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Reviews

Plastic and marine turtles: a review and call for research

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Plastic debris is now ubiquitous in the marine environment affecting a wide range of taxa, from microscopic zooplankton to large vertebrates. Its persistence and dispersal throughout marine ecosystems has meant that sensitivity toward the scale of threat is growing, particularly for species of conservation concern, such as marine turtles. Their use of a variety of habitats, migratory behaviour, and complex life histories leave them subject to a host of anthropogenic stressors, including exposure to marine plastic pollution. Here, we review the evidence for the effects of plastic debris on turtles and their habitats, highlight knowledge gaps, and make recommendations for future research. We found that, of the seven species, all are known to ingest or become entangled in marine debris. Ingestion can cause intestinal blockage and internal injury, dietary dilution, malnutrition, and increased buoyancy which in turn can result in poor health, reduced growth rates and reproductive output, or death. Entanglement in plastic debris (including ghost fishing gear) is known to cause lacerations, increased drag—which reduces the ability to forage effectively or escape threats—and may lead to drowning or death by starvation. In addition, plastic pollution may impact key turtle habitats. In particular, its presence on nesting beaches may alter nest properties by affecting temperature and sediment permeability. This could influence hatchling sex ratios and reproductive success, resulting in population level implications. Additionally, beach litter may entangle nesting females or emerging hatchlings. Lastly, as an omnipresent and widespread pollutant, plastic debris may cause wider ecosystem effects which result in loss of productivity and implications for trophic interactions. By compiling and presenting this evidence, we demonstrate that urgent action is required to better understand this issue and its effects on marine turtles, so that appropriate and effective mitigation policies can be developed.

Keywords: ecosystem effects, entanglement, ghost fishing, ingestion, marine debris, marine turtle, nesting beaches, plastic pollution.

Introduction

Between 1950 and 2015, the total annual global production of plastics grew from 1.5 million t to 299 million t (PlasticsEurope, 2015). As a result, the abundance and spatial distribution of plastic pollution, both on land and at sea, is increasing (Barnes *et al.*, 2009; Jambeck *et al.*, 2015). Indeed, plastic items have become the principal constituent of marine debris, the majority originating from land-based sources, such as landfill sites, with the remaining deriving from human activities, such as fishing (Barnes *et al.*, 2009; Ivar do Sul *et al.*, 2011).

Of particular concern is the longevity of plastic debris and its wide dispersal ability (Barnes *et al.*, 2009; Wabnitz and Nichols,

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2010; Reisser *et al.*, 2014b). It has been recorded worldwide in a vast range of marine habitats, including remote areas far from human habitation (Barnes *et al.*, 2009; Ivar do Sul *et al.*, 2011). Transported across the globe by winds and oceanic currents, high concentrations of floating plastic can accumulate in convergence zones, or gyres, as well as exposed coastlines (Cózar *et al.*, 2014; Reisser *et al.*, 2014b; Schuyler *et al.*, 2014). Enclosed seas, such as the Mediterranean basin, also experience particularly high levels of plastic pollution due to densely populated coastal regions and low diffusion from limited water circulation (Cózar *et al.*, 2015). Once seaborne, plastic persists in the marine environment, fragmenting into smaller pieces as a result of wave action, exposure to UV and physical abrasion (Andrady, 2015). Small particles are highly bioavailable to a wide spectrum of marine organisms (Lusher, 2015). Furthermore, the hydrophobic properties and large surface area to volume ratio of microplastics (fragments of <5 mm in diameter) can lead to the accumulation of contaminants, such as heavy metals and polychlorinated biphenyls (PCBs), from the marine environment. These chemicals, and those incorporated during production (such as plasticizers), can leach into biological tissue upon ingestion, potentially causing cryptic sublethal effects that have rarely been investigated (Koelmans, 2015).

For some species, plastics could present a major threat through ingestion, entanglement, the degradation of key habitats, and wider ecosystem effects (Barnes *et al.*, 2009; Vegter *et al.*, 2014; Gall and Thompson, 2015). Among these species are the marine turtles, whose complex life histories and highly mobile behaviour can make them particularly vulnerable to the impacts of plastic pollution (Arthur *et al.*, 2008; Ivar do Sul *et al.*, 2011; Schuyler *et al.*, 2014). As concern grows for the issue of marine plastic and the associated implications for biodiversity, it is essential to assess the risks faced by key species (Vegter *et al.*, 2014). Understanding vulnerability is necessary for setting research priorities, advising management decisions, and developing appropriate mitigation measures (Schuyler *et al.*, 2014; Vegter *et al.*, 2014). This is particularly pertinent given that marine turtles are of conservation concern and often seen as "flagships" for marine conservation issues (Eckert and Hemphill, 2005).

Here, we carry out a comprehensive review of the state of knowledge concerning this anthropogenic hazard and how it impacts marine turtles, and highlight a range of research and innovative methods that are urgently needed. To do so, we searched ISI Web of Knowledge and Google Scholar for the terms *plastic, plastic pollution, marine debris, marine litter, ingestion, entanglement, entrapment, ghost nets and ghost fishing. Plastic* and *debris* were also searched for in conjunction with *beach, sand, coral reef, sea grass beds*, and *fronts*. Alongside each search term, we also included the word *turtle.* We found that the number of peer-reviewed publications per year (between 1985 and 2014) has generally increased over time (Figure 1a) and a descriptive overview of the 64 peer-reviewed studies is given in Table 1 (Ingestion) and Table 2 (Entanglement). We structure our review in five major sections looking at (i) ingestion, (ii) entanglement, (iii) impacts to nesting beaches, and (iv) wider ecosystem effects and then suggest priorities for (v) future research.

