IEM Committee

From: Conservation Department <conservation@pacificwhale.org>
Sent: Monday, December 03, 2018 3:47 PM
To: IEM Committee
Subject: Testimony on Plastic Bag Reduction
Attachments: PWF testimony on plastic bag exemption.pdf; Currie et al (2017) - Quantifying the risk that marine debris poses to cetaceans in coastal waters of the 4-island region of Maui.pdf; Currie et al (2018) - Nearshore sea surface macro marine debris in Maui County, Hawaii.pdf; Law paper - plastics in the marine environment.pdf

Dear Maui County Council Committee Members,

Re: A BILL FOR AN ORDINANCE AMENDING CHAPTER 20.18, MAUI COUNTY CODE, RELATING TO PLASTIC BAG REDUCTION

In 2016 Maui County passed a plastic bag reduction ordinance reducing the distribution of plastic bags on our island. Currently, there is an exemption that allows thick plastic bags to be provided by retailers because they are considered multi-use. Unfortunately, these bags still take thousands of years to biodegrade and are often treated as single-use plastics. We offer our public testimony **in support of** the removal of these exemptions.

Please find attached our testimony on this issue and three supporting research papers.

Thank you for your consideration,
Jenny Roberts

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Aloha Maui Environment and Infrastructure Committee,

The mission of Pacific Whale Foundation is to protect the ocean through science and advocacy and to inspire environmental stewardship. We fully support the County's plastic bag ordinance passed in 2016 and greatly appreciate your efforts to help reduce single-use plastics on our island throughout the years and currently. We write to provide our **support** for removing exemptions of the plastic bag ban.

The current exemptions for the plastic bag ordinance include sturdy, thick plastic bags designed for multiple re-use, which scientific research shows is just as harmful, if not more, to our environment. Estimates show that 100 billion plastic bags are used by Americans each year. These plastic bags cannot be recycled on our island, are harmful to marine life, and, according to the Center for Biological Diversity, will take at least 1,000 years to degrade in a landfill. Plastics do not biodegrade; instead, they go through a process called photodegrading, meaning they will break up into smaller and smaller pieces, but never truly are gone from the environment. Scientific studies estimate that 100,000 marine animals die from plastic ingestion and entanglement each year, such as a male pilot whale in Thailand recently found deceased with 80 plastic bags in its stomach.

There is a growing body of scientific research demonstrating the negative impact that plastics have on over 700 different species of marine animals (Law, 2017). Specifically here in Maui County waters, a study performed by Currie *et al.* (2017) found plastic debris in all study areas within the 4-islands of Maui County, which also happens to overlap with important habitat for marine life, such as humpback whales, and endangered species such as sea turtles, Hawaiian monk seals, and the Main Hawaiian Islands insular population of false killer whales. This study also found high concentrations of plastic debris where the Au'au, Kealaikahiki, and Alalakeiki channels converge, raising a significant threat of entanglement and ingestion to our marine life which uses these areas.

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A recent baseline study published by Currie *et al.* (2018) in Hawaii found 90% of nearshore macro debris in Maui Nui to be plastics. This further supports the need to reduce sources for plastic pollution, as would be possible through removing exemptions of the current plastic bag ban.

Plastic debris is becoming a looming and increasing threat in today's world, with over 220 million tons of plastic produced each year. Pacific Whale Foundation asks all of its members and supporters to help keep our single-use plastics to a minimum here in Maui County by supporting the proposal to reduce exemptions to the 2016 plastic bag ordinance.

We believe the proposal to reduce exemptions for this current plastic ban bag should be passed and will benefit the ocean, environment and the Hawaiian Islands immensely.

Sincerely,
Jenny Roberts
Conservation Coordinator for Pacific Whale Foundation

Literature Cited:

Currie, J.J, S.H. Stack, J.A. McCordic, and G.D. Kaufman. (2017). Quantifying the risk that marine debris poses to cetaceans in coastal waters of the 4-island region of Maui. *Marine Pollution Bulletin* 121: 69-77.

Currie, J. J., Stack, S. H., Brignac, K. C., & Lynch, J. M. (2018). Nearshore sea surface macro marine debris in Maui County, Hawaii: Distribution, drivers, and polymer composition. *Marine Pollution Bulletin* 138: 70-83.

Law, K. L. (2017). Plastics in the marine environment. *Annual Review of Marine Science* 9: 205-229.

Marine Pollution Bulletin 121 (2017) 69–77

Contents lists available at ScienceDirect

Marine Pollution Bulletin

journal homepage: www.elsevier.com/locate/marpolbul

Quantifying the risk that marine debris poses to cetaceans in coastal waters of the 4-island region of Maui

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ARTICLE INFO

Keywords:
Marine debris
Hawaii
Entanglement
Cetaceans
Risk

ABSTRACT

Marine debris poses considerable threat to biodiversity and ecosystems and has been identified as a stressor for a variety of marine life. Here we present results from the first study quantifying the amount and type of debris accumulation in Maui leeward waters and relate this to cetacean distribution to identify areas where marine debris may present a higher threat. Transect surveys were conducted within the 4-island region of Maui, Hawai'i from April 1, 2013 to April 15, 2016. Debris was found in all areas of the study region with higher concentrations observed where the Au'au, Kealaikahiki, and Alalakeiki channels converge. The degree of overlap between debris and cetaceans varied among species but was largest for humpback whales, which account for the largest portion of reported entanglements in the 4-island region of Maui. Identifying areas of high debris-cetacean density overlap can facilitate species management and debris removal efforts.

1. Introduction

Marine debris, classified as any solid material from man-made origin that enters the marine environment (Coe and Rogers, 1997), presents a serious hazard to ocean habitats across the world. Marine debris poses considerable threat to marine life, biodiversity, and ecosystems (Sheavly and Register, 2007) and has been identified as a stressor for a variety of marine life (Moore, 2008).

The wide distribution of marine debris in conjunction with the low recovery probability of marine mammals that have ingested debris makes debris interactions difficult to quantify. Understanding the risk that marine debris poses to cetaceans in specific regions requires an understanding of the distribution of both the debris as well as the species of concern, which can be used to identify the potential risk for interaction. Debris items, particularly plastics, threaten marine organisms either indirectly by altering habitat or directly through fatal interactions (Wallace, 1985; Carr, 1987; Laist, 1997; Henderson, 2001; Gregory, 2009; Moore et al., 2009; Hong et al., 2013). An estimated 100,000 animals die each year from either ingesting or becoming entangled in debris (Wilks, 2006). Among these are several recorded instances of cetaceans which have died from such interactions (false killer whales: Oleson et al., 2010; minke whales: Pierrepont et al., 2005; pygmy sperm whale: Stamper et al., 2006; beaked whales: Simmonds and Nunny, 2002; harbor porpoise: Baird and Hooker, 2000). With a steady increase in the number of interactions between cetaceans and marine debris (Baulch and Perry, 2014), there is a growing need to understand and assess the risk that debris poses to these species.

Debris entanglement and ingestions have been documented for cetaceans in Hawaiian waters with 55 entanglements with marine debris reported by Bradford and Lyman (2015) from 2007 to 2012. Two of these instances involved Hawaiian spinner dolphins, one of which had a plastic ring/band around its rostrum preventing the mouth from opening. Another instance involved a juvenile humpback whale entangled in over 21 different types of rope and netting. Ingestion of debris is often an underreported metric as it often requires recovery and necropsy of dead animals. However, several instances of ingestion of debris by cetaceans in Hawaiian waters have also been reported (Laist, 1997). To date there has been no published work on the quantification of marine debris and potential interaction with marine mammals in the four-island region of Maui, Hawaii, an area which consists of a large portion of the Hawaiian Islands Humpback Whale National Marine Sanctuary (HIHWNMS).

In this paper we quantify the abundance and distribution of marine debris within the 4-island region of Maui and relate this to potential threats to four resident odontocete species and one migratory mysticete species. Such areas were determined by spatially overlaying the density of marine debris with the densities of each cetacean species, similar to the methods detailed in Williams et al. (2011). Effectively evaluating these threats requires the determination of "risk", or the likelihood that an undesirable event will occur (Harwood, 2000): in this instance the event being marine debris entanglement or ingestion. Williams et al.

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http://dx.doi.org/10.1016/j.marpolbul.2017.05.031
Received 29 March 2017; Received in revised form 12 May 2017; Accepted 13 May 2017
Available online 25 May 2017

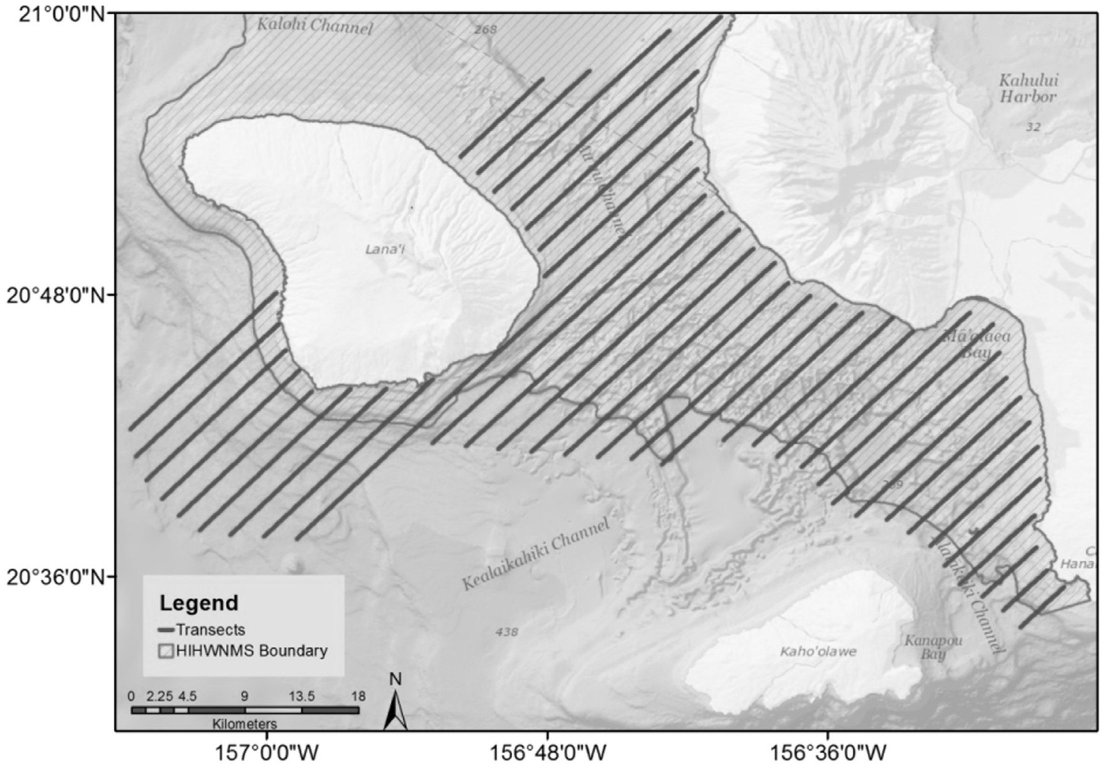

Fig. 1. Map depicting survey transects within the 4-island region of Maui, Hawaii.

(2011) note that the proximity between a particular species and marine debris is a key determinant of risk but does not necessarily result in ingestion or entanglement. As such, relative risk can be determined by multiplying the density of debris with the density of the study species, and the resultant overlap, or co-occurrence, of both a species and marine debris is the risk of interaction (Brown et al., 2015). This is the first study to quantify the potential interaction of marine debris and cetaceans in the Maui 4-island region with the following main objectives: 1) quantify the amount and type of debris accumulation in Maui leeward waters; 2) identify areas within these waters where marine debris may present a higher threat to cetacean species; with the aim of identifying areas where risk is elevated and guide potential mitigation and prevention strategies.

2. Methods

2.1. Study area and survey effort

Line transect surveys were conducted within the 4-island region of Maui, Hawai'i, consisting of the islands Maui, Molokai, Lana'i, and Kaho'olawe, between April 1, 2013 and April 15, 2016 (Fig.1). The starting point of each survey was chosen randomly at the beginning of each survey day. To ensure no missed occurrences of debris and cetaceans, locations of all sightings while both on- and off-effort were recorded during the study period. Survey effort varied my month and time of year covering an area of 1004 km^2 (Fig. 2). The study area consists predominantly of nearshore habitat < 200 m in depth. However, some areas south of Lana'i reached depths up to ~600 m. Survey lines were separated by 1 nautical mile and laid out perpendicular to the depth contours within the study area. Surveys were conducted onboard a 26-foot research vessel equipped with two outboard engines, departing from either Lahaina or Ma'alaea Harbors on Maui. As both on- and off-effort data were used, survey speeds ranged from a minimum of 5 knots when slowing down to pick up debris to 20 knots when transiting the survey area. On-effort surveys were conducted at a consistent speed of 15 knots.

To reduce detectability error, surveys were only conducted when Beaufort and Douglas Sea States were ≤ 3 (Tyson et al., 2011). Four individuals rotated through positions of observers and data recorder. One observer was stationed on the port and starboard sides of the helm, respectively, scanning equal sections of water from the bow to 90^0 on either side using a continuous scanning methodology (Mann, 1999) by naked eye and with reticle binoculars (7 × 50). The boat captain was also an on-effort observer, while the remaining personnel, including the data recorder, did not contribute to the scanning effort. Eye height of observer varied based on observer height, but ranged from 1.6 to 1.9 m. All sightings of both marine debris and of cetaceans were called out by the observers and logged by the data recorder. It is important to note that despite completing line transects, distance sampling was not completed for debris items and precludes traditional distance sampling analysis presented in Williams et al. (2011). As such the results presented here represented presence only sightings, which have not been correct for detectability.

2.2. Data collection

2.2.1. Cetaceans

Four resident odontocete species were recorded when present during the survey period: bottlenose dolphins (*Tursiops truncatus*), Hawaiian spinner dolphins (*Stenella longirostris*), pantropical spotted dolphins (*Stenella attenuata*), and false killer whales (*Pseudorca crassidens*). One migratory mysticete species was recorded when present from December to April during the survey period: humpback whales (*Megaptera novaeangliae*). Upon sighting the species, pod size and sighting location (latitude and longitude) were recorded.

2.2.2. Marine debris

All floating debris items encountered were sampled during the survey period. When a piece of debris was sighted, the item was collected if possible and GPS location (latitude and longitude), and the type of material were recorded. If the item could be collected, it was photographed and recorded. All debris items were classified into the

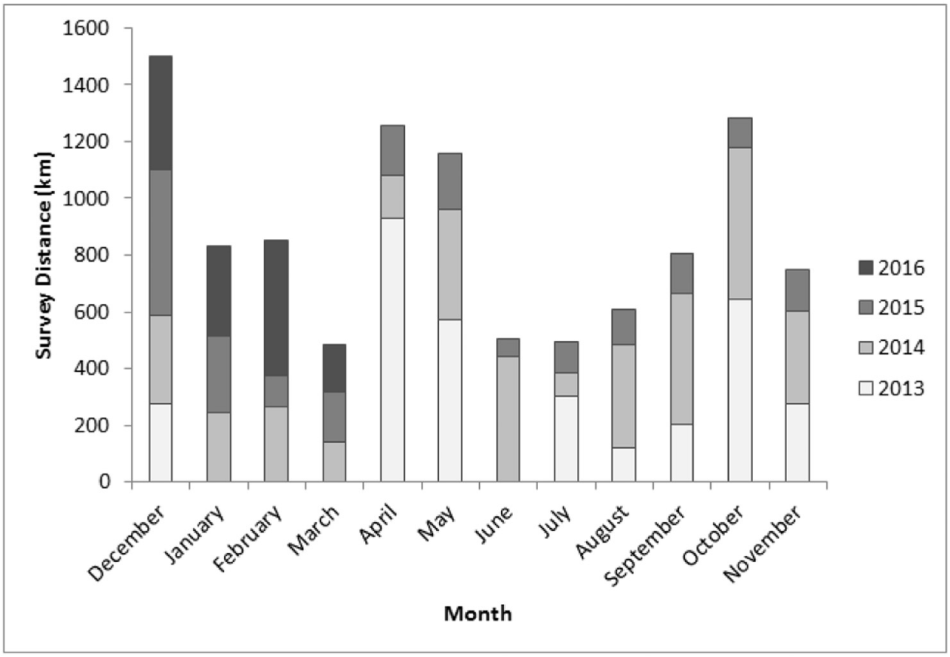

Fig. 2. Survey effort divided by month and year to show sampling effort.

following categories based on their material: plastic, metal, glass, rubber, clothing/fabric, processed lumber. Plastics were further classified into subcategories adapted from Eriksen et al. (2014). Plastic debris identifiable as fishing-related was divided into buoys, fishing line, rope, netting, and other fishing gear. All other plastic debris was categorized as containers (bottles, jugs, crates, etc.), foamed polystyrene, plastic bags and other soft plastic films, plastic fragments, and other plastics. To determine the origin of debris, items were divided into three indicator debris categories (general, land, ocean) based on their likely sources (Blickley et al., 2016). Ocean based debris represented items from recreational boating/fishing and/or commercial fishing activities. Land based debris represented items from land-based recreation, celebrations and dining. General-source debris represented items that could originate from either ocean- or land-based sources and could not be confidently classified into only a single of these categories.

To help quantify the differences in risk, debris was divided into two categories: (1) entanglement risk defined as debris comprised fully or partially of netting, rope, and/or line (2) ingestion risk defined as the remaining debris void of any trailing/entangling gear.

