Eriksen et al. 2013).

Unintentional loss of in-service plastic products can occur when catastrophic events, such as tsunamis, hurricanes, or floods, carry large amounts of material of all kinds into the marine environment, or when gear or cargo is lost during maritime use or transport (see **Figure 2**). A 1975 report made estimates of some of these inputs for all material types, finding that cargo-associated waste (dunnage, pallets, plastic sheeting, etc.) accounted for 88% of waste generated (although not necessarily disposed of) at sea (Natl. Res. Counc. 1975). Waste generated by passengers and crew on ocean vessels accounted for 10%, catastrophes for 2%, and commercial fishing gear loss for <1%. International regulations on the discharge of waste at sea [International Convention for the Prevention of Pollution from Ships (MARPOL) 73/78] have since prohibited the discharge of all nonfood solid waste. Presently, the mass of plastics that enters the ocean from maritime activities or catastrophic events is not known.

The only major source of plastics to the ocean that has been estimated globally is improperly managed plastic waste generated on land (Jambeck et al. 2015). This analysis used data compiled by the World Bank (Hoornweg & Bhada-Tata 2012) on per capita waste generation rate, waste composition, and waste disposal in 192 coastal countries to estimate the total amount of plastic waste generated and the amount that is uncontained because of improper management (including littering). The estimate of waste available to enter the ocean was scaled by populations living within 50 km of the coast, with the understanding that waste generated farther inland might also be transported to the ocean. Because the flux of uncontained waste entering the ocean from land is essentially unmeasured, conversion rates between 15% and 40% were applied to give a first estimate of plastic input to the ocean of 4.8–12.7 million metric tons in 2010. A more refined estimate will require direct measurement of the input rates of plastic waste by river, wind, tidal, and ocean wave transport as well as methodical measurement of waste generation, classification, collection rates, and waste disposal methods for rural areas and urban centers in countries around the world.

2.2. Sampling and Analytical Methods for Marine Plastics

Many of the earliest reports of plastic debris in the ocean were of small floating particles that were captured in surface-towing plankton nets (Carpenter & Smith 1972, Colton et al. 1974, Wong et al. 1974). Other reports included synthetic fibers in water samples (Buchanan 1971), shipboard visual observations of large floating debris (Venrick et al. 1973), seafloor debris in benthic fishing trawls (Holmström 1975), and plastic debris on beaches (Cundell 1973, Dixon & Cooke 1977). Ingestion of plastics by seabirds (Harper & Fowler 1987) and sea turtles (Balazs 1985) began as early as the 1960s. By the very nature of these observations (more formally, the sampling design), plastics of different materials, sizes, and forms were selectively reported. Today, published observations and measurements of plastic debris in all of these reservoirs (coastlines, sea surface, seafloor, and biota) as well as the water column, sediments, and sea ice (**Figure 1**) are numerous and global, yet the most commonly used sampling strategies remain much the same as they were in the 1970s, with relatively little standardization across studies. Thus, when attempting to estimate the mass of plastics in any one marine reservoir, one must carefully consider the sampling methods used to collect each data set (Browne et al. 2015a, Filella 2015).

Plastic marine debris has been reported in sizes ranging from microns to meters. Although widely used, the terms microplastic and macroplastic have no generally agreed-upon definition. Microplastics are most commonly defined as particles smaller than 5 mm (Arthur et al. 2009), but they have also been defined as particles smaller than 1 mm (e.g., Browne et al. 2011) and have been functionally defined (at the lower limit) as particles retained by plankton nets or sieves with variable mesh sizes (Arthur et al. 2009). The smallest particles detected in the marine environment are only a few microns in size (Ng & Obbard 2006), and even smaller, nanometer-sized plastics are hypothesized to exist, but no reliable method has been developed to detect and identify them (Koelmans et al. 2015). The term macroplastic is even more ambiguous, often referring simply to debris larger than microplastics.

The particle size distribution of plastic marine debris has not been satisfactorily measured in any marine reservoir. Although several studies of microplastics in water and sediment have reported particle size information (see Hidalgo-Ruz et al. 2012), the lack of consistency and completeness in size characterization (i.e., equivalent spherical diameter and shape factor) and in concentration measure (i.e., number or mass), as well as other methodological problems, prevents direct comparison of results (Filella 2015). In addition, particle size distribution is dynamic for at least two reasons. First, plastics of variable and largely unknown size continually enter (and perhaps leave) the system. Second, plastics fragment with time because of weathering. Exposure to UV radiation initiates photo-oxidative degradation in plastics that reduces average molecular weight, weakening the material until shear or tensile stresses cause fracturing and fragmentation (Andrady 2015). No experimental studies have described fragmentation under marine exposures, and thus theoretical fragmentation models (Cozar et al. 2014, Eriksen et al. 2014) remain untested. Also, the timescales for fragmentation resulting from weathering-induced degradation are unknown, but they depend on environmental factors that determine photo-oxidation and thermo-oxidation reaction rates. These factors include light exposure, oxygen concentration and temperature, and biotic factors such as biofouling, all of which are extremely variable in the marine environment (Andrady 2015).

To properly quantify the mass of plastic in each marine reservoir requires spatially distributed measurements of all size classes of debris at global scales, a prerequisite far from being met. In fact, the sampling methods typically used to quantify the abundance of plastic marine debris vary by marine reservoir and select for particular debris sizes. At present, all methods ultimately depend on visual selection of items or particles by the human eye. The most direct visual selection methods occur in surveys of debris at the sea surface from ships or aircraft, on beaches or coastlines in person or by aircraft, and on the seafloor by divers or towed underwater camera systems, in which only debris visible to the observer (for direct observation) or to the analyst (for photographs or video) is recorded.