Ingestion

There are two potential pathways by which turtles may ingest plastic; directly or indirectly. Direct consumption of plastic fragments is well documented and has been observed in all marine turtle species (Carr, 1987; Bjorndal *et al.*, 1994; Hoarau *et al.*, 2014; Schuyler *et al.*, 2014; Figure 2a). Accidental ingestion may occur when debris is mixed with normal dietary items. For instance, one study found that juvenile green turtles (*Chelonia mydas*) consumed debris because it was attached to the macroalgae they target directly (Di Beneditto and Awabdi, 2014). Alternatively, plastic ingestion may be a case of mistaken identity. As turtles are primarily visual feeders, they may misidentify items, such as shopping bags, plastic balloons, and sheet plastic, as prey and actively select them for consumption (Mrosovsky, 1981; Tomás *et al.*, 2002; Gregory, 2009; Hoarau *et al.*, 2014). Hoarau *et al.* (2014) found a high occurrence of plastic bottle lids in the loggerhead turtles (*Caretta caretta*) they examined and surmised that the lids' round shape and presence floating near the surface visually resemble neustonic organisms normally preyed upon. Laboratory trials have found that turtles are able to differentiate between colours and so the visual properties of plastic are likely to be important factors determining the probability of ingestion (Bartol and Musick, 2003; Swimmer *et al.*, 2005; Schuyler *et al.*, 2012). A number of studies have found that white and transparent plastics are the most readily consumed colours (Tourinho *et al.*, 2010; Schuyler *et al.*, 2012; Camedda *et al.*, 2014; Hoarau *et al.*, 2014). It is not certain, however, whether this trend is a result of selectivity by the turtles or due to the differing proportions of plastic types and colours in the environment (Schuyler *et al.*, 2012; Camedda *et al.*, 2014). Aside from visual cues, perhaps microbial biofilm formation on plastic debris and the associated invertebrate grazers (Reisser *et al.*, 2014a) cause the particles to emit other sensory cues (such as smell and taste) which could lead turtles to consume them. This, however, remains to be investigated.

Indirect ingestion may occur when prey items, such as molluscs and crustaceans that have been shown to ingest and assimilate microplastic particles in their tissues (Cole *et al.*, 2013; Wright *et al.*, 2013), are consumed by carnivorous species. Although not yet investigated for marine turtles, trophic transfer has been inferred in other marine vertebrates, specifically pinnipeds (McMahon *et al.*, 1999; Eriksson and Burton, 2003). For example, the prey of the Hooker's sea lion (*Phocarctos hookeri*), myctophid fish, ingest microplastic particles. Subsequently, the otoliths (ear bones) of these fish have been found alongside plastic particles within the sea lion scat, suggesting a trophic link (McMahon *et al.*, 1999). This indirect ingestion may lead to sublethal effects that are difficult to identify, quantify and attribute to plastic ingestion as opposed to other water quality issues (Baulch and Perry, 2014; Vegter *et al.*, 2014; Gall and Thompson, 2015). These are discussed later in this section.

It is likely that feeding ecology and diet, as well as habitat use in relation to areas of high plastic density, determine the likelihood and consequences of plastic ingestion (Bond *et al.*, 2014). These differ among turtle life stages, regional populations and species, meaning that there are likely to be inter- and intraspecies variation in the densities and types of plastic encountered and potentially consumed (Schuyler *et al.*, 2014).

Life stage

Both the likelihood of exposure to and consequences of ingestion differ across life stage. Post-hatchlings and juveniles of six of the seven marine turtle species undergo a period of pelagic drifting, known as the "*lost year*". Although flatback turtles (*Natator depressus*) lack an oceanic dispersal stage, their habitat use during the post-hatchling phase is still likely to be influenced by bathymetry and coastal currents (Hamann *et al.*, 2011). Currents transport hatchlings away from their natal beaches, often to oceanic convergence zones, such as fronts or downwelling areas (Bolten, 2003; Boyle