2.3. Spatial analysis

All marine debris and cetacean location data were imported into ArcGIS 10.3 (Environmental Systems Research Institute, 2012) and mapped with the World Mercator projection, using the WGS 1984 datum. The study area was divided into 1004 grid cells each with an area of 1 km^2 (1 km × 1 km). Each grid cell was classified by total distance surveyed. Grid cells with no survey effort were dropped before completing subsequent analysis.

2.3.1. Estimating density of marine debris and odontocetes

Debris density was estimated using the "point density" tool (spatial analyst) in ArcMap to create a density raster, quantifying the number of debris sightings per km^2. Cetacean sightings were analyzed by species. Density of each species was estimated using the "point density" tool (spatial analyst) in ArcMap to create a density raster quantifying the number of cetacean sightings per km^2. To account for potential survey effort bias, cetacean and debris sightings were weighted by distance surveyed per grid cell (1 km^2), assigning greater weights to sightings in grids that received less survey effort.

2.3.2. Assessing overlap of marine debris and cetaceans

To determine the co-occurrence of each cetacean species with debris, weighted density of debris was overlaid with the weighted density of each cetacean species. Then the product of weighted marine debris density and species density was calculated for each cell. This was then converted into a point layer using the "raster to point" tool (conversion) to create a point data layer representing co-occurrence.

2.3.3. Calculating relative risk

Risk areas were predicted for each species by estimating kernel density from the respective exposure point data layer using "kernel interpolation with barriers" tool (geostatistical analyst). Barriers to distribution included the islands of Maui, Molokai, Lana'i, and Kaho'olawe. The output cell size was set to 1 km^2, and the extent set to perimeter of survey area. Bandwidth was calculated using least-squares cross validation (Bowman and Azzalini, 1997) and estimated at 5320 m. The resulting estimates were binned into natural breaks using "Jenks" method in ArcMap and represent low and high risk areas for each species. As such, relative risk can be compared within a species but not between species.

3. Results

3.1. Survey effort

A total of 215 surveys were completed from April 1, 2013 to April, 15, 2016 covering 29,810 km of combined on- and off-effort survey distance (Fig. 3). A total of 45 bottlenose dolphin, 11 spinner dolphin, 22 spotted dolphin, 8 false killer whale, and 636 humpback whale pods were sighted along with 1027 pieces of marine debris.

3.2. Marine debris

Of the 1027 debris items collected, the majority could not be assigned as originating specifically from land or ocean sources (Fig. 4). Based on the shape, size and composition of debris, 88% (n 904) were considered to pose an ingestion risk while 12% (n = 123) were considered to pose entanglement risk.

Plastics were the predominant type of debris recorded within the study area, accounting for 86% of total debris (Fig. 5A). Processed

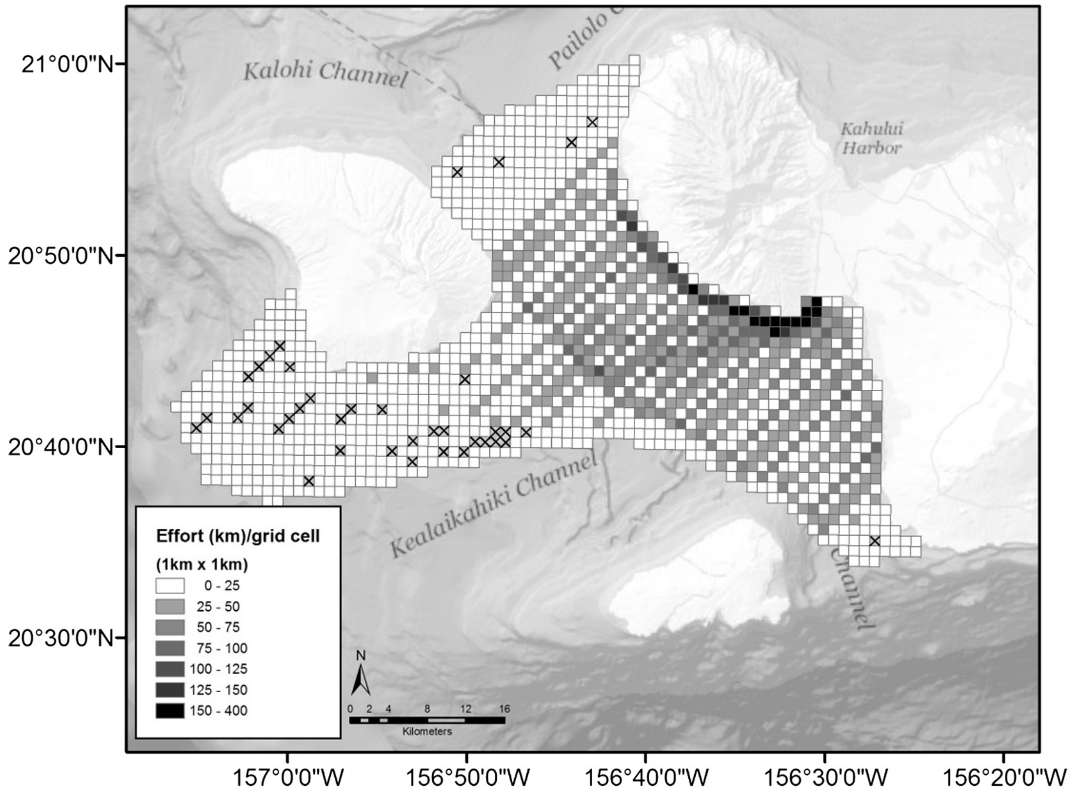

Fig. 3. Survey effort/grid cell conducted between April 1, 2013 and April 15, 2016 within the 4-island region of Maui, Hawaii. *Note*: Grid cells marked with (☒) represent areas with no effort and were not included in analyses.

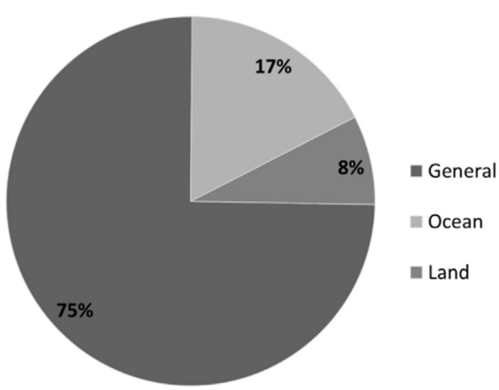

Fig. 4. Origin of marine debris collected between April 1, 2013 and April 15, 2016 within the 4-island region of Maui, Hawaii.

lumber and rubber accounted for 10% of debris, with the remaining 4% attributed to metal, glass and clothing/fabric (Fig. 5A). A small portion (13%, n = 156) of all plastic debris was fishing-related. Of these items, the majority were buoys (63%, n = 99). The remaining fishing-related debris consisted of netting (n = 25), other types of fishing debris (n = 10), ropes (n = 9), and fishing lines (n = 6). The majority of non-fishing related plastics consisted of plastic containers (23%, n = 259), followed by foamed polystyrene (n = 206), plastic fragments (n = 190), plastic bags and other soft plastic films (n = 189), and other plastics (n = 122) (Fig. 5B).

Of the debris collected, 58% (n = 600) exhibited some form of biofouling organisms, with plastics comprising the largest proportion (n = 550, 92%) of biofouled items.

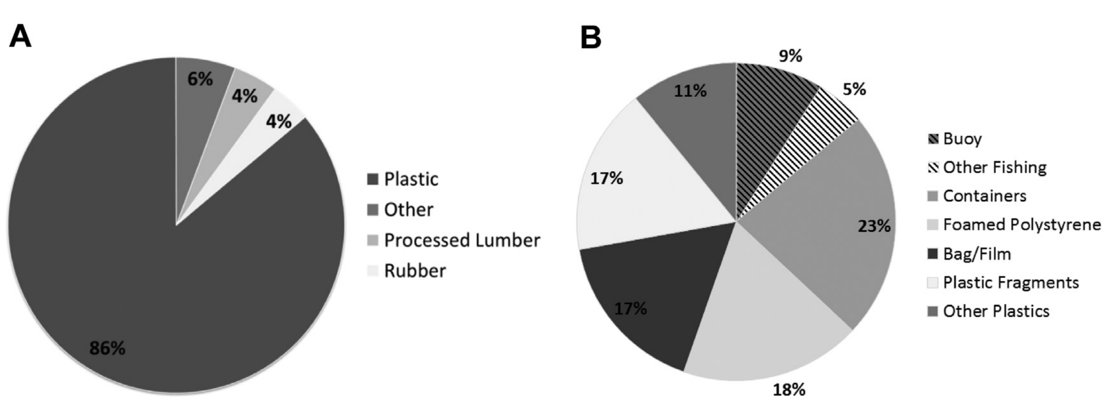

Fig. 5. (A) Type of marine debris collected between April 1, 2013 and April 15, 2016 within the 4-island region of Maui, Hawaii, and (B) subcategories of plastic debris collected between April 1, 2013 and April 15, 2016 within the 4-island region of Maui, Hawaii. Hatched areas indicate fishing-related debris, with "Other Fishing" including all ropes, fishing line, netting, and other fishing related plastic debris.

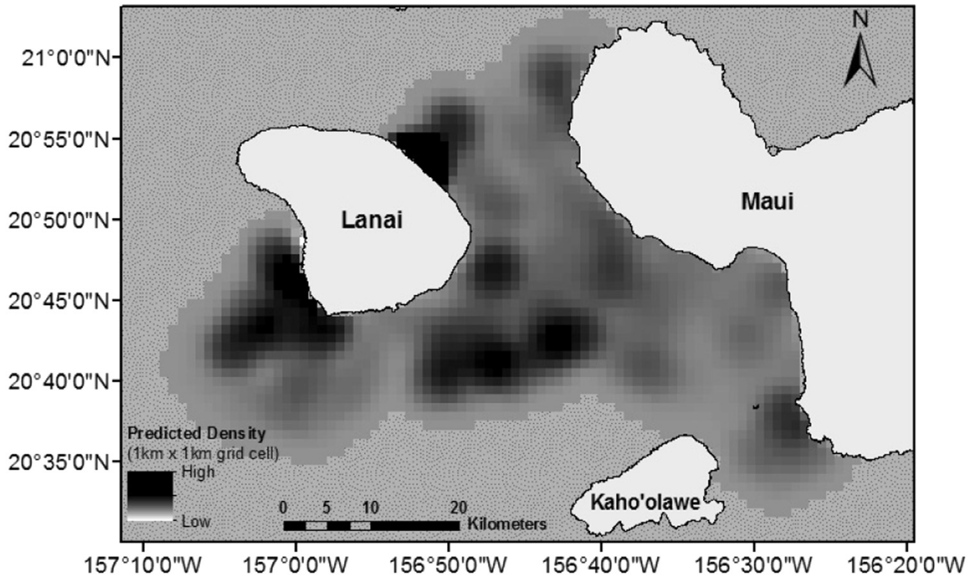

Fig. 6. Predicted weighted densities of marine debris observed April 1, 2013 and April 15, 2016 within the 4-island region of Maui, Hawaii.

3.3. Spatial analysis

3.3.1. Marine debris density

Marine debris was observed in all parts of the survey area. Kernel density estimates of debris showed a trend of higher accumulation between the islands of Maui, Lana'i, and Kaho'olawe in the area where the Au'au, Kealaikahiki, and Alalakeiki channels meet, as well as southwest of Lana'i (Fig. 6).

3.3.2. Cetacean-marine debris interaction risk

Maps were created for each cetacean species showing a density gradient from low density (white) to high density (black) to depict an increasing probability of cetaceans and debris occurring in the same grid cell. These maps may be used to identify both the area of relative risk for a species and the relative probability of an interaction occurring in that area.

3.3.3. Humpback whales

Risk of debris interaction with humpback whales showed highest concentrations between Ma'alaea and Lahaina harbors from near shore waters out to ~7 nautical miles (Fig. 7). The predicted risk for humpback whales covered an area of 827 km^2.

3.3.4. Bottlenose dolphins

Bottlenose dolphins had the largest area of interaction risk between debris and an odontocete species; second largest overall after humpback whales (Fig. 8). Risk was most prominent along the nearshore areas of southwest Maui, extending 10-15 km off shore. The predicted risk for bottlenose dolphins covered an area of 607 km^2.

3.3.5. False killer whales

Risk of debris interaction with false killer whales was concentrated in the center of the 4-island region, where the Au'au, Kealaikahiki, and Alalakeiki channels meet (Fig. 9). The predicted risk for false killer whales covered an area of 404 km^2.

3.3.6. Spotted dolphins

Spotted dolphins showed a clear concentration of high risk of interaction with marine debris off the southeast coast of Lana'i (Fig. 10). The predicted risk for spotted dolphins covered an area of 325 km^2.

3.3.7. Spinner dolphins

Spinner dolphins showed a clear concentration of high risk of interaction with marine debris off the southeast coast of Lana'i (Fig. 11). The predicted risk for spinner dolphins covered an area of

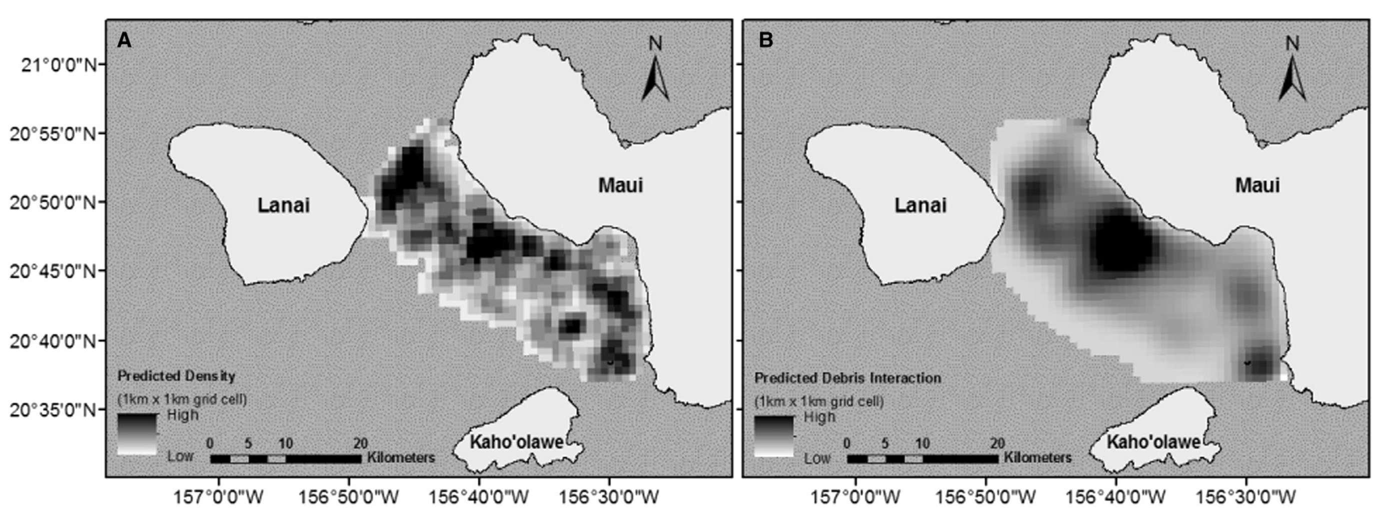

Fig. 7. (A) Predicted weighted density of humpback whales and (B) relative predicted marine debris-humpback whale interaction within the 4-island region of Maui.

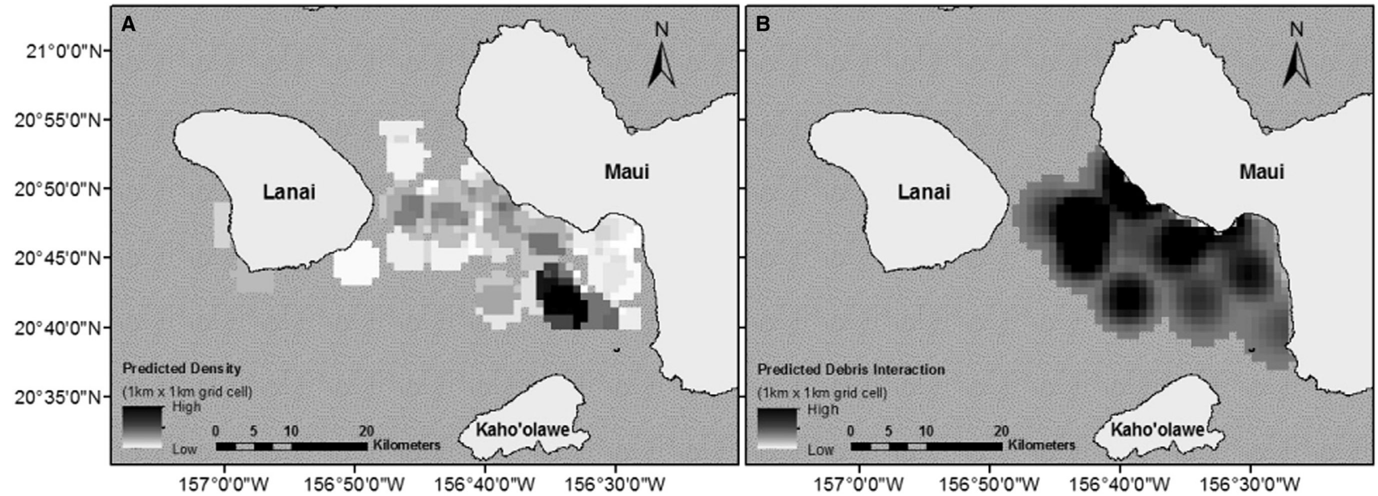

Fig. 8. (A) Predicted weighted density of bottlenose dolphins and (B) relative predicted marine debris-bottlenose dolphin interaction within the 4-island region of Maui.