Rigorous distance sampling protocols exist for at-sea visual surveys, but it may be difficult to satisfy methodological assumptions such as 100% detection rate of objects on a transect line and accurate measurement of the distance to sighted objects (Williams et al. 2011), especially in variable environmental conditions and for objects with variable sizes, colors, and buoyancies (Ryan et al. 2009). In practice, a wide variety of survey protocols are reported in varying levels of detail, often omitting even minimum detection size; thus, it is extremely challenging to compare data sets reporting abundance quantities for visible (macroplastic) floating debris.

In a critical review of 104 studies of stranded intertidal debris, Browne et al. (2015a) found that site selection strongly favors beaches (95% of studies, mostly performed on sandy beaches) over other coastal habitats, and that widely variable sampling methodologies with respect to site selection, types and sizes of measured debris, reported units (counts or mass), and spatial and temporal replication render data sets too disparate to allow for rigorous global-scale assessments.

Visual surveys of the seafloor to quantify debris are still relatively few in number and are particularly challenging because of the inaccessibility and cost of surveying, but they are now more frequently used than traditional bottom-trawling assessments (Pham et al. 2014). Deep surveys in remote regions have demonstrated the presence of plastic debris far from human populations, including at a depth of ~2,500 m in the Charlie-Gibbs Fracture Zone of the Mid-Atlantic Ridge (Pham et al. 2014) and at a depth of ~2,450 m in the Fram Strait (79°N) (Bergmann & Klages 2012), illustrating a potentially large reservoir for plastic debris on the seafloor, albeit an extremely difficult one to quantify.

Small plastic debris (microplastics) in seawater and sediments (and, in one study, sea ice; Obbard et al. 2014) is typically quantified by filtering the medium either in the field (e.g., seawater through plankton nets) or in the laboratory (sieving and/or filtering bulk sediment or water samples) to reduce the volume for analysis (Hidalgo-Ruz et al. 2012). The minimum size of retained particles varies widely depending on the size of the plankton net mesh (53 μm–3 mm), sieve mesh (0.5–2 mm), or bulk sample filter (1.6–2 μm) (Hidalgo-Ruz et al. 2012). Sample processing may include chemical digestion of organic matter and/or density separation, in which the sample is mixed with seawater (plankton net samples) or a high-density salt solution (sediment samples) in which some or all of the common consumer plastics (**Supplemental Table 1**) are expected to float (Löder & Gerdts 2015). Ultimately, the processed sample is subject to visual analysis, with or without the aid of a dissecting microscope, to identify potential plastic particles.

Supplemental Material

Visual detection may introduce several types of errors, including observer bias (Dekiff et al. 2014), misidentification of particles similar in appearance to organic matter, or underdetection of particles that are too small (even under magnification) to be detected by the human eye (Filella 2015). Furthermore, especially as particle size decreases and visual identification becomes less reliable, it is necessary to verify that extracted particles are indeed synthetic polymers. Fourier transform infrared spectroscopy and Raman spectroscopy are the most commonly used methods for material identification, although pyrolysis–gas chromatography with mass spectrometry has also been used to identify polymer type and organic additives (Fries et al. 2013). Because of the time-consuming nature of individual-particle analysis by these techniques, most microplastics studies that verify material type identify only a small number of particles [e.g., <1% of particles identified in Cozar et al. 2014 (67 of 7,359) and Cooper & Corcoran 2010 (56 of 6,082)], and many studies simply confirm polymer identity without details about the number of particles extracted or identified. Not only does the potential for underdetection or misidentification of plastic particles likely increase with decreasing particle size, but procedural contamination, especially by fibers, also becomes a serious concern (Dekiff et al. 2014, McCormick et al. 2014).

2.3. Estimating Marine Terms in the Mass Balance

Of the data sets currently available, the largest and most geographically widespread collection of data sampled and analyzed in a broadly consistent manner is that measured using surface-trawling plankton nets. Van Sebille et al. (2015) assembled nearly 12,000 measurements of plastic abundance collected between 1971 and 2013 and reported in 26 studies. These data were standardized using a rigorous statistical model to account for variance associated with spatial and temporal distribution, trawl length, and wind speed, which affects sampling conditions as well as vertical mixing of plastic particles below the sea surface (Kukulka et al. 2012). The standardized data (**Figure 3**) were then used to scale the outputs of three ocean circulation models that predict debris distribution, in order to estimate the global mass inventory of small (i.e., net-collected) plastics. The three estimates ranged from 93,000 to 236,000 metric tons, with the large variation resulting from the dearth of data available to constrain the model solutions, especially outside of the North Atlantic and North Pacific subtropical gyres. These results are larger than previous global estimates (Cozar et al. 2014, Eriksen et al. 2014) but can still account for only ~1% of the plastic waste estimated to enter the ocean from land in a single year (Jambeck et al. 2015). The standing stock of one size class of debris in a single reservoir is not expected to equal the annual input rate; however, the size of the discrepancy reveals a fundamental gap in understanding of the major pathways and transformations of plastics upon entering the marine environment.

As discussed above, widespread and comparable environmental data simply do not yet exist to estimate the standing stock of plastic debris (especially large debris) floating at the sea surface, or debris of any size sitting on coastlines or on the seafloor. Only a small number of plankton net tows have been used to investigate plastics at depths below the wind-mixed layer, where plastic particles have been detected, albeit in much lower concentrations than at the surface (Doyle et al. 2011). Microplastics of various forms (e.g., pellets, fragments, and fibers) have been detected in beach sediments around the world (Van Cauwenberghe et al. 2015), and those found in deep-sea sediments (Van Cauwenberghe et al. 2013, Woodall et al. 2014, Fischer et al. 2015) have mainly been fibers. Again, variable methods combined with sparse data distribution prevent meaningful budget calculations of plastics in sediments. Two coarse estimates have been made for plastics ingested by marine biota. From an analysis of plastics in the stomachs of 141 mesopelagic fishes, Davison & Asch (2011) estimated an annual plastic ingestion rate in the North Pacific subtropical gyre of 12,000–24,000 tons. Similarly, two entirely different populations of seabirds were estimated to ingest 6 tons per year per population (Kühn et al. 2015). Considering the continuing discovery of plastic ingestion by a growing cohort of marine organisms, biota could be a sizable reservoir for small plastic debris.