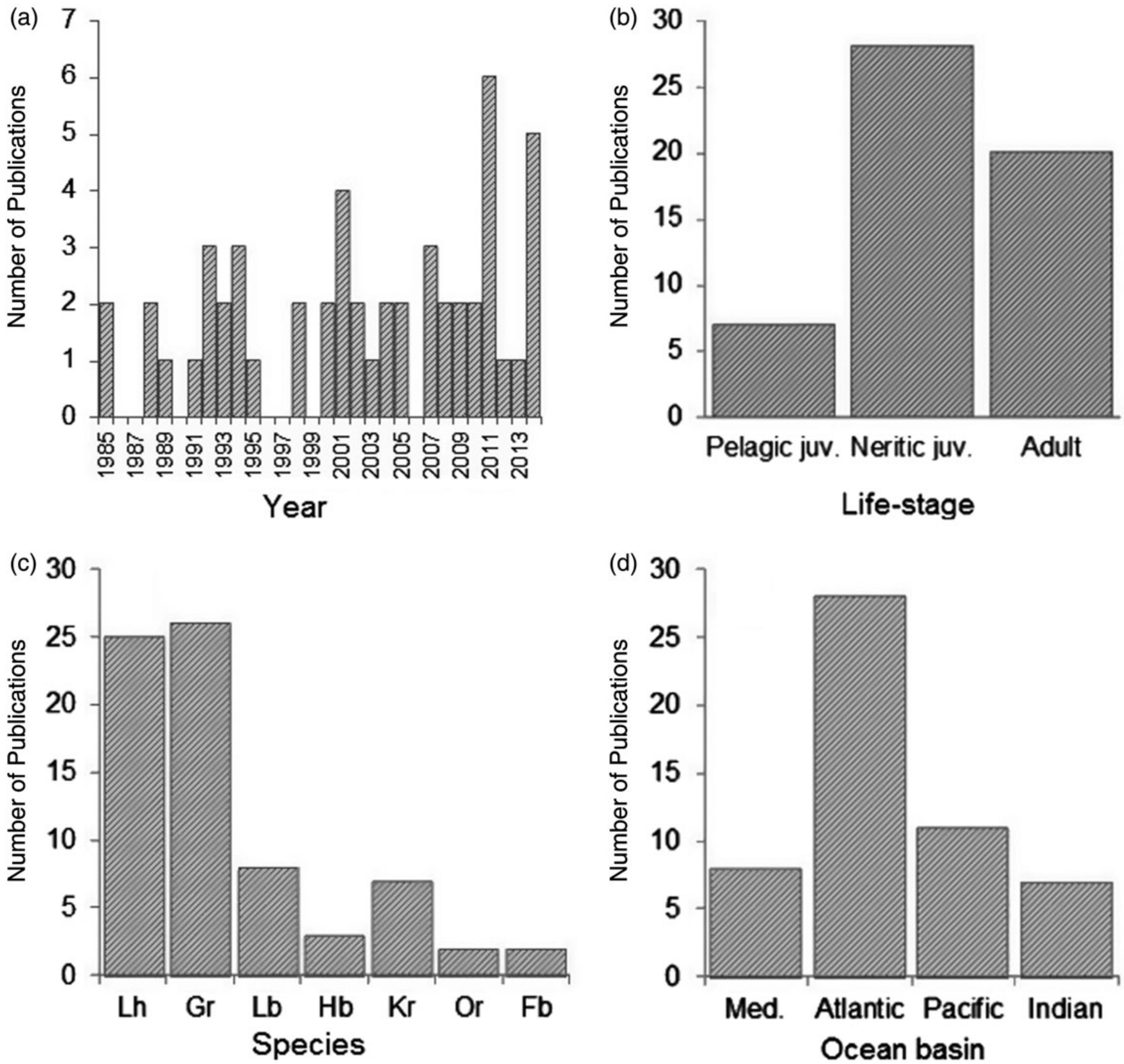

Figure 1. Number of publications returned from literature search per (a) year (between 1985 and 2014), (b) life stage, (c) species (Lh, Loggerhead; Gr, Green; Lb, Leatherback; Hb, Hawksbill; Kr, Kemp's ridley; Or, Olive ridley; Fb, Flatback), and (d) Ocean basin.

et al., 2009; Scott *et al.*, 2014). These areas can be highly productive and act as foraging hotspots for many marine taxa, including fish, seabirds, and marine turtles (Witherington, 2002; Scales *et al.*, 2014; Schuyler *et al.*, 2014). However, along with food, advection also draws in and concentrates floating anthropogenic debris, increasing the likelihood of exposure to plastic. This spatial overlap potentially creates an ecological trap for young turtles (Carr, 1987; Tomás *et al.*, 2002; Battin, 2004; Witherington *et al.*, 2012; Cózar *et al.*, 2014). Their vulnerability is further intensified by indiscriminate feeding behaviour, often mistaking plastic for prey items or accidentally ingesting debris while grazing on organisms that are encrusted on such items (McCauley and Bjorndal, 1999; Schuyler *et al.*, 2012; Hoarau *et al.*, 2014). Additionally, turtles in early life history stages, that are small in size, may be at higher risk of mortality from plastic ingestion due to their smaller, less robust, digestive tracts (Boyle, 2006; Schuyler *et al.*, 2012). During our literature search, we found that of all the life stages, young "*lost year*" juveniles are the most data deficient, but potentially the most vulnerable (Figure 1b).

After the post-hatchling pelagic stage, most populations of chelonid (hard-shelled) species, such as loggerheads, greens, and hawksbills (*Eretmochelys imbricata*), undergo an ontogenetic shift in feeding behaviour where they may switch to benthic foraging in neritic areas (although some populations forage pelagically even in larger size classes; Tomás *et al.*, 2001; Witherington, 2002; Hawkes *et al.*, 2006; Arthur *et al.*, 2008; Schuyler *et al.*, 2012). Some foraging areas experience higher concentrations of plastic debris due to physical processes, for example, frontal systems or discharging rivers, and when such accumulations overlap with turtle foraging grounds, high rates of ingestion may be observed (González Carman *et al.*, 2014). Indeed, González Carman *et al.* (2014) reported that 90% of the juvenile green turtles examined had ingested anthropogenic

Table 1. Summary of all studies on plastic ingestion by marine turtles.