325 km^2, with mostly low densities.

3.3.8. Other Species

Although not the focus of this research, the following species were also sighted during our surveys: short-finned pilot whale (*Globicephala macrorhynchus*), sperm whale (*Physeter macrocephalus*), Hawaiian monk seal (*Neomonachus schauinslandi*), green sea turtle (*Chelonia mydas*), wedge-tailed shearwater (*Ardenna pacifica*) Hawaiian petrel (*Pterodroma sandwichensis*), Laysan albatross (*Phoebastria immutabilis*), brown booby (*Sula leucogaster*), masked booby (*Sula dactylatra*), red-footed booby (*Sula sula*), tropicbirds (*Phaethon spp.*), and various species of sharks.

4. Discussion

4.1. Marine debris composition

Plastic comprised the majority of debris found in this study, a result that aligns with the known prevalence of plastics in the ocean (Coe and Rogers, 1997; Derraik, 2002). Buoyant and slow to degrade, plastics pose the biggest threat to marine mammals in terms of the risk of entanglement or ingestion of large debris items (e.g. Laist, 1997). When classified into subcategories, the majority of plastic debris items were not specifically related to fishing activities. Buoys comprised most of the fishing-related debris with rope, fishing line, and netting representing much smaller proportions. Plastic debris was dominated by plastic containers (e.g. bottles, tubs, baskets) and foamed polystyrene (e.g. disposable plates, cups, and miscellaneous broken pieces of foamed polystyrene). Although reported amounts do not account for size of debris—e.g. a single 10 m section of line and a single 1cmx1cm plastic fragment would have each been counted as one item—these relative proportions suggest that cetaceans face an overall higher risk of ingestion rather than entanglement within the Maui 4-island region. Odontocetes have been shown to be more susceptible to risk of ingestion of marine debris relative to other groups of marine mammals (Laist, 1997). Harmful effects of ingestion include reduced storage volume in the stomach, diminished feeding stimulus, and potential reproductive failure (Derraik, 2002).

Biofouling of debris may also make items more favorable for ingestion by some species. Plastics were found to be the highest biofouling category and as these items degrade in the marine environment, they can affect prey organisms at lower trophic levels (Andrady, 2011). Although indirect consequences of such bottom-up effects on marine mammals are much more difficult to quantify, the potential implications of this warrant further investigation.

4.2. Marine debris distribution

Ocean currents and circulations within the Maui 4-island region are extremely variable and dominated by eddies ranging from 50 to 150 km (Patzert, 1969). Eddies are relatively shallow in depth and surface flow around them can be in excess of 100 cm/s. Observed distribution of

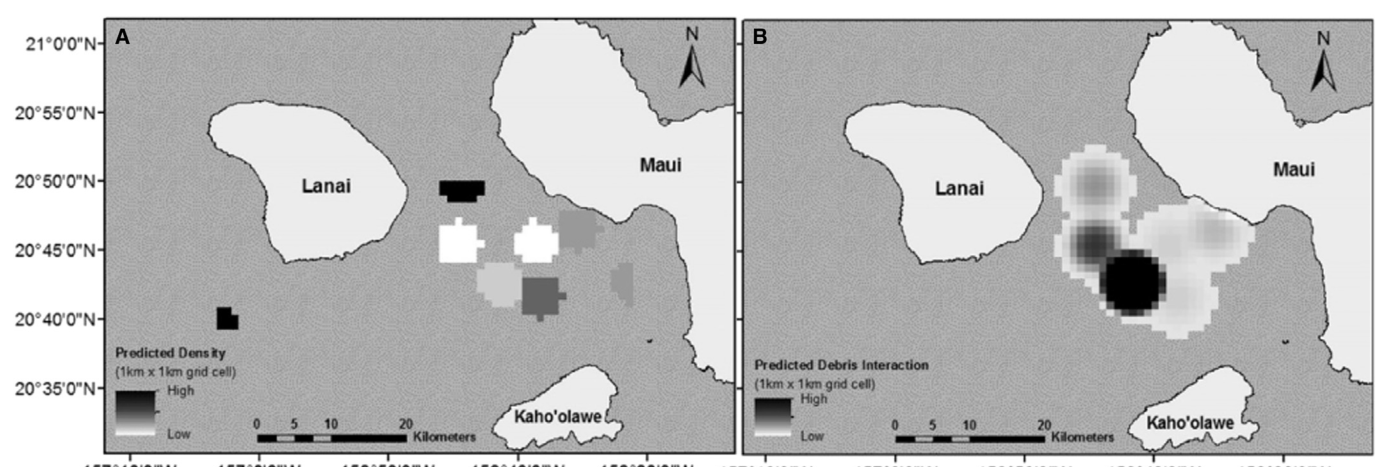

Fig. 9. (A) Predicted weighted density of false killer whales and (B) relative predicted marine debris-false killer whale interaction within the 4-island region of Maui.

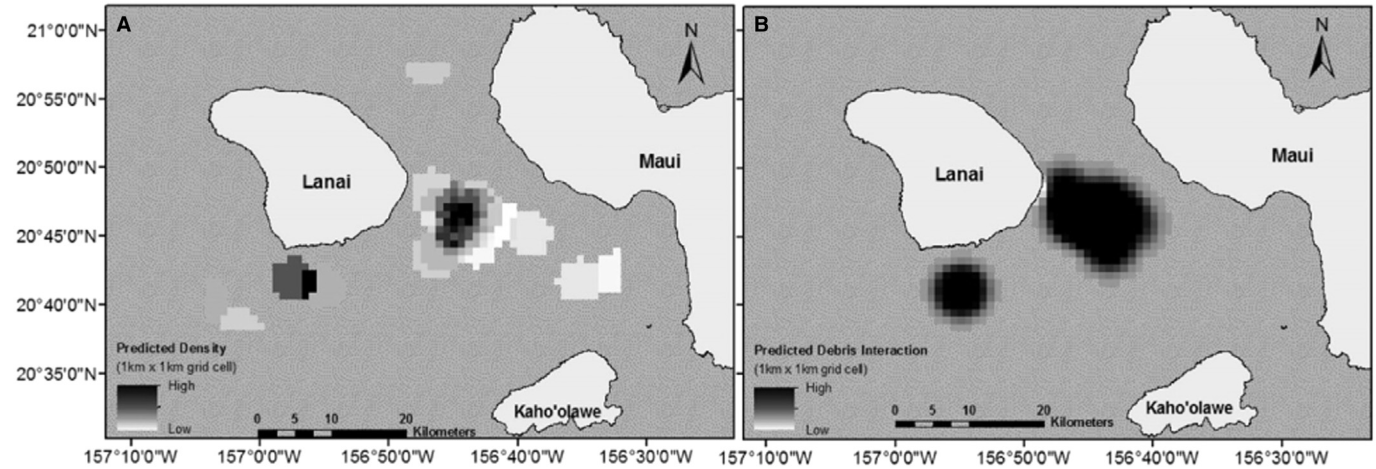

Fig. 10. (A) Predicted weighted density of spotted dolphins and (B) relative predicted marine debris-spotted dolphin interaction within the 4-island region of Maui.

debris is likely driven by local winds blowing through restricted passages between the islands as well as between Mauna Kahalawai and Haleakala volcanos on the island of Maui.

4.3. Mysticete distribution and overlap with debris

Humpback whales are found throughout the entire study area and had a large range of distribution. There is a large area outside Lahaina where there was a great deal of overlap between the distribution of debris and that of humpback whales, with a small site with high risk of interaction located off the south coast of Maui. The high concentration of humpback whales in the four-island region of Maui, Hawai'i likely accounts for the high risk of debris interaction observed and coincides with the high proportion of humpback whales in Hawai'i's report of marine debris entanglements (Bradford and Lyman, 2015).

4.4. Odontocete distribution and overlap with debris

Areas of overlap were found between marine debris distribution and that of all odontocete species encountered in this study. Although relative risk could not be compared among species, each species showed clear areas of high risk of interaction with marine debris. The locations of high risk areas varied across species and, when combined, covered a large portion of the survey area. The four encountered odontocete species display general preferences for certain types of habitats, but none of the species show strong site fidelity within the Maui 4-island region.

Bottlenose dolphins are found in relatively shallow waters in comparison to other odontocete species (Baird et al., 2003). “Hot spots” of higher risk followed this pattern and were accordingly concentrated along the coast of Maui from Ma'alaea Harbor to Lahaina Harbor.

For false killer whales the highest-risk areas were centered between the islands of Maui, Lana'i, and Kaho'olawe, and off McGregor Point, Maui. In Hawaii, these animals have been observed in both shallow (< 200 m) and deep (> 2000 m) waters and move extensively between the main Hawaiian Islands (Baird et al., 2008). Threats to this population are numerous, and the insular (island-associated) population of Hawaiian false killer whales is listed as endangered under the U.S. Endangered Species Act. The most recent abundance estimate for Hawaiian insular false killer whales using mark-recapture photo-identification data from 2000 to 2004 is 123 individuals (CV = 0.72) (Baird and Gorgone, 2005). When compared with other stocks, these abundance estimates indicate that insular false killer whales may have the smallest population size of any odontocete species within the Hawaiian Economic Exclusive Zone (Barlow, 2006). Given the current state of the population, any risk of debris interaction poses a threat to the viability of the population and highlights the need to address the removal of debris within the Hawaiian archipelago.

Pantropical spotted dolphins showed a fairly strong area of overlap with marine debris in the area centered between the islands of Maui, Lana'i, and Kaho'olawe. Spotted dolphins prefer slightly deeper waters than the other odontocete species discussed (Baird et al., 2003), perhaps explaining the second area of high risk for this species in

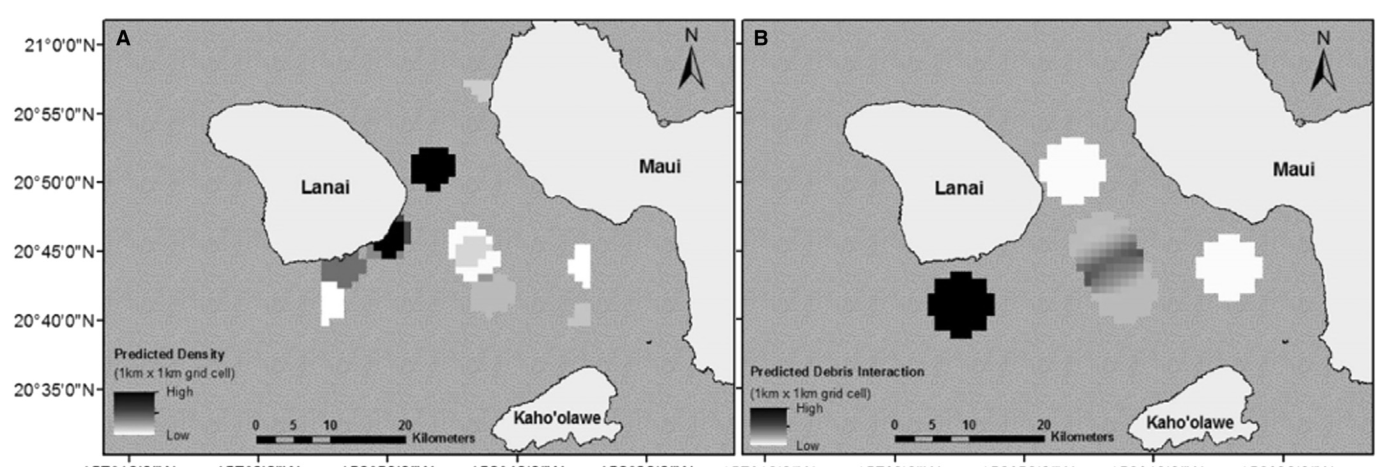

Fig. 11. (A) Predicted weighted density of spinner dolphins and (B) relative predicted marine debris-spinner dolphin interaction within the 4-island region of Maui.

deeper waters south of Lana'i.

Hawaiian spinner dolphins showed an area of relatively high risk south of Lanai, with smaller low risk areas observed through the species sighting range. Spinner dolphins rest nearshore and in bays during the day and forage offshore at night (Thorne et al., 2012). Our survey efforts occurred during daylight hours, likely minimizing their potential distribution within the study area during surveys. For this reason it is difficult to quantify the actual risk toward this species as the results represent minimal potential risk.

4.5. Other species

In addition to the 5 cetacean species mentioned in this report, sightings of rough-toothed dolphins (*Steno bredanensis*), dwarf sperm whales (*Kogia sima*), melon-headed whales (*Peponocephala electra*), and short-finned pilot whales have also been reported for the Maui 4-island region (Baird et al., 2013). The observed wide-scale distribution of marine debris has implications for any species utilizing the Maui 4-island region as it represents a potential for entanglement or ingestion.

5. Conclusion

Overall, the highest-risk area across all species, except spinner dolphins, was the area centered between the islands of Maui, Lana'i, and Kaho'olawe. The area we have identified as highest concern warrants further study, aimed at reducing the risk to cetaceans by reducing debris input and mitigating the impact of existing debris. Further management measures, particularly those aimed at endangered species such as false killer whales, would incidentally help all species sharing the same habitat. Bottlenose, spotted, and spinner dolphins show evidence of island-associated stocks with limited movement between islands (Baird et al., 2001, 2003, 2009; Andrews et al., 2010). Although these species are not currently at risk of extinction, recovery potential for Maui populations may be limited due to the relative isolation from other portions of the species' range. The endangered false killer whales should be a priority species for additional research as their abundance, biology, and ecology in Hawai'i remains poorly studied. Numerous species of sea turtles and Hawaiian monk seals are endangered species not included in this study that would additionally benefit from a reduction in marine debris in Hawaiian waters. The origins of debris presented here should be considered when determining the focus of conservation efforts to reduce debris accumulation. Additional research should focus on the cause and distribution trends of marine debris within the 4-island region of Maui, Hawai'i.

Acknowledgements

The authors would like to thank Eilidh Milligan and Rachael Tancock for their help with initial mapping and plotting of species and debris densities. Our gratitude goes to our volunteers and all research interns between 2013 and 2016 who helped with data collection. Finally we wish to thank Pacific Whale Foundation's members and supporters for providing the funding necessary to conduct this study. This research was conducted under NMFS permit #16479 and LOC#18101.

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Marine Pollution Bulletin 138 (2019) 70–83

Contents lists available at ScienceDirect

Marine Pollution Bulletin

journal homepage: www.elsevier.com/locate/marpolbul

Baseline

Nearshore sea surface macro marine debris in Maui County, Hawaii: Distribution, drivers, and polymer composition

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ARTICLE INFO

Keywords:
Plastic pollution
Monitoring
Accumulation
Polyethylene
Ocean surface currents

ABSTRACT

Located within the subtropical convergence zone, the Hawaiian archipelago is subject to high debris loads. This paper represents the first study to determine the spatial and temporal trends of floating macro debris quantities and polymer composition within Maui County waters. Ocean surveys were conducted from 2013 to 2017 and collected 2095 debris items of which 90% were plastic. Attempts to categorize items by source resulted in only 6% likely from land, 12% from ocean-based sources, 50% from either land or ocean, and 32% from unknown sources. Results found a multi-step process for debris accumulation, with temporal trends linked to survey day and year and spatial trends linked to ocean processes. High- and low-density polyethylene and polypropylene accounted for the majority of polymer types. The results of this study demonstrate minimal debris in Maui originates from land/local sources, and the importance of baseline data to guide further research and mitigation measures.

Marine debris poses a considerable threat to marine life, biodiversity, and ecosystems (Sheavly and Register, 2007; Galloway et al., 2017) and has been identified as a stressor for a variety of marine life (Moore, 2008; Currie et al., 2017). Marine debris can be classified into three categories describing its likely source: land, ocean, and "general", which encompasses both or either land and ocean, as described by Ribic et al. (2012). Previous research has identified ocean-based debris as the primary source of Hawaiian marine debris (Donohue et al., 2001), with proportionately higher ocean-based debris when compared to other regions in the Pacific (Ribic et al., 2012). Once reliable data on where marine debris originates and how it is introduced into the marine environment is available, targeted efforts to stop this problem at the source can be implemented.

Current knowledge of ocean currents in the North Pacific suggests three high-density areas of debris accumulation based on convergence zones (Wakata and Sugimori, 1990; Kubota, 1994; Van Sebille, 2015). One such zone is located just north of the Hawaiian Islands, and has been found to accumulate debris (Donohue et al., 2001; Pichel et al., 2007; Goldstein et al., 2013) (Appendix Fig. 1). The origins of debris north of Hawaii varies greatly, and the resulting accumulation is the result of multi-step processes starting with the Ekman convergence zone, transport via the geostrophic currents, and finally Ekman drift (Kubota, 1994). Marine debris accumulating north of the Hawaiian archipelago can travel through various marine ecosystems including coastlines, remote islands, the open ocean, and subtropical gyres (Derraik, 2002; Barnes et al., 2009). Some work has been conducted to document the rates and process of marine debris accumulation in the Northwestern Hawaiian Islands (Kubota, 1994; Donohue et al., 2001; Dameron et al., 2007; Pichel et al., 2007), but these efforts have been minimal and rates are likely out of date. Further, this work does not include the coastal waters of the Main Hawaiian Islands.