The discussion thus far has focused on a quasi-synoptic view of the spatial distribution of ocean plastics. Even harder to quantify is its variation in time. Given the slow growth in plastics recycling rates (US EPA 2014) compared with the extremely rapid growth in plastics production (Plast. Eur. 2015), the amount of plastic in the ocean has certainly increased with time. Significant

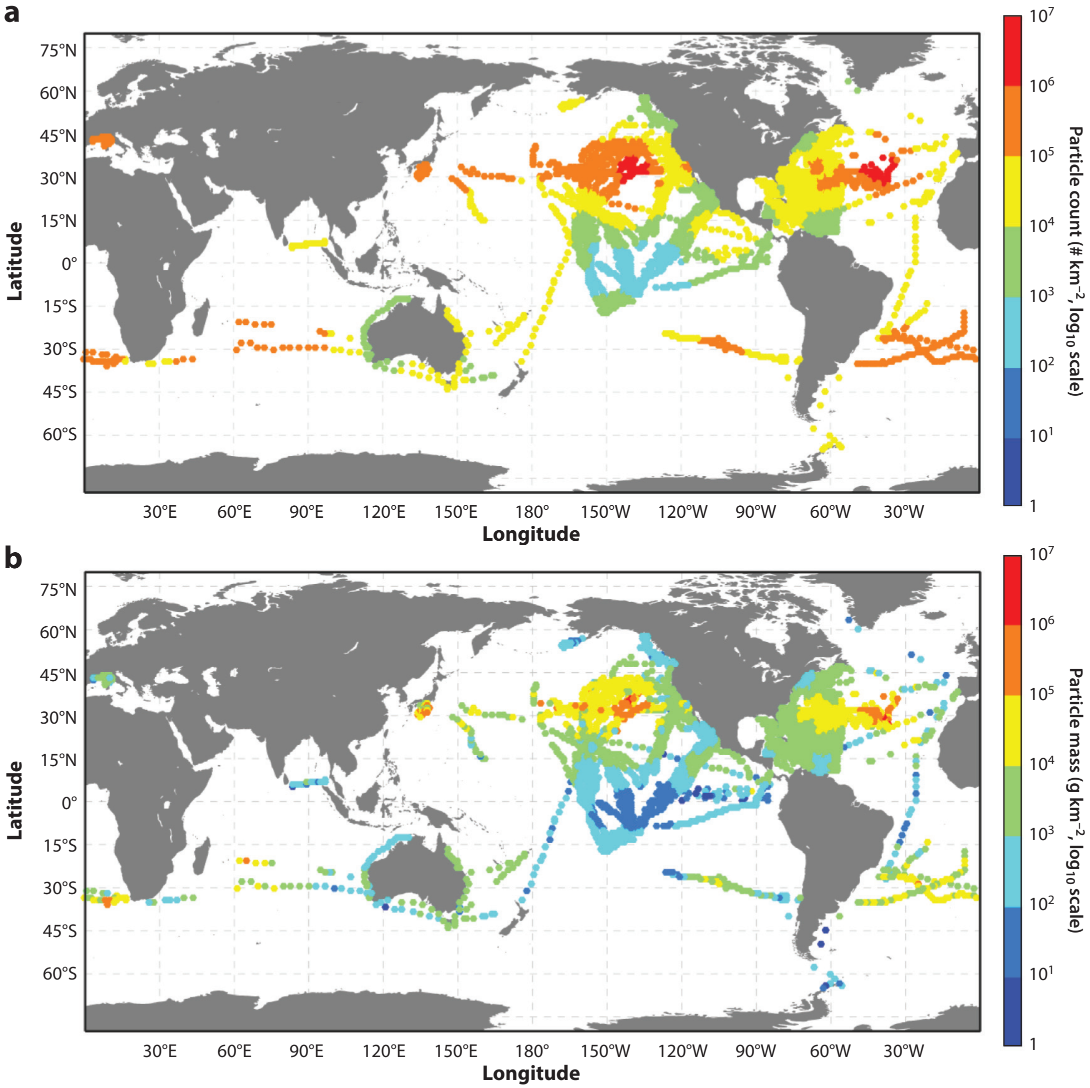

Figure 3

(*a*) Particle count and (*b*) particle mass of plastic samples collected from 11,854 surface-towing plankton net trawls. The data were standardized using a generalized additive model to represent no-wind conditions in the year 2014. Adapted from van Sebille et al. (2015) under the Creative Commons Attribution 3.0 Unported license (**https://creativecommons.org/licenses/by/3.0/legalcode**).

increases in surface microplastics concentrations have been reported over >30-year time periods from records dating back to the 1960s (Thompson et al. 2004) and 1970s (Goldstein et al. 2012), yet these increases have not been detected in more recent long-term data sets (Law et al. 2010, 2014). Trends may be masked by large spatiotemporal variability resulting from factors such as variable sampling conditions (van Sebille et al. 2015), vertical wind-driven turbulent mixing (Kukulka et al. 2012, Brunner et al. 2015), and surface convergences and divergences on scales from meters to thousands of kilometers (Law et al. 2014), which require a large sampling effort to resolve (Goldstein et al. 2013).

Further complicating temporal changes are the unknown rates of plastics transformation within and transport between marine reservoirs. Timescales by which large objects are fragmented to microplastics by weathering-induced and biologically mediated processes, such as grinding in bird gizzards or biting by fishes (Kühn et al. 2015), are not well constrained. Plastics that are initially buoyant may be transported to greater depth upon increased density caused by biofouling (Ye & Andrady 1991, Fazey & Ryan 2016), ingestion by vertically migrating species (Choy & Drazen 2013), or sinking within fecal pellets (Cole et al. 2016) or marine aggregates (Long et al. 2015). Some of these processes have been demonstrated in laboratory or field experiments, but their rates in the environment are unknown. Finally, the residence time of debris on shorelines depends on the physical characteristics of the environment and of the debris. Even in a localized study, measured beach accumulation rates were strongly dependent on sampling frequency (Smith & Markic 2013).