Species	Ocean basin	Study area	Reference	Year of study	*n*	Occurrence %	CCL range	Pelagic juvenile	Neritic juvenile	Adult
Loggerhead (*Caretta caretta*)	Mediterranean Sea	Tyrrhenian sea (Tuscany coast)	Campani *et al.* (2013)	2010–2011	31	71	29.0–73.0	X	✓	✓
		Adriatic sea (Croatia, Slovenia)	Lazar and Gračan (2011)	2001–2004	54	35.2	25.0–79.2	X	✓	✓
		Central Mediterranean (Sicily)	Russo *et al.* (2003)	1994–1998	44	15.9	Unknown	n.a.	n.a.	n.a.
		Central Mediterranean (Italy)	Casale *et al.* (2008)	2001–2005	79	48.1	25.0–80.3	X	✓	✓
		Western Mediterranean (Sardinia)	Camedda *et al.* (2014)	2008–2012	121	14	51.38 ± 1.13	X	✓	✓
		Western Mediterranean (Balearic archipelago)	Revelles *et al.* (2007)	2002–2004	19	37	Unknown	n.a.	n.a.	n.a.
		Western Mediterranean (Spain)	Tomás *et al.* (2002)	n.a.	54	75.9	34.0–69.0	✓	✓	✓
		Eastern Mediterranean (Turkey)	Kaska *et al.* (2004)	2001	65	5	Unknown	n.a.	n.a.	n.a.
	Atlantic ocean	Northeastern Atlantic (Azores, Portugal)	Frick *et al.* (2009)	1986–2001	12	25	9.3–56.0	✓	✓	X
		Northwestern Atlantic (Georgia, USA)	Frick *et al.* (2001)	n.a.	12	0	59.4–77.0	X	✓	✓
		Northwestern Atlantic (Virginia)	Seney and Musick (2007)	1983–2002	166	0	41.6–98.5(SCL)	X	✓	✓
		Northwestern Atlantic (Florida, USA)	Bjorndal *et al.* (1994)	1988–1993	1	100	52	X	✓	X
		Gulf of Mexico (Texas, USA)	Plotkin *et al.* (1993)	1986–1988	82	51.2	51.0–105.0	X	✓	✓
		Gulf of Mexico (Texas, USA)	Plotkin and Amos (1990)	1986–1988	88	52.3	Unknown	n.a.	n.a.	n.a.
		Northwestern Atlantic (New York, USA)	Sadove and Morreale (1989)	1979–1988	103	2.9	Unknown	n.a.	n.a.	n.a.
		Northwestern Atlantic (Florida, USA)	Witherington (1994)	n.a.	50	32	4.03–5.63	✓	X	X
		Gulf of Mexico (Texas and Louisiana, USA)	Cannon (1998)	1994	20	5	Unknown	n.a.	n.a.	n.a.
		Southwestern Atlantic (Brazil)	Bugoni *et al.* (2001)	1997–1998	10	10	63.0–97.0	X	X	✓
	Pacific Ocean	Southwestern (Australia)	Boyle and Limpus (2008)	n.a.	7	57.1	4.6–10.6	✓	X	X
		Central north (Hawaii, USA)	Parker *et al.* (2005)	1990–1992	52	34.6	13.5–74.0	✓	✓	✓
		Northeastern (Shuyak Island, Alaska)	Bane (1992)	1991	1	100	64.2	X	✓	X
		Northeastern (California)	Allen (1992)	1992	1	100	59.3	X	✓	X
		Northeastern (Baja California, Mexico)	Peckham *et al.* (2011)	2003–2007	82	0	Unknown	n.a.	n.a.	n.a.
	Indian Ocean	Southwestern (Reunion Islands)	Hoarau *et al.* (2014)	2007–2013	50	51.4	68.7 ± 4.99	X	✓	✓
		Northeastern (Queensland, Australia)	Limpus and Limpus (2001)	1989–1998	47	0	Unknown	n.a.	n.a.	n.a.
Green (*Chelonia mydas*)	Mediterranean Sea	Central Mediterranean (Sicily)	Russo *et al.* (2003)	1994–1998	1	0	37.8	X	✓	X
	Atlantic ocean	Southwestern Atlantic (Río de la Plata)	González Carman *et al.* (2014)	2008–2011	64	90	31.3–52.2	X	✓	X
		Southwestern Atlantic (Brazil)	Barreiros and Barcelos (2001)	2000	1	100	40.5	X	✓	X
		Southwestern Atlantic (Brazil)	Santos *et al.* (2011)	2007–2008	15	20	35.1–60.0	X	✓	X
		Southwestern Atlantic (Brazil)	da Silva Mendes *et al.* (2015)	2008–2009	20	45	33.0–44.0	X	✓	X
		Southwestern Atlantic (Brazil)	Bugoni *et al.* (2001)	1997–1998	38	60.5	28.0–50.0	X	✓	X
		Northwestern Atlantic (New York, USA)	Sadove and Morreale (1989)	1979–1988	15	6.6	Unknown	n.a.	n.a.	n.a.
		Northwestern Atlantic (Florida, USA)	Bjorndal *et al.* (1994)	1988–1993	43	55.8	20.6–42.7	X	✓	X
		Gulf of Mexico (Texas and Louisiana, USA)	Cannon (1998)	1994	6	33.3	Unknown	n.a.	n.a.	n.a.
		Gulf of Mexico (Texas, USA)	Plotkin and Amos (1990)	1986–1988	15	46.7	Unknown	n.a.	n.a.	n.a.
		Southwestern Atlantic (Brazil)	Guebert-Bartholo *et al.* (2011)	2004–2007	80	70	29–73	X	✓	✓