Currie et al. (2017) presented the first study quantifying the amount and type of marine debris in the nearshore waters of Maui county and related this to cetacean distribution to identify areas where marine debris ingestion or entanglement may present a high risk to marine mammals. Ingestion and entanglement of marine debris by biota has been well documented (Kühn et al., 2015), and effects of plastics on marine life are polymer dependent (Rochman et al., 2013). The current study expands the previous study (Currie et al., 2017) by performing polymer identification and statistical models to find local and ocean-wide variables that explain the accumulation of plastic marine debris floating in Maui County's nearshore waters.

Plastic marine debris is comprised of many different polymers that have specific chemical compositions defining their physical and chemical properties, leading to different environmental fate and effects. Polymer composition of marine debris items will affect vertical

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https://doi.org/10.1016/j.marpolbul.2018.11.026
Received 15 August 2018; Received in revised form 7 November 2018; Accepted 13 November 2018

stratification in the water column and influence interactions with marine organisms (Jung et al., 2018). Debris collected from the surface is expected to consist of floating low-density polymers such as polyethylene (PE) and polypropylene (PP) as opposed to sinking high-density polymers such as polyethylene terephthalate (PET) (Jung et al., 2018). As plastic marine debris is a growing problem in Maui County, a need for identifying polymer type is crucial for understanding the behavior of debris in the ocean environment and to know which polymers whales, dolphins, and other marine life will be exposed to in order to reduce their impact.

Blickley et al. (2016) monitored shoreline debris in Maui and found debris loads were linked to ocean based processes such as winds and currents as well as beach exposure and location. At a single site shoreline debris accumulation rates were as high as 1460 items per day within a 100 meter section, which was largely attributed to ocean based sources (Blickley et al., 2016). A clearer understanding of nearshore debris loads within the waters surrounding the Main Hawaiian Islands should be determine to supplement Blickley et al. (2016) and allow for a better understanding of which mitigation measures would be most effective for Hawaii. This study quantified macro (> 1 cm diameter) marine debris floating in the nearshore waters of Maui County as a complement to existing shoreline surveys. The objectives of this study were to: (1) identify factors influencing marine debris accumulation; and (2) characterize debris type including polymer composition, source (ocean, land, or general), and amount of marine debris within the study area.

The nearshore waters of Maui county that made up the study area (center: 20.73623°N, 156.69085°W) are semi-enclosed by the islands of Maui, Molokai, Lanai, and Kahoolawe and located within the Hawaiian Islands Humpback Whale National Marine Sanctuary. The channels between the islands are akin to drowned land bridges that once connected the surrounding islands. The study area consists predominantly of nearshore habitats with gently sloping shoreline gradients that extend to more complex bathymetry of seamounts and ridgelines (Grigg et al., 2002). The majority of the study area consists of drowned reef features and sandy basins with a depth of < 200 m; however, some areas south of Lana'i reach depths up to ≈600 m.

The detailed methods of data collection for this study are previously described in Currie et al. (2017), with a brief synopsis provided here. Line transects, separated by one nautical mile (1.85 km), were conducted within the leeward waters, up to 18 km from the coasts, of the four islands (Maui, Molokai, Lanai, and Kahoolawe) that comprise Maui County, Hawaii and covered 1004 km^2. Line transects were followed using the research vessel's onboard GPS. The starting point of each survey was chosen randomly at the beginning of each survey day. The transect lines, as followed using the onboard GPS, are presented in Fig. 1, with survey effort shown in Fig. 2. If debris was sighted during a transect, the transect was paused while the vessel changed course to pick up the debris. Once the debris was removed and documented, the vessel navigated back to the transect line and effort was resumed from the pause position. To ensure there were no missed occurrences of debris, all sightings of marine debris, regardless if on transect or travelling between transects, were recorded and used in subsequent analysis.

A minimum of one survey day/week was attempted and planned for the day with the best weather forecast, to allow observers the best conditions for visibility. If no suitable weather days (Beaufort and Douglas Sea States were > 3) were available, multiple surveys were conducted during the next suitable weather window. From April 6, 2013 to October 12, 2017, 767 line transect surveys for macro (> 1 cm diameter) floating debris were completed over 260 days from an 8 m engine-powered research vessel. The collection of debris for this project was done in conjunction with a systematic line transect study for odonotocetes, and it is important to note that despite conducting line transect surveys, distance sampling procedures were not followed for the debris items collected. Therefore, no effective transect width could be calculated and the survey width was limited to the sighting distance of the observer. As such, the results presented here represent presence-only sightings, which have not been corrected for detectability. The transect surveys ensured sufficient coverage of the survey area, but not adhering to systematic survey methods allowed for the highest number of debris items to be collected and was deemed most appropriate for this study.

All marine debris location data were imported into ArcGIS 10.6 (Environmental Systems Research Institute, 2012) and mapped with the World Mercator projection, using the WGS 1984 datum. To determine spatial trends in debris quantities over the entire duration of the study period, the study area was divided into 1004 grid cells each with an area of 1 km^2 (1 km × 1 km). Each grid cell was classified by the count of debris items occurring in that cell and the total survey distance (km) travelled within the boundaries of that cell from April 6, 2013 to October 12, 2017. Quantities of marine debris were summarized per grid cell by dividing the sum of debris counts by the sum of survey effort (km) within each grid cell, resulting in final units of number of debris items/km/grid cell, which henceforth will be referred to as spatial quantities. Grid cells with no survey effort were dropped and not displayed in the final spatial trend map.

All floating macro debris (> 1 cm diameter) items within sighting distance were recorded during the survey period. As such, results presented here likely represent an underestimation of all debris items because smaller items, particularly micro and nano debris items, were not sampled. Observations were undertaken by two experienced observers stationed on the port and starboard sides of the vessel, as well as the boat operator who was stationed at the helm using a continuous scanning methodology (Mann, 1999) by naked-eye or reticle binoculars (Bushnell 7 × 50), while a fourth person acted as a data recorder. No elevated observation platform was used and, as such, each observer's feet were standing ≈28 cm above the waterline. To ensure minimal influence of weather on detectability of debris, surveys were only conducted in the absence of rain and when Beaufort and Douglas Sea States were ≤3. However, there is likely some un-corrected influence of weather on the detectability of debris that should be acknowledged, and the debris counts recorded for this study may represent an underestimate of the true count. When a piece of debris was sighted, the item was collected (if size and conditions allowed) and latitude, longitude, type of material, and percent of organism coverage (biofouling) on the debris item were recorded. Percent organism coverage was determined by visual inspection of the debris item and estimating the proportion of biofouling with respect to total surface area (See Appendix Fig. 2). If the item could not be collected given the feasibility of removing it from the water, it was photographed and recorded but left in the ocean.

To be consistent with the national debris monitoring program, all debris classification was based on standardized source categories established by the United States Environmental Protection Agency (Escardó-Boomsma et al., 1995) and detailed in Ribic et al. (2012). Debris was divided based on the type of debris: plastic, metal, glass, rubber, clothing/fabric, processed lumber; and probable source category: ocean, land, general, or unknown. Ocean-based debris related to ocean recreation and commercial fishing; land-based debris related to land-based recreation and activities; general-sourced debris related to items that could originate from either ocean- or land-based sources (Ribic et al., 2012); and unknown-sourced debris consisted of debris fragments that could not be identified and therefore could not be reliably placed in a source category. The probable source categories (ocean, land, and general) were adapted from Ribic et al. (2012) and the debris item division used in this paper is presented in Appendix Table 1. It should be noted that debris was classified as best as possible in the most likely source category, but the potential for overlap between categories may exist and should be considered when interpreting results.

To gain a better understanding of the type of plastic debris that was collected throughout the survey period, all plastic debris items were

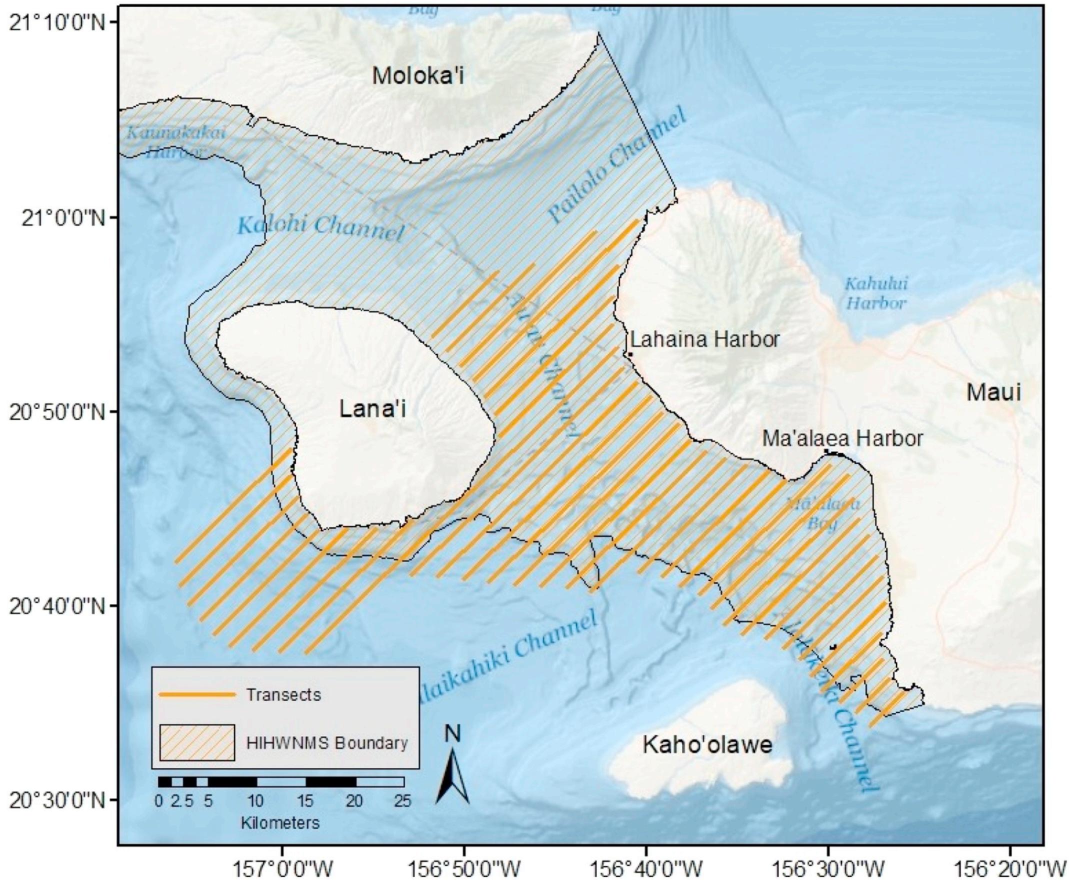

Fig. 1. Map depicting study area and line transects surveyed within Maui County, Hawaii from April 6, 2013 to October 12, 2017 and overlaid with the Hawaii Humpback Whale National Marine Sanctuary (HIHWNMS).
Note: The study area covers approximately ~20% of the HIHWNMS.

subclassified into nine main categories: foam fragments, food packaging fragment, net/rope fragments, plastic fragments, plastic bottles, plastic bags, buoys, jugs and other. Proportions of debris within these nine categories were summarized and presented in the results.

To help identify what items were commonly found throughout the survey period, all intact items that were recorded a minimum of 15 times were further subclassified into 14 categories regardless of debris type: aluminum cans, balloons, beach toys, bottle caps, buckets, buoys, cups, fishing gear, food containers, food wrappers, jugs, nets/ropes, plastic bags, and plastic bottles. Proportions of debris within these 14 categories were summarized and presented in the results.

Photos of debris items were visually inspected for writing or characters that may indicate country of origin (See examples in Appendix Fig. 3). If non-English writing was present, country of origin was

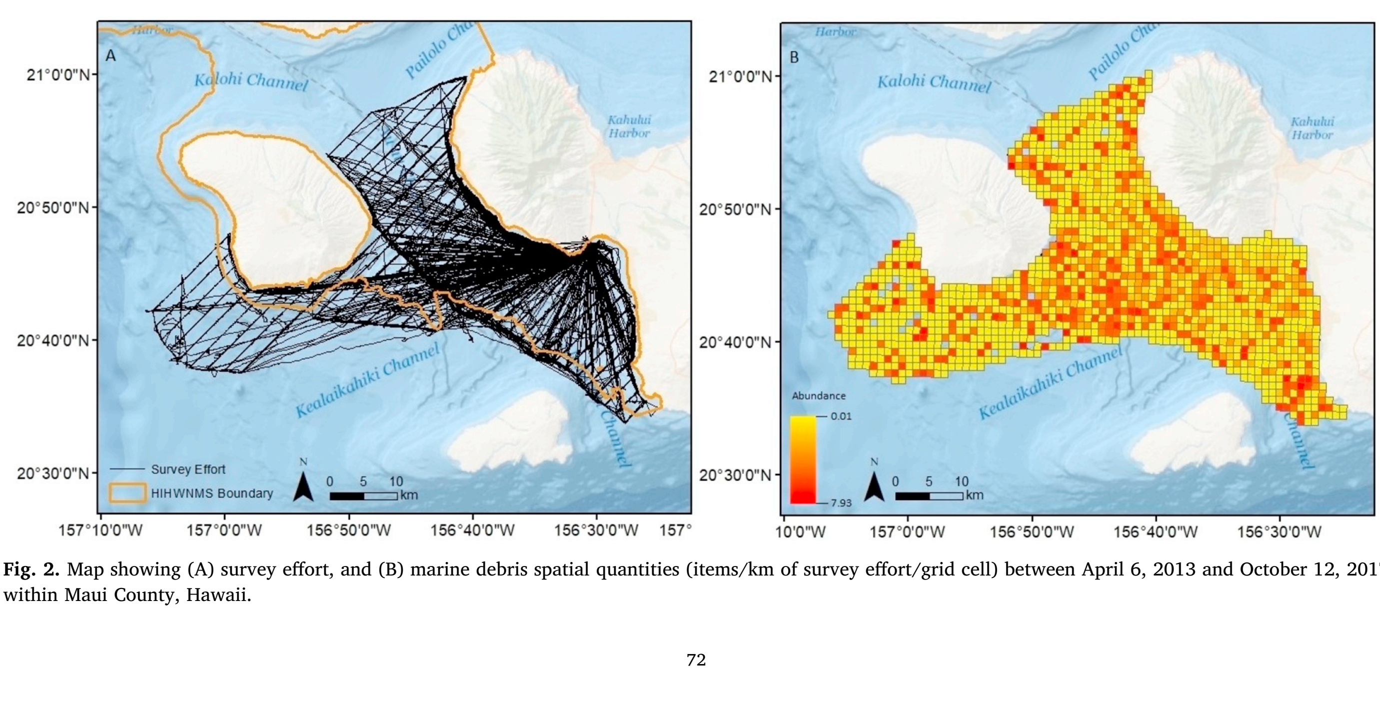

Fig. 2. Map showing (A) survey effort, and (B) marine debris spatial quantities (items/km of survey effort/grid cell) between April 6, 2013 and October 12, 2017 within Maui County, Hawaii.

assumed based upon the language displayed on the debris item.

To assess sources and composition, results were summarized for each survey year by dividing yearly sum of debris counts within each of the four debris source categories (ocean, land, general, unknown) by the sum of the yearly survey effort (km). This resulted in final units of debris count/km/year. A two way ANOVA was used to test for differences in yearly quantities across the source categories and year. To account for unequal sample sizes within the independent variables, a Type III sums of squares was employed in our ANOVA using the Anova () and aov() functions in R (R Core Team, 2017; Fox and Weisberg, 2011). Before conducting the statistical analyses, the lack of normality was addressed by log transforming the data prior to analyses (Zar, 1984).

Debris located within 50 m of each other were classified as a cluster of debris and considered an indicator of localized high accumulation. The mean number of debris items per cluster was calculated for each year to determine if cluster concentration varied with time.

All hard plastics collected from April 19, 2017 to July 11, 2017 were analyzed with a PerkinElmer attenuated total reflectance Fourier transform infrared spectrometer Spectrum Two (Waltham, MA) (ATR FT-IR) for polymer identification using the method described in Jung et al. (2018). Air-dried pieces were weighed to ± 1 g or for smaller pieces to ± 0.00001 g. Pieces were not cleaned prior to analysis, but were cut with a razor blade when needed to expose a clean, smooth, and uncontaminated inner surface. Items that contained more than one part (e.g., a bottle and a cap) were separated into multiple pieces for analysis. All samples were assigned a color, opacity, and weathering code. Weathering codes were assigned visually as 1 = mild, 2 = moderate, and 3 = severe based on the intensity of square fracturing on the surface of the sample with 1's having the least and 3's having the most (Appendix Fig. 4). Polymers were identified from spectra using absorption bands, criteria, and the decision tree described in Jung et al. (2018). A float test in ethanol and deionized water solutions with densities of 0.931 and 0.941 g/mL was performed as outlined in Jung et al. (2018) on 21 unknown PE samples to differentiate between low- and high-density polyethylene (LDPE, HDPE).

Daily debris counts (items/day) were analyzed separately for land-based, ocean-based, general-source, and unknown-source as processes leading to changes in accumulation are likely to be different for each category. However, there is value in understanding if all debris collected, regardless of source, is influenced more by land or ocean drivers. To determine this, two models of daily debris counts (items/day) were tested, one model using only land-based variables/processes and a second model using only ocean based variables/processes, as described below. The set of drivers (land or ocean) resulting in the lowest Akaike's Information Criterion (AIC) model was then considered to be the most influential set of variables for describing overall debris trends within the study and presented in the results.