Perhaps the smallest term in the mass balance framework is the output of plastics from the marine environment. Removal mechanisms include transport onshore after ingestion by marine animals or during catastrophic events, intentional removal during research or cleanup efforts, and biodegradation. Although the timescale of biologically mediated mineralization of plastic materials in most environments is not known, it is probably at least decades or centuries and is almost certainly longer in the ocean (Andrady 2015). Thus, it is suspected that the marine environment is essentially a sink for plastic debris.

3. IMPACTS OF PLASTIC DEBRIS ON THE MARINE ECOSYSTEM

The 1975 US National Research Council report discussed a variety of marine litter interactions with potential impacts on the marine ecosystem and on human activities, most of which are the subject of continued study today. The potential impacts included entanglement by debris leading to injury, trapping, or drowning; ingestion of debris causing physical injury, obstruction of the gut, or accumulation of indigestible material in the gut; debris damaging or clogging gills; floating debris acting as a substrate for long-distance transport of rafting organisms; debris on the seafloor providing shelter for small animals; floating or seafloor debris attracting fish or other marine life; floating debris as a navigational hazard, interfering with ship propellers, or clogging water intake pipes; and seafloor debris interacting with marine equipment, such as fishing gear. However, despite a collection of cited reports documenting particular instances of debris impacts, the dearth of available data led the authors to conclude that the overall impact of marine litter was predominantly aesthetic (Natl. Res. Counc. 1975).

In the intervening decades, hundreds of publications have documented encounters between marine debris and nearly 700 species of marine wildlife (Gall & Thompson 2015). For particular species or populations, documented encounters occur frequently. For example, 95% of 1,295 beached seabird (northern fulmar) carcasses in the North Sea contained plastic in their stomachs (van Franeker et al. 2011), and 83% of 626 North Atlantic right whales examined in 29 years of sighting photographs had evidence of at least one entanglement in rope or netting (Knowlton et al. 2012). The prevalence of such encounters and the increasing evidence of widespread contamination of marine habitats with plastic debris naturally leads to concern about adverse impacts ranging from the subcellular level to populations or community structures that might alter ecosystem functioning. However, care must be taken to distinguish evidence of contamination (of habitats or organisms) by plastic debris and its associated chemicals from evidence of impacts, or responses to encounters with debris. For example, although it is generally and reasonably perceived that a stomach full of nonnutritive plastic is not beneficial to an organism, evidence is required to demonstrate that this ingested plastic causes specific harm. Correlative evidence, such as an inverse relationship between fat deposition and amount of ingested plastics in seabirds (Connors & Smith 1982), might support a causative impact; however, an equally valid hypothesis is that ingestion of plastics is a consequence of animals with reduced fat reserves being malnourished and eating plastic, or that reduced fat reserves stem from an entirely different environmental stressor. Rochman et al. (2016) conducted a critical and systematic review of published literature on the perceived, tested, and demonstrated impacts of anthropogenic debris (all materials in all environments) as a function of debris size and affected level of biological organization (i.e., assemblage, population, organism, and suborganism levels; note that the construct did not account for some behavioral or physiological responses, such as altered feeding, movement, or growth).

A comprehensive review of the literature on encounters with and biological impacts of plastic marine debris is beyond the scope of this article, and I refer readers to several recent reviews for more detail (Gall & Thompson 2015, Kiessling et al. 2015, Kühn et al. 2015, Lusher 2015, Rochman 2015, Rochman et al. 2016). Here, I present an overview of the types of encounters documented between marine organisms and plastic debris and the potential and demonstrated impacts of such encounters to convey the state of understanding, including major gaps that require further research. The demonstrated impacts presented here are derived from an analysis by Rochman et al. (2016), selecting only for marine debris that wholly or partially consists of plastic (**Figure 4**, **Table 1**). This framework is a useful way not only to evaluate the available evidence of impacts of particular sizes and types of debris, but also to identify impacts of concern that have not been rigorously tested.

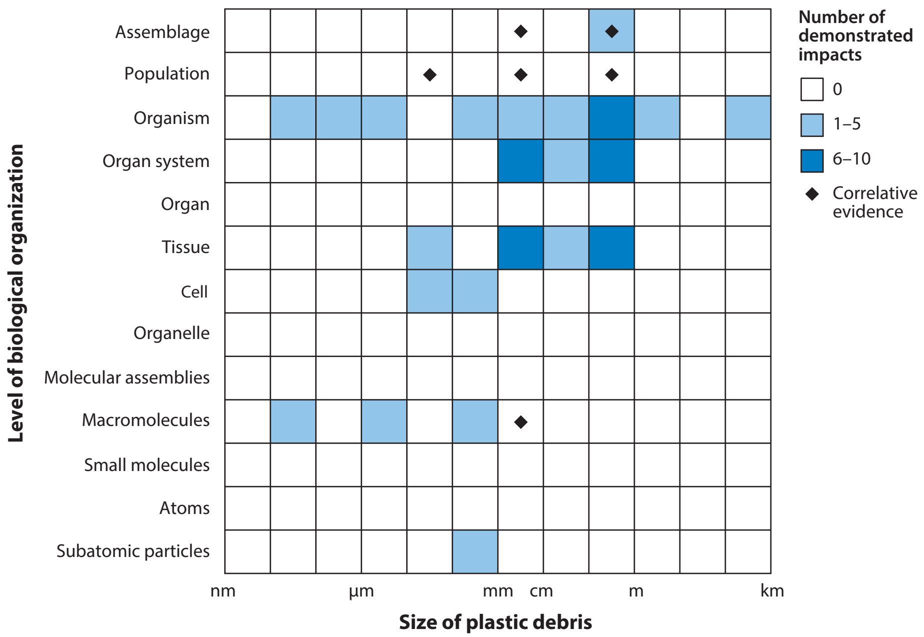

Figure 4

Demonstrated impacts of plastic marine debris as a function of debris size and affected level of biological organization. Each matrix cell represents the number of impacts identified from the peer-reviewed literature through the year 2013, taken from an analysis by Rochman et al. (2016) for impacts caused only by plastic marine debris. Diamonds in cells indicate correlative evidence supporting at least one impact. Impacts in multiple matrix cells may have been demonstrated in a single paper, and thus there are more impacts shown in this figure than there are published studies listed in **Table 1**.