		Southwestern Atlantic (Brazil)	Di Beneditto and Awabdi (2014)	n.a.	49	59.2	Unknown	n.a.	n.a.	n.a.
		Southwestern Atlantic (Brazil)	Tourinho *et al.* (2010)	2006 – 2007	34	100	31.5 – 56.0	X	✓	X
		Southwestern Atlantic (Brazil)	Stahelin *et al.* (2012)	2010	1	100	39	X	✓	X
		Southwestern Atlantic (Brazil)	Poli *et al.* (2014)	2009 – 2010	104	12.5	24.0 – 123.5	X	✓	✓
		Northwestern Atlantic (Florida, USA)	Foley *et al.* (2007)	2000 – 2001	44	2	Unknown	n.a.	n.a.	n.a.
	Pacific Ocean	Southwestern (Australia)	Boyle and Limpus (2008)	n.a.	57	54.3	5.5 – 11.3	✓	X	X
		Southeastern (San Andres, Peru)	Quiñones *et al.* (2010)	1987	192	42	Unknown	n.a.	n.a.	n.a.
		Southeastern (Galápagos Islands, Ecuador)	Parra *et al.* (2011)	2009 – 2010	53	3.3	53.0 – 93.0	X	✓	✓
		Central north (Hawaii, USA)	Parker *et al.* (2011)	1990 – 2004	10	70	30.0 – 70.0	X	✓	✓
		Northeastern (Baja California, Mexico)	López-Mendilaharsu *et al.* (2005)	2000 – 2002	24	0	Unknown	n.a.	n.a.	n.a.
		Northeastern (Gulf of California)	Seminoff *et al.* (2002)	1995 – 1999	7	29.5	Unknown	n.a.	n.a.	n.a.
	Indian Ocean	Northeastern (Torres Strait, Australia)	Garnett *et al.* (1985)	1979	44	0	Unknown	n.a.	n.a.	n.a.
		Northwestern (UAE)	Hasbún *et al.* (2000)	1997	13	0	35 – 105.5	X	✓	✓
		Northwestern (Oman)	Ross (1985)	1977 – 1979	9	0	Unknown	n.a.	n.a.	n.a.
Leatherback (*Dermochelys coriacea*)	Mediterranean Sea	Central Mediterranean (Sicily)	Russo *et al.* (2003)	1994 – 1998	5	40	131 – 145	X	X	✓
	Atlantic ocean	Northeastern Atlantic (Gwynedd, Wales)	Eckert and Luginbuhl (1988)	1988	1	100	256	X	X	✓
		Northeastern Atlantic (Bay of Biscay)	Duguy *et al.* (2000)	1978 – 1995	87	55	Unknown	n.a.	n.a.	n.a.
		Northeastern Atlantic (Azores)	Barreiros and Barcelos (2001)	2000	1	100	144	X	X	✓
		Northwestern Atlantic (Sable Island, Nova Scotia)	Lucas (1992)	1984 – 1991	2	100	Unknown	n.a.	n.a.	n.a.
		Northwestern Atlantic (New York, USA)	Sadove and Morreale (1989)	1979 – 1988	85	11.7	Unknown	n.a.	n.a.	n.a.
		Southwestern Atlantic (Brazil)	Bugoni *et al.* (2001)	1997 – 1998	2	50	135 – 135	X	X	✓
	Pacific Ocean	Central-north Pacific (Midway Island)	Davenport *et al.* (1993)	1993	1	100	Unknown	n.a.	n.a.	n.a.
	All	General	Mrosovsky *et al.* (2009)	1885 – 2007	408	34	Unknown	n.a.	n.a.	n.a.
Hawksbill (*Eretmochelys imbricata*)	Atlantic ocean	Gulf of Mexico (Texas, USA)	Plotkin and Amos (1990)	1986 – 1988	8	87.5	Unknown	n.a.	n.a.	n.a.
		Southwestern Atlantic (Brazil)	Poli *et al.* (2014)	2009 – 2010	15	33.3	30.9 – 91.2	X	✓	✓
	Pacific Ocean	Northeastern (Costa Rica)	Arauz Almengor and Morera Avila (1994)	1992	1	100	24.5	✓	X	X
Kemp's ridley (*Lepidochelys kempii*)	Atlantic ocean	Northwestern Atlantic (New York, USA)	Burke *et al.* (1994)	1985 – 1989	18	0	Unknown	n.a.	n.a.	n.a.
		Northwestern Atlantic (New York, USA)	Sadove and Morreale (1989)	1979 – 1988	122	0	Unknown	n.a.	n.a.	n.a.
		Northwestern Atlantic (Florida, USA)	Bjorndal *et al.* (1994)	1988 – 1993	7	0	28.6 – 66.2	X	✓	✓
		Gulf of Mexico (Texas and Louisiana, USA)	Cannon *et al.* (1998)	1994	167	5.4	Unknown	n.a.	n.a.	n.a.
		Gulf of Mexico (Texas, USA)	Plotkin and Amos (1988)	1986 – 1988	104	29.8	Unknown	n.a.	n.a.	n.a.
		Gulf of Mexico (Texas, USA)	Shaver (1991)	1983 – 1989	101	29	5.2 – 71.0	✓	✓	✓
		Gulf of Mexico (Texas, USA)	Shaver (1998)	1984	37	19	Unknown	n.a.	n.a.	n.a.
Olive ridley (*Lepidochelys olivacea*)	Atlantic ocean	Southwestern Atlantic (Brazil, Parabia)	Mascarenhas *et al.* (2004)	2004	1	100	66	X	X	✓
		Southwestern Atlantic (Brazil)	Poli *et al.* (2014)	2009 – 2010	2	100	60.0 – 63.3	X	✓	✓
Flatback (*Natator depressus*)	Indian Ocean	Northeastern (Darwin, Australia)	Chatto (1995)	1994	1	100	25.5	X	✓	X

CCL, curved carapace length.

Table 2. Summary of all studies on entanglement in plastic debris by marine turtles.