To account for potential nonlinear relationships between debris counts and explanatory variables (Ribic et al., 2012), Generalized Additive Models (GAM) were constructed using the 'mgcv' package in R (Wood, 2017), using a gamma of 1.4 to avoid overfitting (Ribic et al., 2012; Wood, 2006). Daily debris counts (items/day) were modelled for each source category and log-transformed for normality as a function of survey variables, environmental variables, and process-based variables (partially adapted from Ribic et al., 2012 and explained below), with an offset term for daily survey effort (km/day). Explanatory variables were tested for pairwise correlations using the stats package in R (R Core Team, 2017). To account for non-normality in variables, the Spearman correlation coefficient (*rs*) was used to assess correlations. If variables were highly correlated ($rs \geq 0.7$) (Gonzalez-Suarez et al., 2013) only the variable that provided the lowest AIC value was retained.

To model temporal trends, a coded survey day (Ribic et al., 2012) was used, where 1 represented the first survey day (April 6, 2013) and 1634 represented the last survey day (October 12, 2017). Variations in within-year deposition may be the result of human activity and/or extreme weather events, which as described in Ribic et al. (2012), can consistently be captured with month. Therefore, within-year trends were modelled using month and between year variations using year as an explanatory variable.

The following variables were associated with each survey date using historical National Data Buoy Center (NDBC) data (www.ndbc.noaa.gov): wind speed (km/h) and direction (degrees), peak gusts (m/s), wave height (m), dominant and average wave period (s), dominant wave direction (direction), sea level pressure (hPa), air temperature (Celsius), and sea surface temperature (Celsius). A total of 24 active buoys are deployed within 500 km of the Hawaii islands chain, with the majority concentrated around the island of Oahu (National Climatic Data Center's (https://www.ncdc.noaa.gov), 2018). Two of these active buoys were selected based on their proximity and location relative to the study area and the metrics they recorded. The two data sources were evaluated independently for analysis of ocean-based debris, as variables differed between the two sources. Only the source data set providing the lower AIC model was presented in the final results. Data were compiled from April 2013 to October 2017 from the following two data buoys: Station 51,205 (NDMC, 2018a) located 41 km NW of center of the study region (center: 20.73623°N, 156.69085°W), and Station 51,003 (NDBC, 2018b) located 436 km WSW of the center of the study region. Final selection of stations 51,205 and 51,003 for analysis was based on the availability of continuous data for the entire duration of the study period as well as physical location. Before analysis was conducted, data were quality controlled by removing missing data, denoted with variable number of 9's. To assess the impacts of these variables on daily debris count, weather variables were modelled at the time of survey and the day prior.

Debris retention and accumulation on beaches in Maui is known to be impacted by ocean factors such as wave and tide height (Blickley et al., 2016). As such, both land and ocean variables were considered when evaluating potential drivers of land-based debris. For land based variables, each survey date was associated with the following data taken from station KLIH1 – 1615680 (National Climatic Data Center's (https://www.ncdc.noaa.gov), 2018) located 26 km NE of the center of the study region (center: 20.73623°N, 156.69085°W): average daily wind speed (km/h) and direction (degrees), fastest wind speed (km/h) and direction (deg), and precipitation (Y/N). Ocean-based variables were taken from Station 51,205 (NDMC, 2018a).

Each month of the survey period was classified by the presence of the following: (1) El Nino-Southern Oscillations (ENSO) event, (2) La Nina-Southern Oscillations (LNSO) event, or (3) no event. Data used in analysis were taken from the NOAA Climate Prediction Center (2018) ENSO monthly categorization table.

Monthly sea surface temperatures from April 2013 to October 2017 were downloaded from NOAA Earth System Research Laboratory Physical Sciences Division (2018). The 1.0° latitude by 1.0° longitude global grid was loaded into ArcMap (Environmental Systems Resource Institute, 2018) and the contour tool in the spatial analyst extension was used to create an 18 °C isotherm. The proximity (distance in km) of this isotherm to the center of the study region (center: 20.73623°N, 156.69085°W) was then calculated in ArcMap. Each month of the survey period was then classified by the distance to the 18 °C isotherms. The 18 °C isotherm can be used as an index for the proximity of the Subtropical Convergence Zone (STCZ) (Pichel et al., 2007), with the expectation that debris loads are higher when STCZ is closer to Hawaii (Ribic et al., 2012).

To determine if tourism influenced land-source debris items, the total monthly visitor days from 2013 to 2017 were obtained from the Hawaii Tourism Authority (HTA, 2018). Monthly visitor days, calculated by multiplying total monthly visitor count (tourists/month) by the average monthly visitor duration (days), ranged from 1.30 million to 2.10 million. To facilitate analysis, monthly visitor days were classified into four categories ranging from 1 (lowest number of visitor days) to 4 (highest number of visitor days) as follows: 1 (1.30–1.49 million); 2

(1.50–1.69 million); 3 (1.70–1.89 million); 4 (1.90–2.10 million). All 54 months of the survey period were assigned a value of 1 to 4, corresponding to the appropriate ranges as determined from HTA data set. To facilitate interpretation of results, the average monthly visitor days and air temperatures (°C) from January 2013 to December 2017 for Maui were summarized and are presented in Appendix Fig. 5.

Given the potential variability in general- and unknown-source debris, a combination of ocean and land-based variables described in previous sections were used in selecting the best general- and unknown-source models, similar to that of Ribic et al. (2012).

Analysis began with testing of a full candidate model, including all possible explanatory variables and using AIC to rank the models (Burnham and Anderson, 2002) and determine which variables were candidates for removal from the model. Variables were then removed in a stepwise manner until a minimum AIC value was reached. Significance was assessed at $\alpha = 0.05$ and the minimum AIC models were presented in the results.

From April 6, 2013 to October 12, 2017, 38,270 km was surveyed (Fig. 2A), and 2095 pieces of marine debris were documented. Marine debris was observed in all parts of the survey area (Fig. 2B). Debris spatial quantities (total debris items/km of survey effort/grid cell) over the total survey period showed a trend of higher accumulation between the islands of Maui, Lanai, and Kahoolawe in the area where the Auau, Kealaikahiki, and Lalakeiki channels meet (Fig. 2B).

Of the 2095 debris items documented, the majority of the debris was classified as general-sourced debris (Fig. 3A). Plastics were the predominant type of debris recorded within the study area, accounting for 90% of total debris (Fig. 3B).

Quantities of land, ocean, general-source, and unknown-source debris varied between years, with 2017 having the highest quantity of debris observed over the five year study period (Table 1). Quantities were found to vary between year (Sum Sq: 0.73, F-value: 33.40, p-value: < 0.001) and source category (Sum Sq: 1.58, F-value: 34.42, p-value: < 0.001). Of the debris that could be identified as land or ocean based, the majority was ocean based; which in some years was four times the concentration of land based debris (Table 1).

The proportion of ocean-based debris was highest in 2013. There was a general decreasing trend in general-source debris and a general increasing trend of unknown-source debris throughout the survey period. Land-based debris increased with time having highest proportion in 2016 and 2017 (Table 1).

An overall increase in weekly debris counts (items/week) was observed throughout the study period, with a steep increase from March–May in 2017 (Fig. 4). A 354.5% increase in all debris quantities (items/km effort) was observed in 2017 when compared to average of the previous four years, the majority of which was attributed to general- and unknown-source debris (Table 1). With the exception of 2015, the maximum yearly cluster concentration (number of debris items accumulated within 50 m of each other) increased with year (Table 1). The mean cluster size was highest during 2016 and 2017, with maximum debris cluster in 2017 nearly double the average of the previous four years (Table 1).

The majority of plastic debris items consisted of plastic (36%) and foam (14%) fragments (Fig. 5A). For items that were found whole, plastic bottles (16%) and buoys (14%) accounted for nearly one-third of these collected (Fig. 5B).

Of the debris documented, 73.3% (n = 1536) exhibited some form of biofouling, with plastics comprising the largest proportion (n = 1425, 92.7%) of biofouled items. The amount of biofouling varied by item, but was highest for buckets (avg = 55.8%) and lowest for balloons (avg = 1.4%) (Fig. 6). Eight items contained biofouling not native to Hawaiian waters, including blue mussels (*Mytilus edulis*), chitons (*Mopalia*), and/or limpets (*Lottia*), which were initially identified in the field before being photographed and sent to the Department of Aquatic Resources for confirmation when possible. Foreign writing allowed for assessment of probable country of origin for 23 items. Of these items, 10 items displayed Japanese characters, 7 displayed Chinese characters, and 4 displayed Korean characters. Two items displayed characters which could have belonged to either the Japanese or Chinese languages, and therefore could not be identified to a country. The majority of these items (13 items, 56.5%) were identified as coming from ocean sources, while the remaining 10 items could have originated from either land- or ocean-based sources.

The subset of 252 hard plastic debris items collected from April 14 to July 11, 2017 was analyzed to determine polymer composition. The majority (52.5%) of debris items analyzed was classified as severely weathered, with the remaining items being mildly (27%) and moderately (25.5%) weathered. Weathering code did not impact polymer composition, because pieces were cut to reveal an inner surface for ATR FT-IR measurements. All pieces consisted of polymers that would float in seawater, based on the polymer density, except for one PET bottle. HDPE, LDPE, and polypropylene (PP) accounted for the largest proportion of debris sampled (Fig. 4a & b). PP makes up the greatest proportion of sampled debris by mass, while HDPE and LDPE make up the greatest proportion by count (Fig. 7A & B).

The lowest AIC model for the entire dataset, regardless of debris source category, included ocean-based sets of variables/processes. Data from Buoy 51,003 with a one day offset resulted in the best fit model for this dataset. The most significant driver of debris counts (items/day) was coded survey day (Table 2; Fig. 8B). Additional factors, in decreasing order of significance included: the interaction between wave height (m) and dominant direction (degrees) (Table 2; Fig. 9), non-

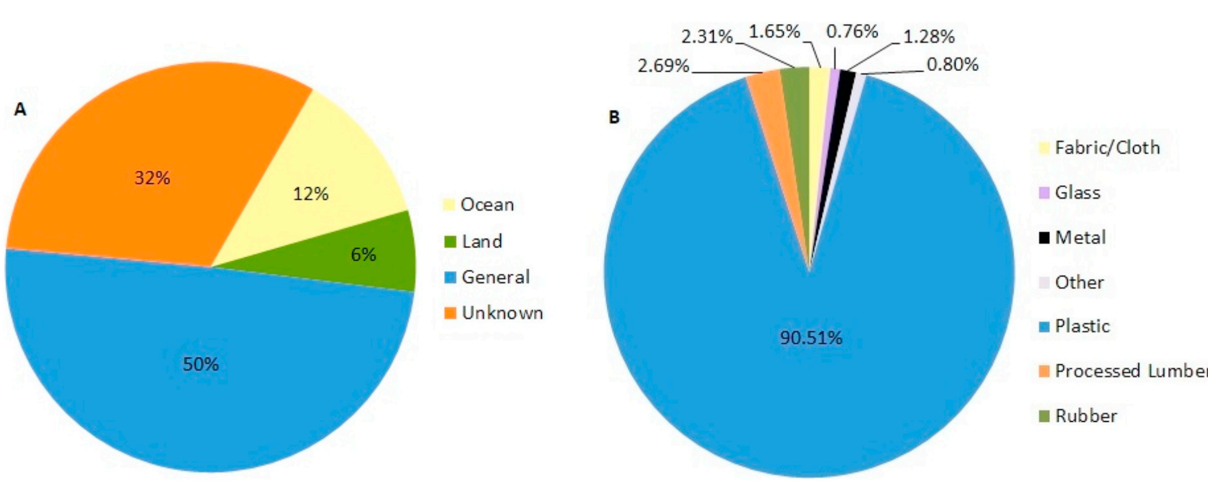

Fig. 3. Proportions of debris (A) origin and (B) material collected between April 6, 2013 and October 12, 2017 within the coastal waters of Maui County, Hawaii.

Table 1
Yearly quantities, proportions, and cluster sizes of marine debris items summarized by total and source categories documented between April 6, 2013 and October 12, 2017 within the coastal waters of Maui County, Hawaii.

Year	2013	2014	2015	2016	2017
Debris summary					
Total debris count	402	276	236	328	853
Total survey effort (km)	5985	7963	5067	4218	4248
Total survey days	47	62	43	36	31
Quantities of debris					
Quantities of all debris (items/km effort)	0.067	0.036	0.048	0.080	0.201
Quantities of ocean-based debris (items/km effort)	0.013	0.006	0.008	0.011	0.009
Quantities of land-based debris (items/km effort)	0.003	0.003	0.004	0.009	0.009
Quantities of general-source debris (items/km effort)	0.036	0.018	0.027	0.033	0.097
Quantities of unknown debris (items/km effort)	0.015	0.008	0.008	0.027	0.085
Proportions of debris					
Proportion of ocean-based debris	19.40	17.54	17.43	14.29	4.69
Proportion of land-based debris	4.44	8.07	9.13	10.71	4.34
Proportion of general-source debris	53.73	50.88	56.02	41.37	48.53
Proportion of unknown debris	22.39	23.51	17.43	33.63	42.44
Debris clusters					
Mean items/cluster[a] (standard deviation)	1.52 (1.49)	1.43 (1.49)	1.20 (0.88)	1.80 (2.10)	2.21 (3.06)
Maximum items/cluster[a]	10	13	8	14	21

[a] Debris items located within 50 m of each other were considered part of the same cluster.

ENSO/LNSO months (Table 2), and distance to the 18 °C isotherm (Table 2; Fig. 8A). The combination of low wave height (< 1 m) and wind coming from 150° or 350° resulted in high debris counts (Fig. 9). Debris counts fluctuated with distance to the 18 °C isotherm, but there was a general decreasing trend of debris with increasing distance to the 18 °C isotherm (Fig. 8A). Strong temporal trends in debris counts were evident with peaks observed in April 2015 and February 2017, each of which were preceded by dips in debris counts. A significant increase in debris counts was observed during non-ENSO/LNSO months (Table 2).

The lowest AIC model for ocean-based debris was fit using variables from Buoy 51,003, with a 1 day offset. Ocean-based debris counts (items/day) showed significant nonlinear relationships with wind direction, air temperature, and survey date; and significant linear relationships with sea level pressure and wave period (Table 3; Fig. 10). Air temperature could indicate an influence of the STCZ, as temperatures would be coldest in late winter and early spring when the zone is closest to Hawaii. As such, air temperature may represent a lag effect of the STCZ with high accumulation being observed after the zone reaches its closest point to Maui. However, further research is needed to determine the exact lag-time and potential connection between STCZ and air temperature.

Ocean-based debris counts remained fairly constant until June 2016, which saw the sharpest increase in debris until December 2016 followed by the sharpest decrease in debris until October 2017 (Fig. 10B). Peaks in ocean-based debris counts varied based on wind directions, with peaks occurring when wind direction was 0, 75, 125, and 200° (Fig. 10C).

Land-based debris counts showed significant nonlinear relationships with water temperature and average wind speed (Table 4; Fig. 11). Land-based debris counts were highest during low wind speed intervals (6–10 km/h) and high speed intervals (22–24 km/h), with variations from 6 to 12 mph (Fig. 11A). There were minimal changes to land-based

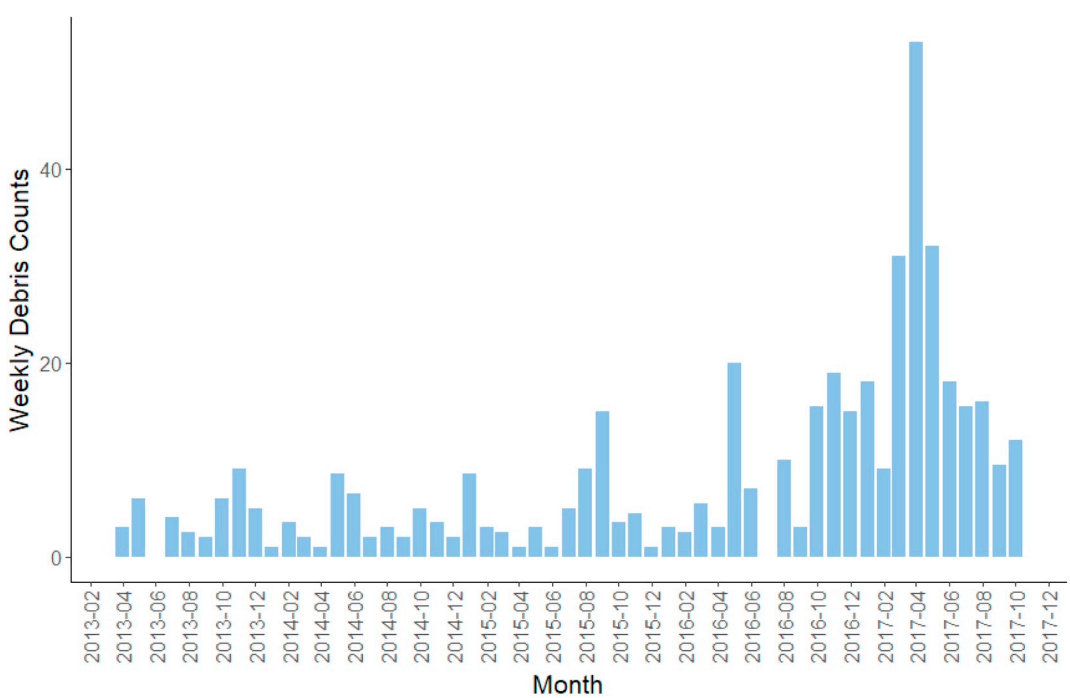

Fig. 4. Weekly counts (items/week) of debris items collected between April 6, 2013 and October 12, 2017 within the coastal waters of Maui County, Hawaii.