Table 1 Peer-reviewed studies demonstrating evidence of impacts of plastic marine debris

Study	Animal	Encounter type	Predominant debris type	Impact (response)
Allen et al. 2012	Grey seals	Entanglement	MF line, net, rope	Constriction
Beck & Barros 1991	Manatees	Entanglement	MF line, bags, other debris	Death
Campagna et al. 2007	Elephant seals	Entanglement	MF line, fishing jigs	Dermal wound
Croxall et al. 1990	Fur seals	Entanglement	Packing bands, fishing gear, other debris	Dermal wound
Dau et al. 2009	Seabirds, pinnipeds	Entanglement	Fishing gear	External wound
Fowler 1987	Fur seals	Entanglement	Trawl netting, packing bands	Death
Fowler 1987	Fur seals	Entanglement	Trawl netting, packing bands	Reduced population size
Good et al. 2010	Invertebrates, fish, seabirds, marine mammals	Entanglement	Derelict gillnets	Death
Moore et al. 2009	Seabirds, marine mammals	Entanglement	Plastic, fishing line	Death
Pham et al. 2013	Gorgonians	Entanglement	Fishing line	Damage/breakage
Vélez-Rubio et al. 2013	Sea turtles	Entanglement	Fishing gear	Death
Winn et al. 2008	Whales	Entanglement	Plastic line	Dermal wound
Woodward et al. 2006	Whales	Entanglement	Plastic line	Dermal wound
Beck & Barros 1991	Manatees	Ingestion	MF line, bags, other debris	Death
Bjorndal et al. 1994	Sea turtles	Ingestion	MF line, fish hooks, other debris	Intestinal blockage, death
Brandão et al. 2011	Penguins	Ingestion	Plastic, fishing gear, other debris	Perforated gut, death
Browne et al. 2013	Lugworms (laboratory)	Ingestion	Microplastics	Biochemical/cellular, death
Bugoni et al. 2001	Sea turtles	Ingestion	Plastic bags, ropes	Gut obstruction, death
Carey 2011	Seabirds	Ingestion	Plastic particles, pellets	Perforated gut
Cedervall et al. 2012	Fish (laboratory)	Ingestion	Nanoparticles	Biochemical/cellular
Connors & Smith 1982	Seabirds	Ingestion	Plastic pellets, foam	Biochemical/cellular
Dau et al. 2009	Seabirds, pinnipeds	Ingestion	Fishing hooks	Internal wound
de Stephanis et al. 2013	Sperm whale	Ingestion	Identifiable litter items	Gastric rupture, death
Fry et al. 1987	Seabirds	Ingestion	Plastic fragments, pellets, identifiable litter	Gut impaction, ulcerative lesions
Jacobsen et al. 2010	Sperm whales	Ingestion	Fishing gear, other debris	Gastric rupture, gut impaction, death
Lee et al. 2013	Copepods (laboratory)	Ingestion	Micro- and nanoplastics	Death
Oliveira et al. 2013	Fish (laboratory)	Ingestion	Microplastics	Biochemical/cellular

(*Continued*)

Table 1 (*Continued*)

Study	Animal	Encounter type	Predominant debris type	Impact (response)
Rochman et al. 2013a–c	Fish (laboratory)	Ingestion	Microplastics	Biochemical/cellular
Ryan 1988	Birds (laboratory)	Ingestion	Microplastics	Reduced organ size
Vélez-Rubio et al. 2013	Sea turtles	Ingestion	Marine debris	Gut obstruction
Wright et al. 2013	Lugworms (laboratory)	Ingestion	Microplastics	Biochemical/cellular
Von Moos et al. 2012	Mussels (laboratory)	Ingestion and gill uptake	Microplastics	Biochemical/cellular
Katsanevakis et al. 2007	Epibenthic megafauna	Interaction (contact)	Plastic bottles, glass jars	Altered assemblage
Lewis et al. 2009	Sessile invertebrates (coral reef)	Interaction (contact)	Lobster traps	Altered assemblage
Uneputty & Evans 1997	Assemblage on sediment	Interaction (contact)	Plastic litter	Altered assemblage
Chiappone et al. 2002	Sessile invertebrates (coral reef)	Interaction (contact)	MF line, lobster trap, hook and line gear	Tissue abrasion
Chiappone et al. 2005	Sessile invertebrates (coral reef)	Interaction (contact)	Hook and line gear	Tissue abrasion
Uhrin & Schellinger 2011	Seagrass	Interaction (contact)	Crab pots, tires, wood	Breakage, suffocation, death
Özdilek et al. 2006	Sea turtles	Interaction (obstruction)	Waste, medical waste	Reduced population size
Widmer & Hennemann 2010	Ghost crabs	Interaction (obstruction)	Beach litter, mostly plastic	Reduced population size
Widmer & Hennemann 2010	Ghost crabs	Interaction (substrate)	Beach litter, mostly plastic	Altered assemblage
Goldstein et al. 2012	Marine insects	Interaction (substrate)	Microplastics	Increased population size

This table is based on analysis by Rochman et al. (2016) for publications through the year 2013, extracting studies for plastic marine debris only. Shading indicates correlative evidence only. Abbreviation: MF, monofilament line.