Species	Ocean basin	Study area	Reference	Year of study	*n*	CCL range	Pelagic juvenile	Neritic juvenile	Adult	Debris type
Loggerhead (*Caretta caretta*)	Atlantic ocean	Northeastern (Boa Vista, Cape Verde Islands)	Lopez-Jurado *et al.* (2003)	2001	10	62.0–89.0	X	✓	✓	Fishing
		Northeastern (Terceira Island, Azores)	Barreiros and Raykov (2014)	2004–2008	3	37.3–64.1	X	✓	✓	Fishing/land-based
	Mediterranean Sea	Tyrrhenian sea (Island of Panarea, Sicily)	Bentivegna (1995)	1994	1	48.5	X	✓	X	Land-based
		Central Mediterranean (Italy)	Casale *et al.* (2010)	1980–2008	226	3.8–97.0	✓	✓	✓	Fishing/land-based
Green (*Chelonia mydas*)	Indian Ocean	Northeastern (Darwin, Australia)	Chatto (1995)	1994	1	35	X	✓	X	Fishing
		Northeastern (Australia)	Wilcox *et al.* (2013)	2005–2009	14	Unknown	n.a.	n.a.	n.a.	Fishing
Hawksbill (*Eretmochelys imbricata*)	Indian Ocean	Northeastern (Darwin, Australia)	Chatto (1995)	1994	1	32.5	X	✓	X	Fishing
		Northeastern (Australia)	Wilcox *et al.* (2013)	2005–2009	35	Unknown	n.a.	n.a.	n.a.	Fishing
Olive ridley (*Lepidochelys olivacea*)	Indian Ocean	Northeastern (McCluer Island, Australia)	Jensen *et al.* (2013)	Unknown	44	Unknown	n.a.	n.a.	n.a.	Fishing
		Northeastern (Australia)	Wilcox *et al.* (2013)	2005–2009	53	Unknown	n.a.	n.a.	n.a.	Fishing
		Northeastern (Australia)	Chatto (1995)	1994	2	64	X	X	✓	Fishing
	Atlantic Ocean	Southwestern (Brazil)	Santos *et al.* (2012)	1996–2011	18	2.01–80.0	X	✓	✓	Fishing
Flatback (*Natator depressus*)	Indian Ocean	Northeastern (Darwin, Australia)	Chatto (1995)	1994	1	25.5	X	✓	X	Land-based
		Northeastern (Australia)	Wilcox *et al.* (2013)	2005–2009	3	Unknown	n.a.	n.a.	n.a.	Fishing
Multiple	Indian Ocean	Northeastern (Australia)	Wilcox *et al.* (2014)	2005–2012	336	Unknown	n.a.	n.a.	n.a.	Fishing

CCL, curved carapace length.

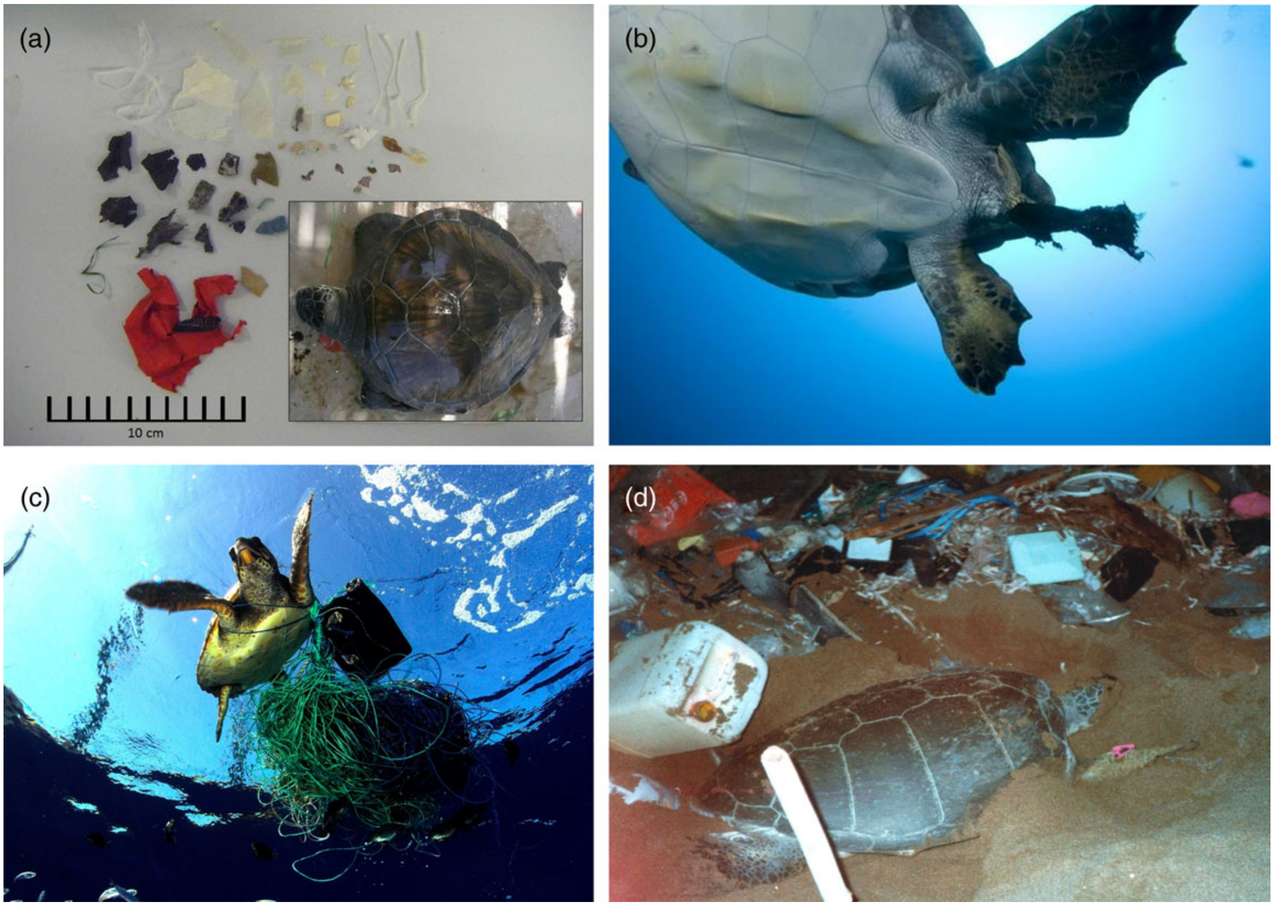

Figure 2. Plastics and marine turtles: (a) plastic fragments extracted from the digestive tract of a necropsied juvenile green turtle (inset), found stranded in northern Cyprus (photo: EMD); (b) plastic extruding from a green turtle's cloaca in Cocos Island, Costa Rica (photo: Cristiano Paoli); (c) loggerhead turtle entangled in fishing gear in the Mediterranean Sea (north of Libya) (photo: Greenpeace©/Carè©/Marine Photobank); (d) female green turtle attempting to nest among beach litter, northern Cyprus in 1992 before the commencement of annual beach cleaning (photo: ACB).