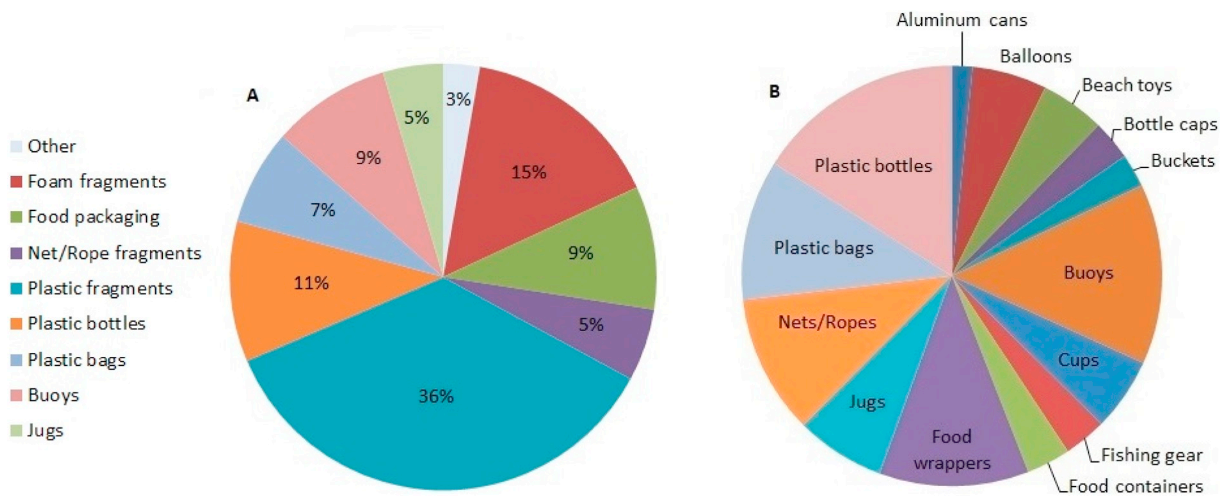

Fig. 5. Proportions of (A) plastic debris items (n = 1986) divided into subcategories and (B) intact debris items (n = 887) divided into commonly sighted subcategories groups collected between April 6, 2013 and October 12, 2017 within the coastal waters of Maui County, Hawaii.
Note: Panels A and B were created separately and the same item may be used in both figures. As such, these figures should be evaluated independently.

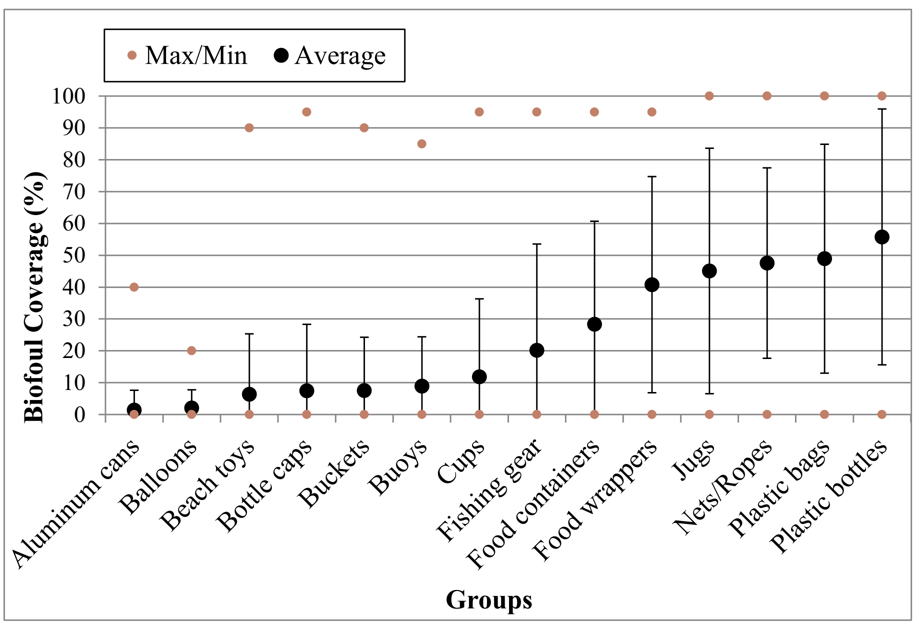

Fig. 6. Average percent biofouling observed on whole debris items divided into identifiable groups collected between April 6, 2013 and October 12, 2017 within the coastal waters of Maui County, Hawaii.
Note: Error bars represent the standard deviations.

debris accumulation on survey days when air temperatures ranged from 24.0 to 25.0 °C, with a gradual reduction in debris seen when temperatures exceeded 25.0 °C (Fig. 11B). The months with the highest temperatures correspond to months with lowest monthly visitor days: August, September, and October (Appendix Table 2).

The lowest AIC model for general-source debris was fit using variables from Buoy 51,003, with a 1 day offset; the same as the ocean-based model. General-source debris counts showed significant nonlinear relationships with survey date, and marginally significant linear relationships with year (Table 5; Fig. 12). General-source debris counts gradually declined throughout the survey period (Fig. 12). Although not significant, within year counts showed an increasing positive linear relationship with increasing years (Table 5).

The lowest AIC model for unknown-source debris was fit using variables from Buoy 51,003, with a 1 day offset; the same as the ocean-based and general-source debris models. Unknown-source debris counts showed significant nonlinear relationships with water temperature, and significant linear relationships with survey date, ENSO, wind speed, and peak gusts (Table 6; Fig. 13). Unknown-source debris counts showed a varying trend with temperature (Fig. 13). Significant positive trends of unknown-source debris counts with survey date, southern oscillations, and wind speed were observed, while a negative linear trend was found with increasing peak gusts (Table 6).

The observed variation in significant variables based on marine debris source category aligns with work presented in Ribic et al. (2012). Overall, the difference in the sets of variables included in the lowest AIC model for each source category and for all debris, regardless of source, suggests a wide variety of drivers are likely responsible for the variability in debris quantities observed in the nearshore waters of Maui County. The prevalence of survey date and ocean based variables with a one day offset in four of the five lowest AIC models suggests these processes are linked to temporal and ocean based drivers and are best captured using time and an offshore buoy data set (Buoy 51,003).

As reported in Currie et al. (2017), plastics comprised the majority

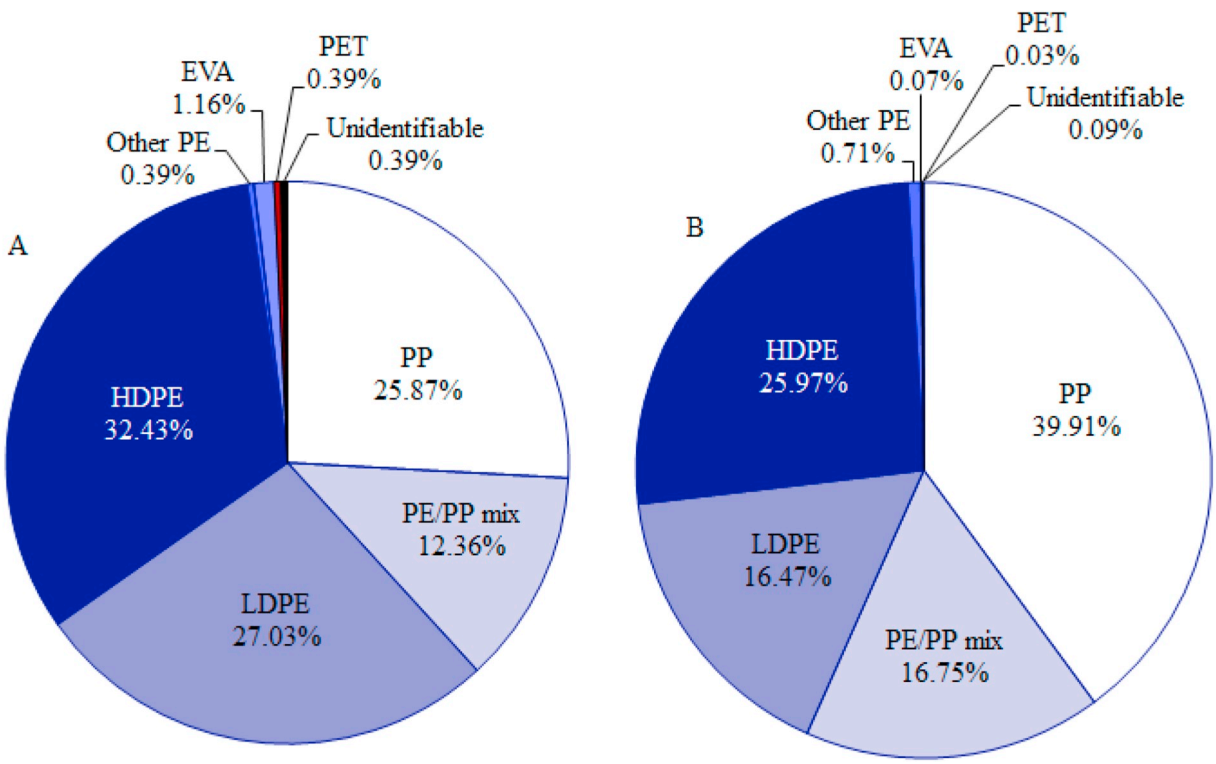

Fig. 7. Percent of polymers identified in debris sub-sample (n = 252) as a function of total (A) count and (B) mass. Low-density polymers expected to float on seawater are shown in white and blues.
Note: Polyethylene terephthalate (PET) (n = 1), high-density polyethylene (HDPE), low-density polyethylene (LDPE), other PE is a piece that had an obvious PE spectrum but with additional non-PE peaks (n = 1), polyethylene and polypropylene mixture (PE/PP mix) as defined in Jung et al. (2018), polypropylene (PP), ethylene vinyl acetate (EVA), and unidentifiable (n = 1) were pieces that produced very noisy spectrum that could not be identified.

of floating macro debris found in this study region; a result that aligns with the known prevalence of floating plastics in the ocean (Coe and Rogers, 1997; Derraik, 2002). The increasing trend in debris quantities throughout the duration of the study aligns with the global trend of increasing debris deposition in our oceans (Eriksen et al., 2014). The large increase in debris quantities observed in 2017 is likely related to the higher number of small scale debris clusters, which were observed more frequently and in larger sizes in 2017. Although observers changed throughout the survey period, the number of observers remained constant. As such, it is unlikely that differences in observers accounted for the substantial increase in debris observed in 2017. Similarly, weather conditions were also kept consistent (BSS and DSS ≤ 3) throughout the study period and weather changes likely do not account for the observed increase.

The predominantly northwest surface currents in leeward areas of Maui County occur, in part, from Ekman transport along with wind-driven eddy effects resulting from the northeast trade winds interacting with the land masses of the islands (Chavanne et al., 2002). These eddies likely result in the convergence pattern of debris seen in the channels that separate the four islands of the region; opposing eddies in the lee of Maui Island may cause areas of lower current velocities, which allows marine debris to accumulate.

Debris from Japan, China, and Korea most likely drifted to the Hawaiian Islands after transport in the Subtropical Gyre and subsequent northwest surface currents toward the leeward waters of Maui (e.g., Chavanne et al., 2002; Howell et al., 2012). Although plausible country of origin was determined via markings on the debris, it is virtually impossible to determine the exact point at which any particular item entered the marine environment. For example, debris with Japanese writing may have entered the ocean in coastal Japan, from an offshore fishing vessel, or from a tourist visiting Hawaii.

The composition of debris presented here supports previous reports that the majority of marine debris in and around the Hawaiian Islands originates from far offshore rather than local land-based sources

Table 2
Results of top generalized additive model used for determining the linear and nonlinear relationships between all debris counts and variables, based on data collected within the coastal waters of Maui County, Hawaii between 2013 and 2017.

Factor		edf[a]	F-value	p-Value	R^2	Dev. expl.
All debris nonlinear	s(18 °C isotherm)	8.22	1.95	0.05	0.45	52.1%
	s(wave height, dominant wave direction)	9.69	2.61	0.003		
	s(survey date)	7.60	7.57	< 0.0001		

Factor		Estimate	t-Value	p-Value	R^2	Dev. expl.
All debris linear	La Nina Oscillation	0.52	0.81	0.42	0.45	52.1%
	No Oscillation	1.35	2.21	0.02		

[a] edf is the estimated degrees of freedom accounting for the smoothing function.

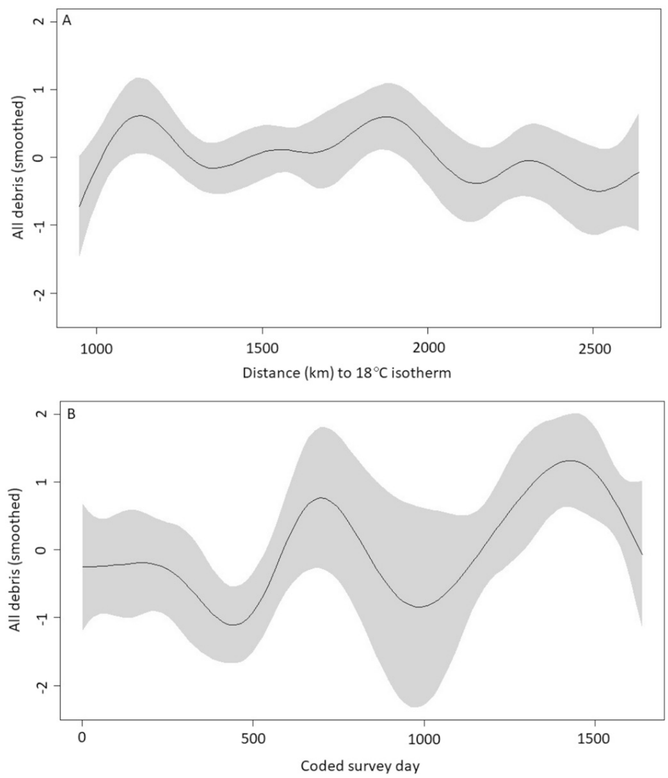

Fig. 8. Results of generalized additive model showing the significant non-linear relationships of all debris and (A) distance to 18 °C isotherm and (B) survey date for debris collected within coastal waters of Maui County, Hawaii between 2013 and 2017.
Note: Survey date is coded so that 1 = April 6, 2013 and 1634 = October 12, 2017.

(Donohue et al., 2001; Ribic et al., 2012). The proportion of general-source debris recorded in this study is slightly higher than the 30–40% recorded in other shoreline surveys in Hawaii (Ribic et al., 2012). This is likely attributed to the addition of items to the general-source category for this study, not included in the original designation by Ribic et al. (2012). Furthermore, the high proportion of unknown-source debris and the observed biofouling suggests that few items were littered into the environment recently, and thus could have come from very distant locations.

The subset of samples analyzed in 2017 suggests that only low-density floating debris were observed in the Maui County region. The high proportion of severely weathered debris suggests that few items were littered into the environment recently and most come from distant sources. These results are congruent with previous studies that found PE and PP to dominate sea surface plastic marine debris in the North Pacific (Brandon et al., 2016), North Atlantic (ter Halle et al., 2016), Indian Ocean (Syakti et al., 2017), Mediterranean Sea (Pedrotti et al., 2016), Ross Sea (Cincinelli et al., 2017), coastal Southern Malaysia (Ng and Obbard, 2006), and on beaches of Kauai, Hawaii (Cooper and Corcoran 2010). Any debris made of PET, PVC, PS, nylon, and other denser polymers entering Hawaiian waters from local sources would sink and the collection of only visible floating debris for this study explains the near absence of this type of polymer in the analysis. As such, a large amount of the marine debris is likely going undetected, as it is sinking through the water column or on the sea floor. The one piece of PET plastic collected and analyzed during the survey would have naturally sunk, but the item was a bottle that still had air inside, which kept it afloat until sample collection. Had it filled with water, it would have certainly sank and been deposited on the sea floor.

As has been shown in previous work by Ribic et al. (2010, 2011, 2012), the complex relationships of debris accumulation and varying drivers leads to temporal patterns of debris accumulation. Blickley et al. (2016) found debris accumulation on Maui's beaches fluctuated on a monthly and daily basis and resuspension of debris was related to wind, tides and wave height. Ocean based phenomena such as ENSO events, as well as proximity to the STCZ, as indicated by the 18 °C isotherm (Pichel et al., 2007), did have a significant impact on overall debris quantities and unknown-source debris observed in this study. Although not assessed in this study, there could be a delayed pulse of debris accumulation occurring after strong El Nino years, explaining the significance positive debris counts during non ENSO event months observed in this study. For example, the 2016 moderate to strong El Nino could have caused the high accumulation event observed in 2017, and not be predicted by ENSO months or proximity to the STCZ used in our models due to an untested lag effect. This movement of the STCZ further south and closer to Hawaii could have resulted in deposition of high amounts of debris into Hawaii's dynamic coastal waters. This may have caused the increase in debris accumulation observed in 2017, as the debris was subject to movement through this dynamic system. The role of ENSO events and the STCZ on coastal debris accumulation warrants additional research to help understand the potential connection to high accumulation events.