The types of encounters that have been described in the literature can be loosely categorized into three groups: entanglement, ingestion, and interaction. Entanglement refers to debris encircling, constricting, or entrapping a marine animal and includes so-called ghost fishing, or the continued trapping of wildlife by derelict fishing gear. Ingestion of plastic debris may be intentional, accidental, or indirect (through prey that has ingested plastic) by animals ranging in size from planktonic invertebrates to large marine mammals. Interaction includes nonentangling contact with debris, such as collision or blanketing, as well as debris presenting an obstruction, providing shelter, or acting as a substrate for growth and/or transport.

Gall & Thompson (2015) reported that 85% of publications about marine debris encounters described incidences of entanglement by or ingestion of debris, with at least 17% of affected species categorized as near threatened to critically endangered on the International Union for Conservation of Nature and Natural Resources (IUCN) Red List of Threatened Species. The vast majority (92%) of the debris in reported encounters with individual organisms was plastic. Entanglement has now been reported for 344 species, including 100% of marine turtles, 67% of seals, 31% of whales, and 25% of seabirds, as well as 89 species of fish and 92 species of invertebrates (Kühn et al. 2015). Entanglements most commonly involve plastic rope and netting

(Gall & Thompson 2015) and other components of derelict fishing gear (Kühn et al. 2015) but may also be caused by packing or strapping bands (e.g., Fowler 1987) and other litter that can form entangling loops. Hazards of entanglement include bodily harm, such as injury to dermal tissue (a demonstrated impact; **Table 1**); interference with growth, potentially causing deformations; and restricted movement affecting swimming, feeding, and the ability to escape predators. These hazards might ultimately result in drowning, starvation, or predation of individuals. Multiple studies have demonstrated death caused by entanglement (**Table 1**).

Reports of ingestion of plastic debris are widespread and increasing as investigators study a broader range of marine organisms. Some of the earliest reports documented ingestion of plastic debris in seabirds, sea turtles, a manatee, and cetaceans (Ryan 2015), and plastic ingestion has now been documented for 233 marine species, including 100% of marine turtles, 36% of seals, 59% of whales, and 59% of seabirds, as well as 92 species of fish and 6 species of invertebrates (Kühn et al. 2015, Wilcox et al. 2015). In contrast to entanglement, no particular form or item is typically associated with ingestion, although the size of the ingested debris is obviously limited by the size of the ingesting organism. For example, plastic fibers and small particles have been detected in filter-feeding oysters and mussels (e.g., Van Cauwenberghe & Janssen 2014) and suspension-feeding barnacles (Goldstein & Goodwin 2013); larger litter items, such as potato chip bags and cigarette box wrapping, have been found in the stomachs of large pelagic fish (Jackson et al. 2000); and very large debris items, including 9 m of rope, 4.5 m of hose, two flowerpots, and large amounts of plastic sheeting, were found in the stomach of a stranded sperm whale (de Stephanis et al. 2013).

Ingested debris may have a variety of consequences for the consuming organism. Large volumes of debris have been hypothesized to reduce storage capacity in the stomach (McCauley & Bjorndal 1999) and to cause false satiation, leading to a reduced appetite (Day et al. 1985), and they have also been shown to cause obstruction of the gut (**Table 1**). The ingested debris can cause internal injury, such as a perforated gut, ulcerative lesions, or gastric rupture, potentially leading to death (**Table 1**). In laboratory studies, several biochemical responses and impacts at the cellular level caused by ingestion of plastics have also been demonstrated, such as oxidative stress (Browne et al. 2013), changes in metabolic parameters (Cedervall et al. 2012), reduced enzyme activity (Oliveira et al. 2013), and cellular necrosis (Rochman et al. 2013c). At least eight studies have demonstrated the death of an organism because of ingestion of plastic marine debris (**Table 1**), but no studies have presented direct evidence of this impact on a population (**Figure 4**).

Animals that ingest plastic debris may also be at risk of contamination by chemicals associated with plastics that are incorporated during manufacture or that accumulate from contaminated environmental matrices such as sediment or seawater. Many of these substances are known to be persistent, bioaccumulative, and toxic (PBT), with at least 78% of the priority pollutants identified by the US EPA known to be associated with plastic marine debris (Rochman et al. 2013a). PBT substances are typically hydrophobic and therefore readily sorb out of seawater onto other hydrophobic substances, such as sediment, organic matter, and now plastic (Rochman 2015). In fact, because of their strong attraction to PBT substances, some plastics are utilized as passive sampling devices to measure chemical contaminants in a variety of environmental matrices (Lohmann 2012).

The sorption of chemicals from seawater to plastic particles has been clearly demonstrated (e.g., Ogata et al. 2009, Hirai et al. 2011, Rochman et al. 2013b), and the rate and extent of accumulation depend on the polymer type, the physical and chemical properties of the plastic (especially those resulting from weathering and biofilm formation), the particle surface area, and the chemical exposure throughout the particle's drift history (Rochman 2015). Because weathering and biofouling processes continually alter the particle surface in ways that increase the affinity for chemical sorption, it has been hypothesized that the accumulation of chemicals onto plastic debris will increase with time in seawater, potentially rendering them more hazardous to animals that ingest the debris (Rochman 2015).

The risk to marine organisms from ingestion of plastic debris with chemical contaminants is presently an area of primary research (for detailed reviews, see Koelmans 2015 and Rochman 2015). Many of these chemicals are already known to have adverse effects on organisms; thus, the question is more about the extent of the transfer of chemicals from plastic to the animal tissue upon ingestion. This extent will depend on the chemical concentration in the plastic and the body burden already present in the animal from other exposure pathways, such as through the food web (Teuten et al. 2009) or uptake from seawater through the dermis or gills (Koelmans et al. 2014a). Chemical transfer depends on the fugacity gradient between the ingested plastic and gut tissue, which could be affected by the presence of natural food, as well as the residence time of plastic in the gut (Koelmans 2015). Chemicals will move toward the phase with a lower concentration en route to equilibrium. As such, Gouin et al. (2011) have even suggested, using thermodynamic modeling, that a relatively uncontaminated piece of plastic could essentially clean a contaminated animal by moving chemicals from the animal tissue to the plastic.