debris and postulated that, aside from the high concentrations of debris, poor visibility (caused by estuarine sediment) and therefore a reduced ability to discriminate among ingested items may also be a factor.

Species

The results from our literature search show that, of all peer-reviewed publications (between 1985 and 2014; $n = \sim 6668$) looking at marine turtles, the proportion that investigated occurrences of plastic ingestion is relatively low, ranging from 1 to 2% depending on species. We found that the majority of these studies focused on loggerhead ($n = 24$; 44%) and green turtles ($n = 23$; 43%) in contrast to a small number of reports on the leatherback (*Dermochelys coriacea*; $n = 7$, 13%), Kemp's ridley (*Lepidochelys kempii*; $n = 7$; 13%), hawksbill ($n = 3$; 6%), olive ridley (*Lepidochelys olivacea*; $n = 2$; 4%), and flatback turtles ($n = 2$; 4%; Figure 1c). These biases, however, are broadly reflected by those observed for general turtle studies (green = 35%, loggerhead = 31%, leatherback = 14%, hawksbill = 9%, olive ridley = 5%, kemps ridley = 4%, and flatback = 1%). This relationship demonstrates the need for caution when interpreting apparent patterns based on the number of observations of plastic ingestion among species.

We also found that the majority of research was carried out in the Atlantic Ocean basin ($n = 28$ of 55 publications on plastic ingestion by turtles; Figure 1d). These strong biases towards certain species/regions demonstrate a need to expand research to better understand plastic ingestion for the taxon, globally.

Among marine turtles, there are profound interspecific differences in feeding strategies, diet, and habitat use that could result in varying likelihoods of exposure to and consequences of plastic ingestion (Bjorndal, 1997; Schuyler *et al.*, 2014). For example, the generalist feeding strategy of loggerhead turtles seems to put them at high risk of ingesting plastic, but their ability to defecate these items, due to a wide alimentary tract, however, demonstrates a certain degree of tolerance (in adults and subadults; Bugoni *et al.*, 2001; Tomás *et al.*, 2001, 2002; Hoarau *et al.*, 2014). This, though, may not mitigate the sublethal effects which may occur as a result of plastic ingestion (see the Ecological effects section). Although not heavily studied when compared with the other turtle species (Figure 1c), ingestion rates by Kemp's ridley turtles appear to be low. This may be because they specialize in hunting active prey, such as crabs, which plastic debris is less likely to be mistaken for (Bjorndal *et al.*, 1994). Nonetheless, a potential issue for benthic feeding, carnivorous marine turtle species, such as Kemp's ridley, olive ridley, loggerhead, and flatback turtles, is indirect ingestion of microplastics through consumption of contaminated invertebrate prey, such as molluscs and crustaceans (Parker *et al.*, 2005; Casale *et al.*, 2008) and any associated sediments. Green turtles too are mostly benthic feeders but are largely herbivorous (Bjorndal, 1997). Their preference for sea grass or algae may lead to a greater likelihood of ingesting clear soft plastics resembling their natural food in structure and behaviour. A study in southeastern Brazil found that 59% of juvenile green turtles stomachs contained flexible and hard plastic debris (clear, white, and coloured) and Nylon filaments (Di Beneditto and Awabdi, 2014); another found that 100% of green turtle stomachs examined contained at least one plastic item (Bezerra and Bondioli, 2011). Hawksbills, although omnivorous, prefer to consume sponges and algae, acting as important trophic regulators on coral reefs (León and Bjorndal, 2002). While clean-up surveys on coral reefs show that plastic is present in such habitats (Abu-Hilal and Al-Najjar, 2009), data on the ingestion rates and selectivity for hawksbills are lacking (Figure 1c). Peer-reviewed studies investigating ingestion by flatbacks are also scarce, but we found reports that in 2003, a flatback turtle died following ingestion of a balloon (Greenland and Limpus, 2003) and in 2014, four out of five stranded post-hatchling flatback turtles had ingested plastic fragments (StrandNet Database, 2015). Pelagic species that forage on gelatinous prey, such as leatherbacks, are also susceptible to plastic ingestion and Mrosovsky *et al.* (2009) estimated that approximately one-third of all adult leatherbacks autopsied from 1968 to 2007 had ingested plastic. This is thought to be due to similarities to prey items, such as jellyfish, acting as sensory cues to feed (Schuyler *et al.*, 2014).

Ecological effects

The effects of plastic ingestion can be both lethal and sublethal, the latter being far more difficult to detect and likely more frequent (Hoarau *et al.*, 2014; Schuyler *et al.*, 2014; Gall and Thompson, 2015). Tourinho *et al.* (2010) reported that 100% of stranded green turtles ($n = 34$) examined in southeastern Brazil had ingested anthropogenic debris, the majority of which was plastic, but the deaths of only three of these turtles could be directly linked to its presence. Damage to the digestive system and obstruction is the most conspicuous outcome and is often observed in stranded individuals (Figure 2b; Camedda *et al.*, 2014). The passage of hard fragments through the gut can cause internal injuries and intestinal blockage (Plotkin and Amos, 1990; Derraik, 2002). Accidental ingestion of plastic fishing line may occur when turtles consume baited hooks (e.g. Bjorndal *et al.*, 1994). As the line is driven through the gut by peristalsis, it can become constricted, causing damage, such as tearing to the intestinal wall (Parga, 2012; Di Bello *et al.*, 2013).