Significant variables for all debris sources were a mix of local and large scale phenomena, suggesting a complex process of drivers, mostly relating to ocean based variables, are responsible for the observed variations in debris quantities within Maui County. There were five variables identified as significant drivers of ocean-based debris, suggesting multiple factors lead to the accumulation of ocean-based debris within Maui County. The low R^2 value suggests other untested factors are contributing to ocean-based debris fluctuations. The increase in ocean-based debris count with temperature could be contributed to the proximity of the STCZ to Maui, which is closer during colder periods and is known to concentrate debris (Pichel et al., 2007). This is further supported by the significance of the 18 °C isotherm in increasing and decreasing all debris counts regardless of source, suggesting that observed debris likely originate outside the Hawaiian Islands. Temperature could also serve as a proxy for various oceanic processes (Ryan et al., 2009) that were not specifically tested here. The large fluctuations in ocean-based debris over time observed between 2015 and 2016 aligned with one of the strongest El Niño events observed since 1950 (ENSO, 2016). The absence of the ENSO variable from the lowest AIC model for the ocean-based debris likely resulted from debris that arrived via the ocean being classified as unknown-source due to inability to identify the source. Additionally, interaction effects were not tested and only considering the ENSO variable on its own could further explain the absence from this model. This is further supported by the inclusion of the ENSO variable in the model looking at all debris regardless of source category.

The general increase in ocean-based debris from 100 to 250° corresponds to wind coming from a direction unobstructed by any of the four islands, with the peak of 200–250° (from Southwest) representing the least obstructed area between Kahoolawe and Lanai.

The increase in sea level pressure read at the buoy southwest of the study area is indicative of a shift in the high pressure ridge over the islands, which causes a stalling of the prevailing Northeast trade winds common to the Hawaiian Islands (Garza et al., 2012). The observed increase in ocean-based debris quantities with increasing sea level pressure suggests this stalling of trade winds slows the transport of ocean-based debris out of the study area. This is further supported by the significant increase in debris observed during non-ENSO/LNSO months for all debris, as this is when a slowing of the trade winds is expected.

The general decrease in land-based debris with water temperature observed could be attributed to seasonal beach use by residents and tourists, not detected using visitor count categories. Sea surface temperatures are hottest in September and October (National Data Buoy

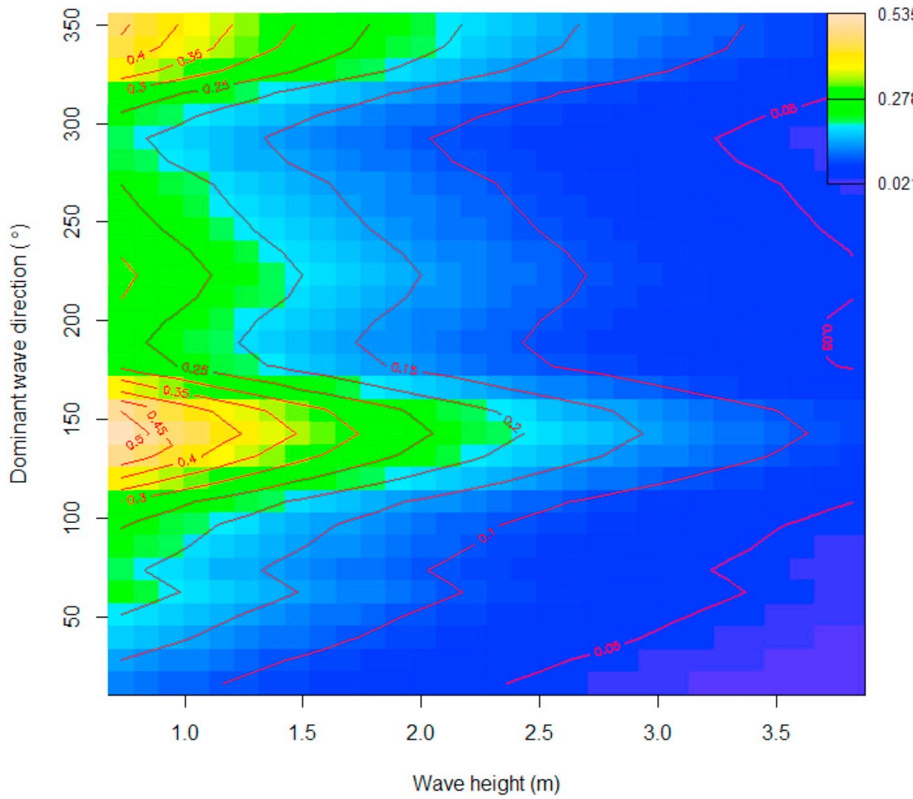

Fig. 9. Results of generalized additive model showing the significant relationship of the interaction between wave height and direction on all debris counts collected within coastal waters of Maui County, Hawaii between 2013 and 2017.

Center 2018a), which corresponds with the lowest monthly average visitor days for the island of Maui (Hawaii Tourism Authority, 2018) and could explain the reduction in debris counts from 25 to 27 °C. Temperatures in the range of 24–25 °C correspond to the peak tourism months of December to March which may explain the peak observed in land-based debris at these temperatures. The authors believe water temperature in Hawaii could be an inverse measure to indicate the number of beach users. However, further research is needed to confirm these trends and to assess if tourists contribute more or less marine debris than local residents. Population levels have been shown to impact debris deposition (Thiel et al., 2011) and variations in visitors to Maui may impact the amount of debris observed through direct deposition and/or deposition via transport systems such as streams to the ocean system (Ribic et al., 2012).

Wind is a primary driver of debris transport over land and deposition into coastal waters (Blickley et al., 2016). The initial trend of high land-based debris deposition with low wind speeds could relate to the number of beach users because calm, windless days are favored for beach activities, which are common throughout Maui. The variability observed in land-based debris deposition with wind speed is not surprising as the amount and rate of land-based debris collection in this study was likely the result of complex drivers that vary with beach site. Blickley et al. (2016) showed that land-based accumulation on beaches was mostly influenced by wind, which was also the most significant term for accumulation of land-based debris in Maui's nearshore waters.

General-source debris represented the largest proportion of data collected in this study and can be affected by land-based or ocean-based processes (Barnes et al., 2009; Ribic et al. 2012). Survey date was the only variable that was significant in the general-source model and common with the lowest AIC model for ocean-based debris. This suggests that general-source debris accumulation in Maui County has a temporal component, which may be linked to ocean processes as suggested in Ribic et al. (2012).

The decreasing non-linear trend in general-source debris counts from 2013 to 2017 is worth noting as it corresponds with increasing linear estimates of yearly debris counts. Although the yearly estimates

Table 3
Results of top generalized additive model used for determining the linear and nonlinear relationships between ocean-based debris counts and variables, based on data collected within the coastal waters of Maui County, Hawaii between 2013 and 2017.

Factor		edf[a]	F-value	p-Value	R^2	Dev. expl.
Ocean-based nonlinear	s(wind direction)	7.86	2.41	0.02	0.14	43.6%
	s(air temp)	3.28	5.70	< 0.0001		
	s(survey date)	7.27	3.21	< 0.001		

Factor		Estimate	t-Value	p-Value	R^2	Dev. expl.
Ocean-based linear	Wave height	0.09	1.37	0.17	0.14	43.6%
	Dominant wave period	−0.06	−2.13	0.03		
	Sea level pressure	0.13	3.26	< 0.001		

[a] edf is the estimated degrees of freedom accounting for the smoothing function.

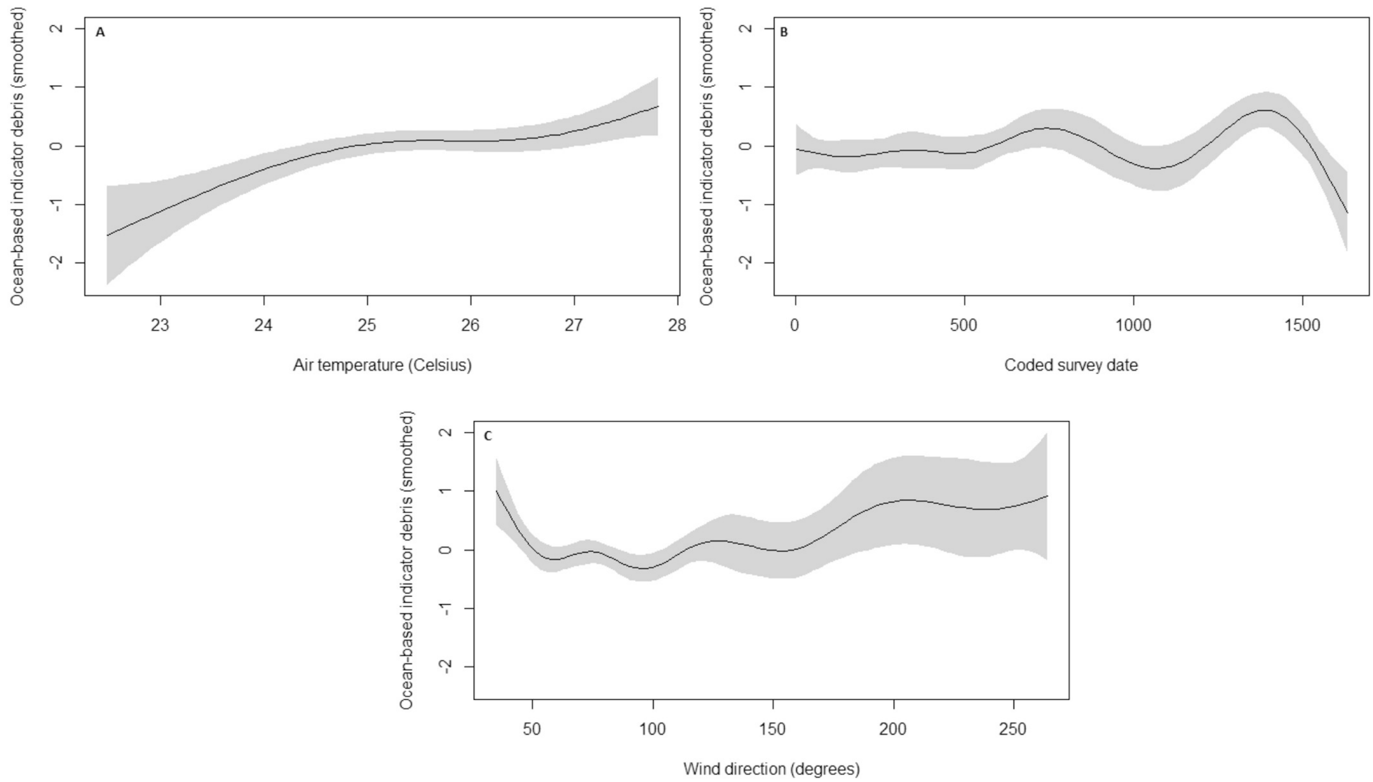

Fig. 10. Results of generalized additive model showing the significant non-linear relationships of ocean-based debris and (A) air temperature, (B) survey date, and (C) wind direction for debris documented within coastal waters of Maui County, Hawaii between 2013 and 2017.
Note: Survey date is coded so that 1 = April 6, 2013 and 1634 = October 12, 2017. Fig. 10C represents trends over recorded over the dominant wind direction, which ranged from 35 to 264° throughout the survey period.

are not significant, these results would suggest a strong temporal component that varies on a daily and yearly basis and is likely a proxy for a process not tested in our models.

There were six variables identified as significant drivers of unknown-source debris, all relating to ocean-based environmental variables suggesting the majority of debris in this category has been floating in the ocean for prolonged periods of time. This is further supported by the fragmented nature of debris in this category, likely being broken down from sun, wind, and wave action. The variation in unknown-source debris counts with water temperature is likely the result of the movement of the STCZ to Maui, which is known to concentrate debris (Pichel et al., 2007). The high significance counts of unknown-debris during non ENSO event months is similar to what was observed for all debris and likely is the result of a similar process and a delayed pulse after a strong ENSO month. The increasing trend of unknown-debris counts with increasing wind speed and the decreasing trend of unknown-debris counts with increasing peak gusts is likely attributed to the topography of the Maui Nui region. Further, it suggests unknown-debris is largely influenced by wind driven currents and waves with increasing wind speed increasing accumulation and high gusts moving debris either onshore, or beyond the study region.

The use of an unknown-source category represents an important addition to the categories presented by Ribic et al. (2012), for the Hawaii region as it represents a large portion of the observed debris. The expansion of categories proposed by Ribic et al. (2012) to include an unknown-source of debris fragments is further strengthened by the fact that plastic fragments make up an estimated 96% of the plastics found in the North Pacific (Robards et al., 1997). As such, identifying significant drivers for this category will contribute to a better understanding of debris accumulation in the North Pacific.

To mitigate the problem of marine debris an understanding of how it gets into our environment is required. To the best of the author's knowledge, this is the first published study in Hawaii to conduct systematic ocean surveys as a method of quantifying marine debris and, as

Table 4
Results of top generalized additive model used for determining the linear and nonlinear relationships between land-based debris counts and variables, based on data collected within the coastal waters of Maui County, Hawaii between 2013 and 2017.

Factor		edf[a]	F-value	p-Value	R^2	Dev. expl.
Land-based nonlinear	s(wave height)	1.82	0.85	0.47	0.32	57.1%
	s(average wave period)	2.80	2.310	0.07		
	s(water temperature)	2.19	3.71	0.02		
	s(average wind speed)	7.02	2.89	0.01		

Factor		Estimate	t-Value	p-Value	R^2	Dev. expl.
Land-based linear	Dominant wave direction	−0.01	−1.33	0.19	0.32	57.1%

[a] edf is the estimated degrees of freedom accounting for the smoothing function.

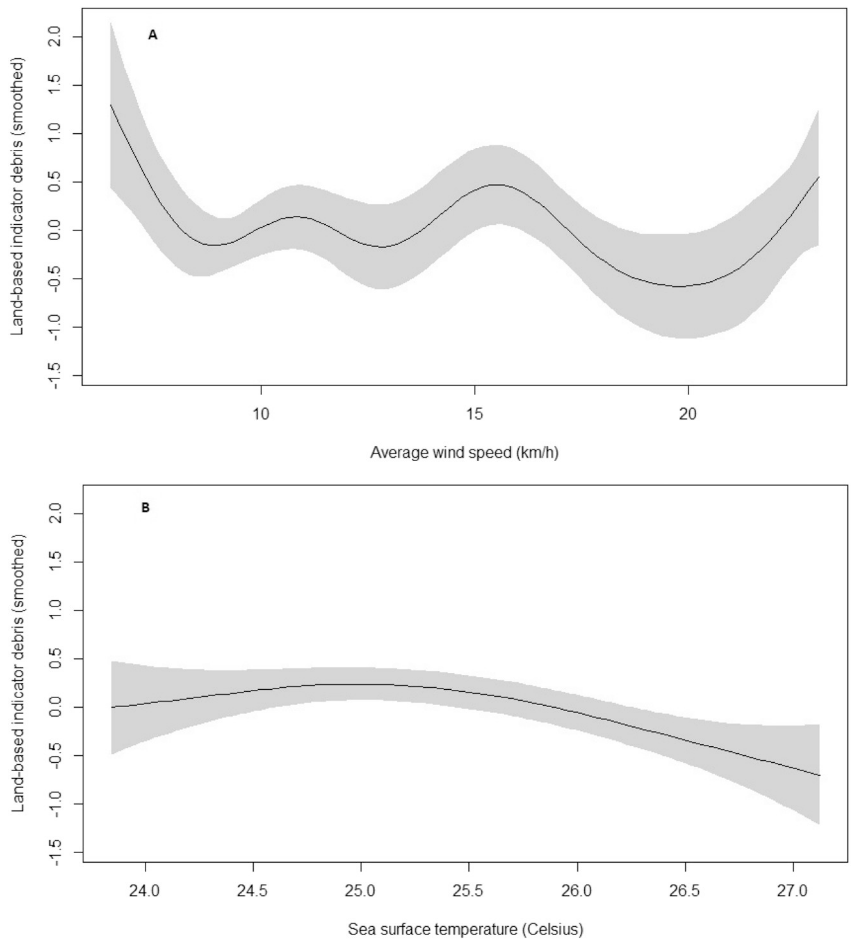

Fig. 11. Results of generalized additive model showing the significant non-linear relationships of land-based debris and (A) wind speed and (B) sea surface temperature for debris documented within coastal waters of Maui County, Hawaii between 2013 and 2017.

such, provides valuable baseline information on the sources and accumulation patterns of pollution at the sea surface in this region. Removal efforts are useful in getting debris out of the environment, but quantifying the types, sources, and amounts of debris is essential to stopping this problem at the source. Systematic research is needed at the regional level because each area will have its own unique drivers and trends. Citizen science programs, such as Pacific Whale Foundation's Coastal Marine Debris Monitoring Program, can serve as a low cost method of obtaining data so researchers can monitor debris types and loads. Baseline data are important to obtain so that trends can be detected and these data are crucial for managers and lawmakers to implement informed, scientifically-backed policies and mitigation measures.

Disclaimer: Certain commercial equipment, instruments, or materials are identified in this paper to specify adequately the experimental procedure. Such identification does not imply recommendation or endorsement by the National Institute of Standards and Technology, nor does it imply that the materials or equipment identified are necessarily the best available for the purpose.

Table 5
Results of top generalized additive model used for determining the linear and nonlinear relationships between general-source debris counts and variables, based on data collected within the coastal waters of Maui County, Hawaii between 2013 and 2017.