The ability of chemicals to transfer from plastics to animals upon ingestion has been clearly demonstrated in laboratory animals for a variety of plastic-chemical-animal combinations (e.g., Teuten et al. 2009, Besseling et al. 2013, Chua et al. 2014). However, studies must ultimately demonstrate that the experimental fugacity gradient is representative of environmental conditions. For example, "clean" test organisms may have very low chemical concentrations in their tissues compared with organisms in nature, and experimental chemical loads on plastics are often much higher than those in environmental samples (Koelmans 2015). In one of the more environmentally relevant studies thus far, in which laboratory fish were fed contaminated food, contaminated food mixed with virgin plastics, or contaminated food mixed with environmentally contaminated plastics, bioaccumulation of chemicals from plastics occurred (Rochman et al. 2013c). This study also demonstrated an adverse biological response (liver stress) in fish for diets that included plastics, and that the response was amplified for plastics with sorbed contaminants. Because the plastics used in this experiment were contaminated in the natural environment (three-month exposure in seawater), this experiment used environmentally relevant concentrations on the plastic (albeit in laboratory fish) and also replicated exposure to a complex mixture of chemicals rather than a single chemical in isolation. Because of the practically innumerable potential mixtures of hazardous chemicals that might be associated with plastic debris and the multitude of environmental factors governing their transfer into marine organisms, generalizing the biological impact of this type of contamination may not be possible. However, a well-designed risk assessment for particular organisms or habitats and particular plastic types and chemicals could be useful to quantify harm and inform management strategies.

The third class of encounters of marine organisms with plastic debris is classified here as interaction; it includes nonentangling contact with debris as well as other specific interactions between debris and organisms. Fishing gear has been shown to cause tissue abrasion and breakage when colliding with sessile invertebrates in a coral reef ecosystem, and a variety of plastic and nonplastic debris items on the seabed have caused changes to ecological assemblages (i.e., through the colonization of debris and the use of objects as refuge) and death by suffocation upon contact (**Table 1**). It is hypothesized that seafloor debris acts as a barrier, preventing light penetration (Uneputty & Evans 1997), reducing the exchange of oxygen, and preventing the delivery of settling organic matter to sediments, with consequences for marine life (Green et al. 2015). And on beaches, correlative evidence suggests that litter could obstruct turtle hatchling migration to the ocean (Özdilek et al. 2006) and ghost crab burrowing activity (Widmer & Hennemann 2010).

Floating anthropogenic debris has long been known to serve as a substrate for rafting organisms ranging from microorganisms to sessile and mobile invertebrates, and it is also known to attract swimming animals that aggregate below the debris [see the review by Kiessling et al. (2015)]. Microbial communities on floating plastic fragments differ from one another and from those in surrounding seawater (Zettler et al. 2013), suggesting that the presence of this substrate affects ecological assemblages. Long-distance transport of floating debris with associated organisms is known to occur (e.g., Calder et al. 2014), and the establishment of nonnative or potentially invasive species transported by floating debris has been hypothesized but not yet demonstrated (Rochman et al. 2016).

In total, 70 cases of demonstrated biological impacts resulting from encounters with plastic marine debris have been identified (**Figure 4**, **Table 1**). Of these, 45 responses occurred at suborganism levels, 23 at the organism level (i.e., death of individuals), and 2 at the assemblage level. Correlative evidence supports an additional 7 impacts, including all impacts affecting population size. The majority of impacts were due to ingestion of plastic debris, which were demonstrated for both small debris (<1 mm in size; laboratory experiments only) and large debris (observational samples only). All but two studies of impacts caused by entanglement were from field observations of mostly large stranded animals, whereas impacts caused by nonentangling contact with debris were demonstrated from a combination of environmental data and manipulative field experiments.

The lack of evidence of biological impacts of plastic marine debris is apparent in **Figure 4**, but this should not be interpreted as a lack of impacts. In only one case did Rochman et al. (2016) find that a particular impact was hypothesized and properly tested but not found [a study by Browne et al. (2008), who observed laboratory ingestion by and translocation of micron-sized plastic particles in mussels without significant short-term effects on the animals]. Rather, in most cases the necessary studies to test more ecologically relevant impacts (e.g., at the population level) have yet to be done. It may not be necessary to fill the matrix of **Figure 4** in order to answer important questions. Browne et al. (2015b) proposed using adverse outcome pathways to infer linkages between contamination and demonstrated impacts from suborganism to population levels of biological organization. Given the multiple stressors in the natural environment, it may be difficult to tease apart the ecological impacts caused solely by plastic marine debris. However, there is already clear evidence of impacts on individuals, and models predicting population size and growth rate that incorporate environmental data on habitat conditions, life history, and exposure to contamination may also be useful to quantify impacts on a particular population (Browne et al. 2015b).

4. RISK ANALYSIS

As discussed above, substantial advances have been made in the scientific understanding of marine plastics. Although many fundamental questions remain about the amount and distribution of plastic debris and its biological impacts on populations and ecosystems, there is ample evidence of widespread contamination by plastics in forms that present serious hazards to organisms, with the likelihood that plastic input to the marine environment will continue to increase with time. Risk assessment is one available tool to use existing information, including observational and experimental data as well as statistical and process models, to evaluate the relationships between hazards and impacts in a way that can guide the design of prevention or mitigation measures (US EPA 1998). The risk assessment framework is, in principle, quite simple: The risk, or probability of a particular adverse outcome, is a product of the exposure to a hazard and the adverse response to the hazard, which is a function of the exposure amount. The challenge lies in quantifying these parameters using limited data, especially when investigating hazards or populations spanning large spatial scales, or hazards with a wide range of potential effects, as with plastic marine debris. These challenges are substantial, but several informative spatial risk analyses have recently been carried out.

To evaluate the risk of entanglement of sea turtles by derelict fishing nets in the Gulf of Carpentaria (Australia), Wilcox et al. (2012) used numerical models of surface ocean currents together with beach cleanup data on the occurrence of derelict fishing nets to predict the spatial distribution of drifting nets. They used the best available data (bycatch records from a prawn trawl fishery) to estimate the spatial distribution of sea turtles and then computed the probability of sea turtle encounters with derelict nets as the product of these two fields. In the absence of experimental data about the response by sea turtles upon encountering derelict nets, they assumed that an encounter (exposure) resulted in an entanglement. The risk model, which predicted previously unknown high-risk areas, was then validated by comparison with independent data on entanglements from stranded sea turtles.

A similar approach was taken to assess the global risk of plastic ingestion by sea turtles (Schuyler et al. 2014) and seabirds (Wilcox et al. 2015). As in the study by Wilcox et al. (2012), these studies utilized physical models of surface ocean circulation, but with time- and space-dependent inputs of plastic waste, to calculate the distribution of floating plastic debris. To estimate exposure to debris, they used debris concentration together with maps of species-specific habitat (for turtles) and range (for seabirds). However, in contrast to the approach of Wilcox et al. (2012), Schuyler et al. (2014) used a logistic regression model to predict the risk, or probability of plastic ingestion, based on the life history stage, species, and mean debris density at the time and location of stranded or bycatch turtles that had ingested plastic. They found that although debris exposure (or encounter) was a significant factor in the risk prediction model, encounter alone was not a sufficient predictor of debris ingestion. Similarly, Wilcox et al. (2015) found that the best-performing risk prediction model included seabird genus, body size, date of study, and sampling method, in addition to exposure.

The risk assessment framework formalizes the obvious notion that where there is no exposure (or encounter with the hazard), there is no risk. However, risk analysis can uncover potentially unexpected patterns in risk distribution. For example, by the Wilcox et al. (2015) model, the highest risk of plastic ingestion to seabirds is not in subtropical gyres, where high concentrations of debris are known to occur, but rather in the Southern Ocean, where debris concentrations are relatively low but the number of seabird species is very high. Similarly, an analysis using a framework that was similar but designed to evaluate optimal locations to remove floating debris in order to minimize ecosystem impacts (crudely represented by the spatial overlap between primary production and debris concentrations) found that collection would be most effective off the coast of China and in the Indonesian archipelago near large sources of debris from land, rather than in the high-plastics-concentration subtropical gyres (Sherman & van Sebille 2016).

As in the study by Sherman & van Sebille (2016), risk assessment models can provide guidance in the design of effective and resource-efficient management measures. The sea turtle entanglement risk analysis by Wilcox et al. (2012) predicted a common drift pathway for derelict fishing nets entering the Gulf of Carpentaria. If nets could be intercepted near the typical entry point, the exposure to hazardous nets, and therefore the risk of entanglements in downstream regions of high turtle density, would decrease. Although not strictly on marine debris, a risk assessment study of seal bycatch identified different mitigation strategies for each of two fisheries off South Australia (Goldsworthy & Page 2007). In the gillnet fishery, where several high-risk sea lion subpopulations were located within a fishing area that accounted for less than 10% of total fishery effort and total catch, the recommendation was to reallocate fishing effort. For the lobster trap fishery, gear modifications were proposed to reduce bycatch risk without the consequence of a fishery catch reduction. Finally, not to be overlooked is the utility of the same risk assessment models in evaluating the success of implemented management actions (Goldsworthy & Page 2007).

5. CONCLUSIONS

It is widely recognized that standardized sampling methodology and reporting are critically lacking in the detection, quantification, and characterization of plastic debris in the marine environment. We must develop robust and efficient methods to determine plastics distribution on coastlines, in the water column, in sediments, and on the seafloor. This will require determining the size-frequency distribution of plastic debris, from nanoparticles to large debris such as derelict fishing gear and debris from natural disasters, which will also address questions about sources, transport, and transformations of plastics as well as exposure and risk for particular marine organisms or habitats. Especially for ocean plastics, existing platforms such as ships of opportunity or autonomous vehicles could be exploited for widespread and efficient data collection if in situ plastic particle detection technologies were developed. On coastlines worldwide, informed and motivated citizen scientists already participate in beach cleanups (e.g., Ocean Conserv. 2014); perhaps there is potential to expand the scope or frequency of these volunteer efforts to collect additional data on spatial or temporal patterns of plastic debris accumulation. Finally, as important now as in the earliest days of ocean plastics research are the discovery and reporting of plastic particles in environmental samples collected for other purposes. A sharp eye for plastics in biological samples (such as marine aggregates or fecal pellets), sediment and sea ice cores, particle traps, and deep-water samples could provide valuable clues in the challenging mystery of the fate of plastics in the sea.

As scientific attention focuses on smaller and smaller particles, it is rapidly becoming apparent that plastic debris is everywhere—in lakes and streams, in soils and sand, in our homes, and in the air we breathe. Whether this ubiquitous presence poses a risk to human health remains to be determined and warrants further study (see, for example, Vu & Lai 1997 on human health risks of exposure to synthetic fibers). The great successes of polymer science have produced materials that are unmatched in their utility, low cost, and versatility, but their persistence in the environment and a lack of careful consideration of their end-of-life management have led to environmental problems. The ultimate solution to environmental plastic pollution is to prevent contamination in the first place, first and foremost by a reduction in use, followed by capture and reuse, recycling, and energy recovery (Koelmans et al. 2014b), which will hopefully result in less new plastic being produced and progress toward a more circular and sustainable economy.

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The author is not aware of any affiliations, memberships, funding, or financial holdings that might be perceived as affecting the objectivity of this review.

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