Factor		edf[a]	F-value	p-Value	R^2	Dev. expl.
General nonlinear	s(month)	6.83	1.66	0.12	0.30	43.1%
	s(survey date)	5.29	6.26	< 0.001		
	s(water temperature)	6.95	8.05	0.11		

Factor		Estimate	t-Value	p-Value	R^2	Dev. expl.
General linear	Year 2014	3.95	1.49	0.14	0.30	43.1%
	Year 2015	9.52	1.80	0.07		
	Year 2016	14.41	1.83	0.07		
	Year 2017	19.48	1.85	0.06		

[a] edf is the estimated degrees of freedom accounting for the smoothing function.

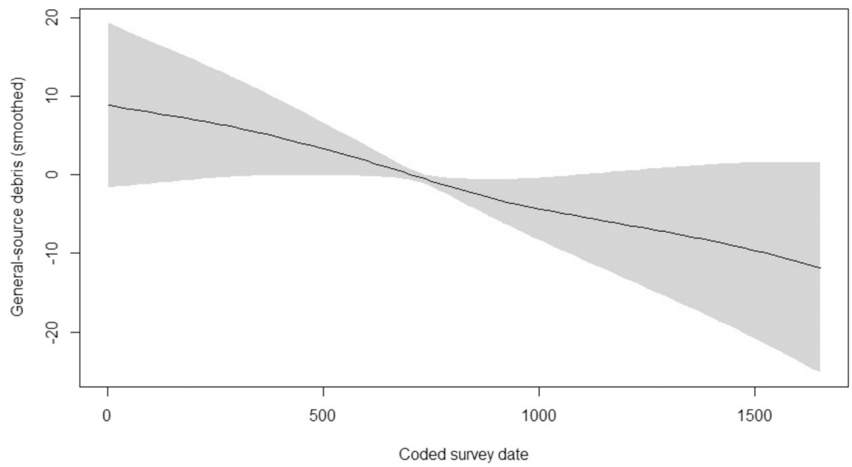

Fig. 12. Results of generalized additive model showing the significant non-linear relationships of general-source debris and survey date for debris collected within coastal waters of Maui County, Hawaii between 2013 and 2017.
Note: Survey date is coded so that 1 = April 6, 2013 and 1634 = October 12, 2017.

Table 6
Results of top generalized additive model used for determining the linear and nonlinear relationships between unknown-source debris counts and variables, based on data collected within the coastal waters of Maui County, Hawaii between 2013 and 2017.

Factor		edf[a]	F-value	p-Value	R^2	Dev. expl.
Fragments nonlinear	s(average wave period)	1.27	1.51	0.17	0.45	51.5%
	s(water temperature)	4.41	3.92	< 0.001		

Factor		Estimate	t-Value	p-Value	R^2	Dev. expl.
Fragments linear	Survey date	0.001	7.37	< 0.0001	0.45	51.5%
	La Nina Oscillation	0.682	2.33	< 0.01		
	No Oscillation	1.01	4.81	< 0.0001		
	Average wind speed	1.46	2.36	< 0.01		
	Peak gusts	−1.32	−2.44	< 0.01		

[a] edf is the estimated degrees of freedom accounting for the smoothing function.

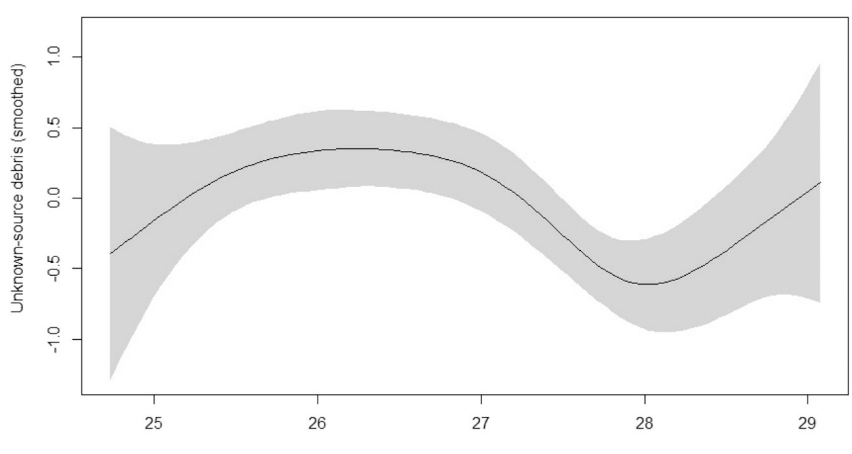

Fig. 13. Results of generalized additive model showing the significant non-linear relationships of unknown-source debris and water temperature for debris documented within coastal waters of Maui County, Hawaii between 2013 and 2017.
Note: Survey date is coded so that 1 = April 6, 2013 and 1634 = October 12, 2017.

Acknowledgements

The authors would like to thank Jessica McCordic, Grace Olson, and Abigail Machernis for help with data collection and quality control and the Pacific Whale Foundation research interns who helped with data collection and input from 2013 to 2017. We would also like to thank Jared Ragland and Mark Manual for their guidance and comments on the initial review of this manuscript. Special thanks goes to Rachael Lallo for creating several appendix figures. We thank Pacific Whale Foundation's members and supporters for providing the funding necessary to conduct this study.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.marpolbul.2018.11.026.

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Plastics in the Marine Environment

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Annu. Rev. Mar. Sci. 2017. 9:205–29

First published online as a Review in Advance on September 7, 2016

The *Annual Review of Marine Science* is online at marine.annualreviews.org

This article's doi:
10.1146/annurev-marine-010816-060409

Keywords

plastic debris, ocean pollution, contamination, impacts, risk analysis, research priorities

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).

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.

Supplemental Material

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,

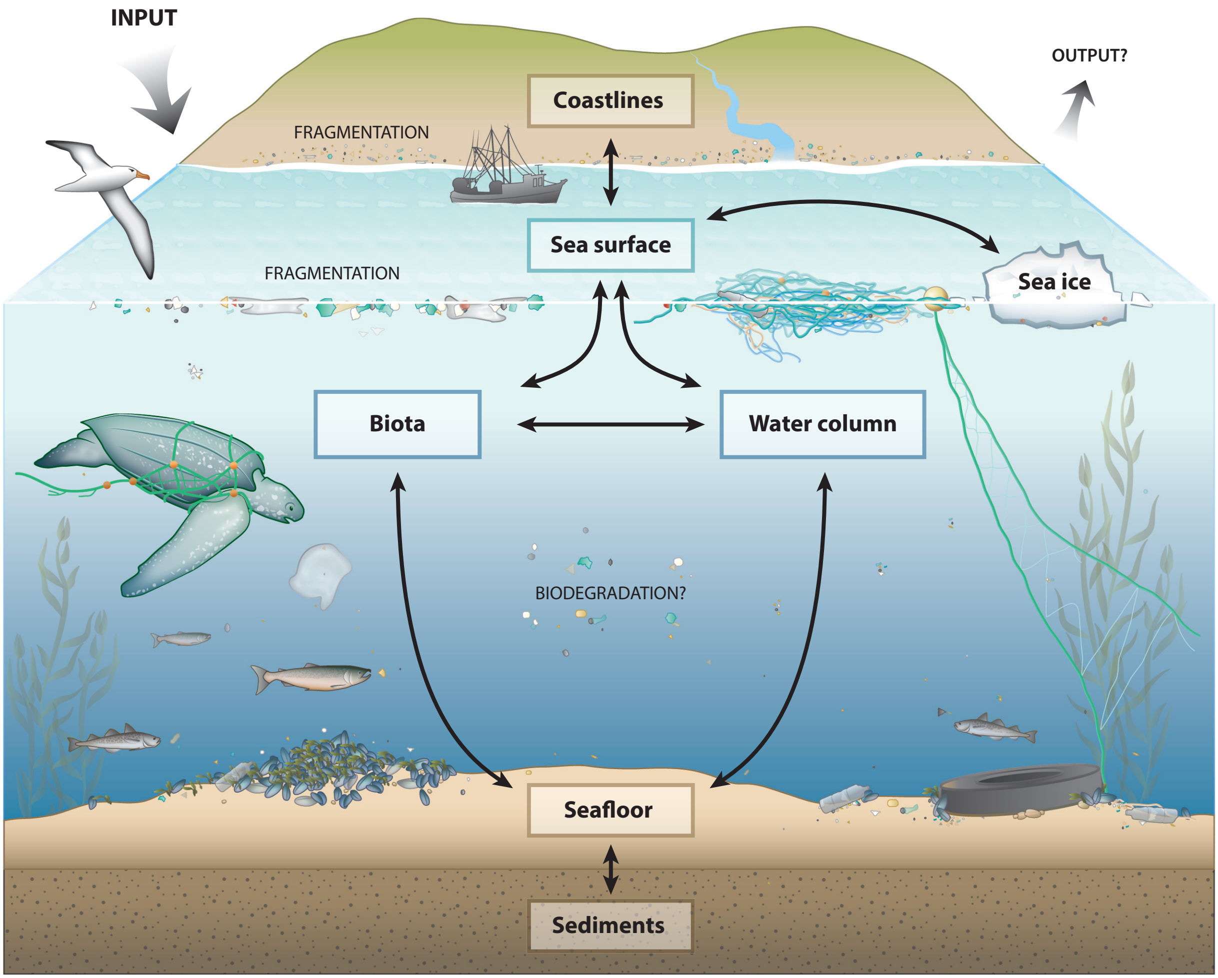

Figure 1

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

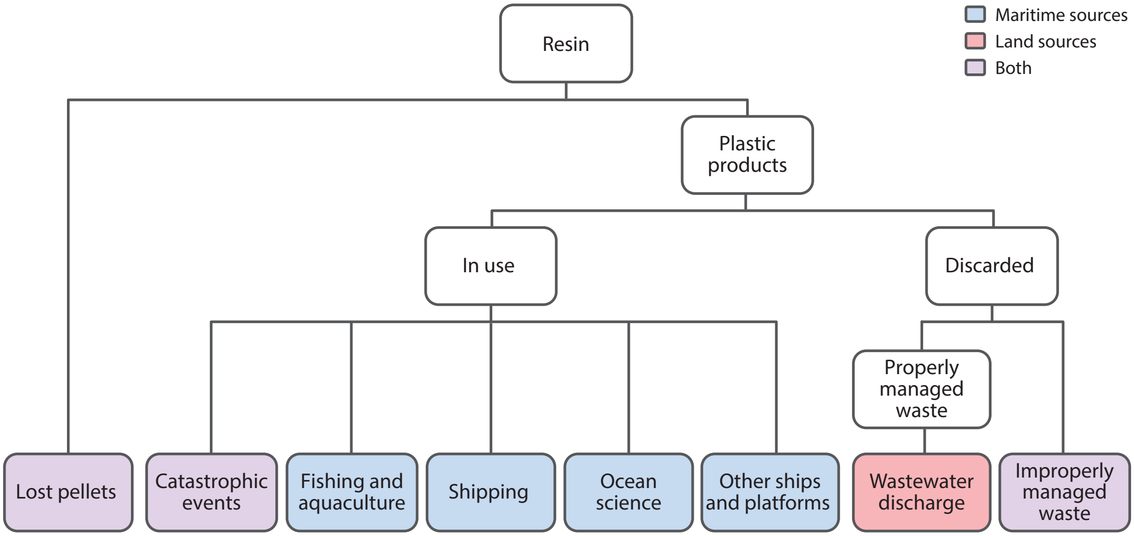

Figure 2

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

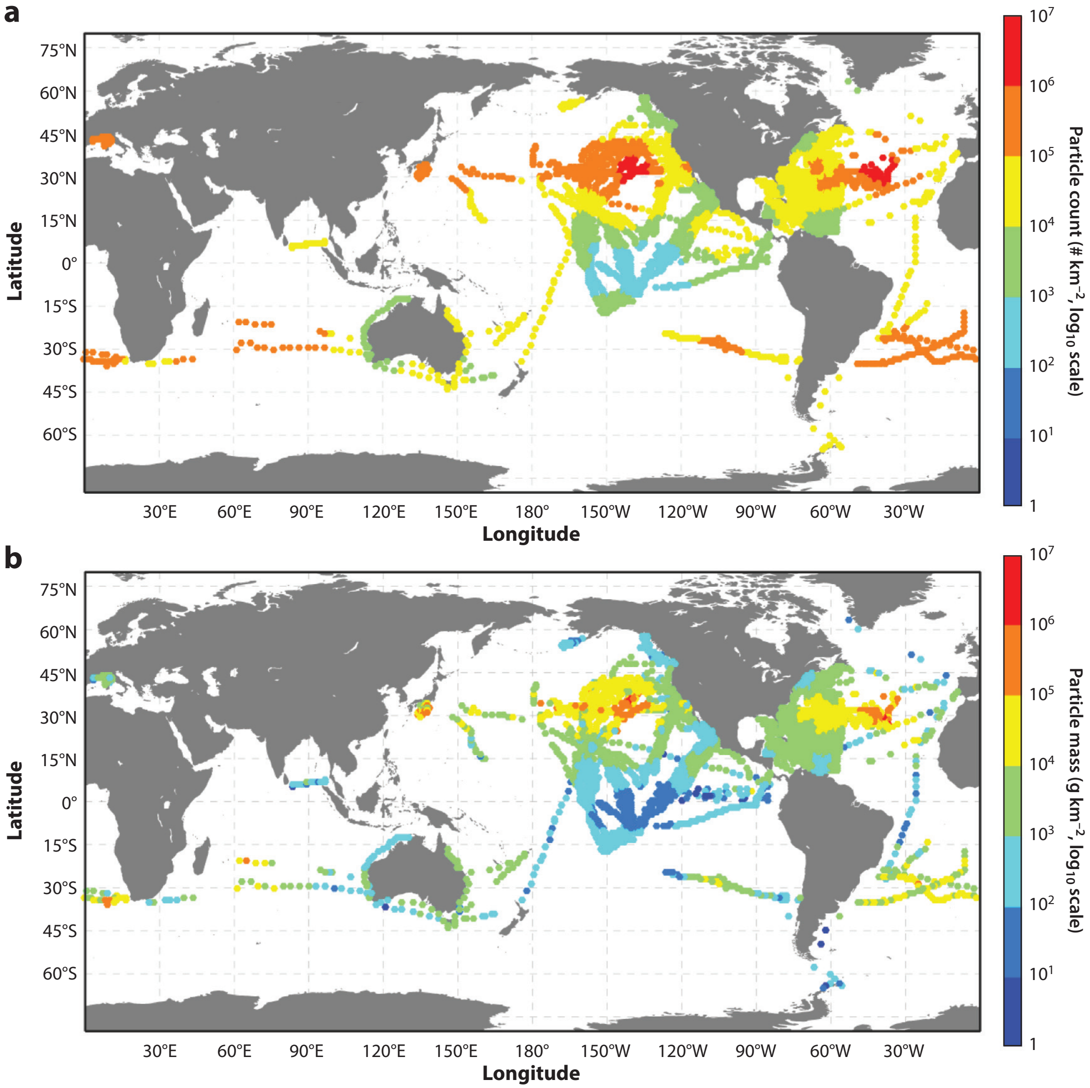

Figure 3

(*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.

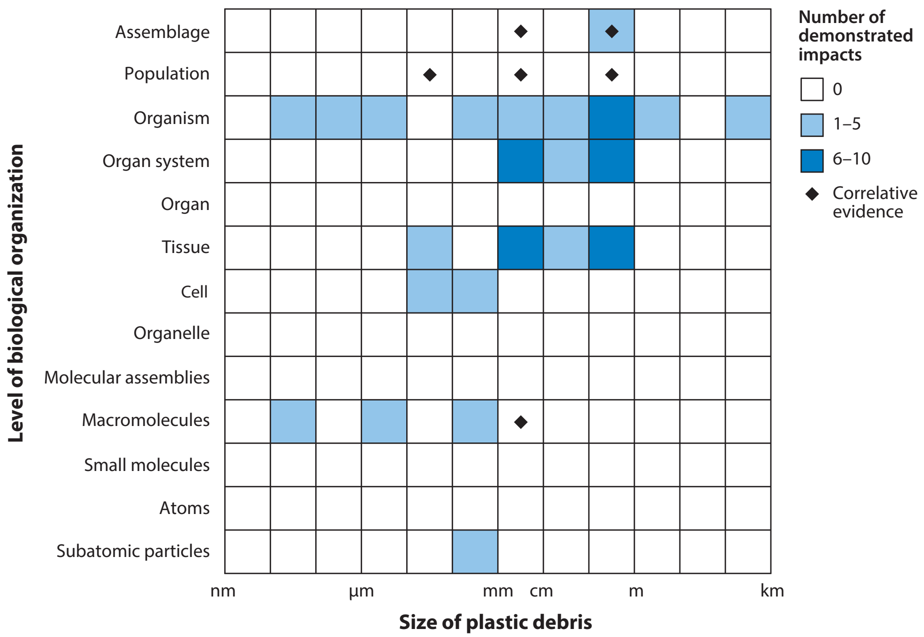

Figure 4

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.

DISCLOSURE STATEMENT

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

ACKNOWLEDGMENTS

The frameworks presented in this review are a product of collaborative discussions within the Marine Debris Working Group of the National Center for Ecological Analysis and Synthesis at the University of California, Santa Barbara, supported by Ocean Conservancy. I bear full responsibility for the interpretation and any omissions or errors. I thank C.R. Rochman for helpful discussions about ecological impacts and for compiling **Figure 4**. The review was improved by comments on various drafts by J. Jambeck, N.J. Mallos, C.R. Rochman, and R.C. Thompson. My work on this review was supported by the National Science Foundation (OCE-1260403).

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Annual Review of
Marine Science

Volume 9, 2017

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Errata

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