In some cases, the sheer volume of plastic within the gut is noticeable during necropsy or possibly via X-ray or internal examination. Small amounts of anthropogenic debris, however, have been found to block the digestive tract (Bjorndal *et al.*, 1994; Bugoni *et al.*, 2001; Schuyler *et al.*, 2014; Santos *et al.*, 2015). For example, Santos *et al.* (2015) found that only 0.5 g of debris (consisting of mainly soft plastic and fibres) was enough to block the digestive tract of a juvenile green turtle, ultimately causing its death. Additionally, hardened faecal material has been known to accumulate as a result of the presence of plastic and the associated blockage to the gastrointestinal system (Davenport *et al.*, 1993; Awabdi *et al.*, 2013). On the contrary, it is possible for significant amounts of plastic to accumulate and remain within the gut without causing lethal damage (Hoarau *et al.*, 2014). For example, Lutz (1990) reported that plastic pieces remained in the gut of a normally feeding captive turtle for four months. In the long term, however, a reduction in feeding stimulus and stomach capacity could lead to malnutrition through dietary dilution which occurs when debris items displace food in the gut, reducing the turtles ability to feed (McCauley and Bjorndal, 1999; Plot and Georges, 2010; Tourinho *et al.*, 2010). Experimental evidence has shown that dietary dilution causes post-hatchling loggerheads to exhibit signs of reduced energy and nitrogen intake (McCauley and Bjorndal, 1999). Post-hatchlings and juvenile turtles are of particular concern because their smaller size means that starvation is likely to occur more rapidly which has consequences for the turtle's ability to obtain sufficient nutrients for growth (McCauley and Bjorndal, 1999; Tomás *et al.*, 2002).

The presence of large quantities of buoyant material within the intestines may affect turtles' swimming behaviour and buoyancy control. This is especially crucial for deep diving species, such as the leatherbacks (Fossette *et al.*, 2010) and small benthic foragers, such as flatbacks. Additionally, plastic ingestion can compromise a female's ability to reproduce. For example, plastic was found to block the cloaca of a nesting leatherback turtle, preventing the passage of her eggs (Plot and Georges, 2010; Sigler, 2014).

Long gut residency times for plastics may lead to chemical contamination as plasticizers, such as Bisphenol A and phthalates, leach out of ingested plastics and can be absorbed into tissues, potentially acting as endocrine disrupters (Oehlmann *et al.*, 2009). Additionally, due to their hydrophobic properties, plastics are known to accumulate heavy metals and other toxins, such as PCBs, from the marine environment which can also be released during digestion (Cole *et al.*, 2015; Wright *et al.*, 2013). Such contaminants have been shown to cause developmental and reproductive abnormalities in many taxa, such as egg-shell thinning and delayed ovulation in birds as well as hepatic stress in fish (Azzarello and Van Vleet, 1987; Wiemeyer *et al.*, 1993; Oehlmann *et al.*, 2009; Rochman *et al.*, 2013a,b; Vegter *et al.*, 2014). To date, the knowledge base regarding these issues in marine turtles is limited.

Indirectly ingested microplastics may pass through the cell membranes and into body tissues and organs where they can accumulate and lead to chronic effects (Wright *et al.*, 2013). The implications of trophic transfer, of both the microplastics and their associated toxins, are as yet unknown (Cole *et al.*, 2013; Wright *et al.*, 2013; Reisser *et al.*, 2014a) and worthy of investigation.

It is possible that the sublethal effects of plastic ingestion, including dietary dilution, reduced energy levels, and chemical contamination, may lead to a depressed immune system function resulting in an increased vulnerability to diseases, such as fibropapillomatosis (Landsberg *et al.*, 1999; Aguirre and Lutz, 2004). Stranded juvenile green turtles in Brazil exhibit both high occurrence of plastic ingestion and incidences of this disease (Santos *et al.*, 2011). Additionally, plastic ingestion may impact health and weaken the turtle's physical condition which could impair the ability to avoid predators and survive anthropogenic threats, such as ship strikes and incidental capture by fisheries, issues which already threaten many marine turtle populations (Lewison *et al.*, 2004; Hazel and Gyuris, 2006; Hoarau *et al.*, 2014). Other longer term consequences could include reduced growth rates, fecundity, reproductive success, and late sexual maturation which could have long-term demographic ramifications for the stability of marine turtle populations (Hoarau *et al.*, 2014; Vegter *et al.*, 2014).

In summary, the potential effects of plastic ingestion on marine turtles are diverse and often cryptic, making it difficult to identify a clear causal link. The sheer scale of possibilities, though, makes this topic one that is in urgent need of further research.

Entanglement

Entanglement in marine debris, such as items from land-based sources and lost fishing gear (known as "ghost gear"), is now recognized as a major threat to many marine species (Figure 2c; Gregory, 2009; Wilcox *et al.*, 2013; Vegter *et al.*, 2014). Their sources are difficult to trace, but their widespread distribution indicates that ocean currents and winds may be dispersal factors (Santos *et al.*, 2012; Jensen *et al.*, 2013; Wilcox *et al.*, 2013). Entanglement is one of the major causes of turtle mortality in many areas including northern Australia and the Mediterranean (Casale *et al.*, 2010; Jensen *et al.*, 2013; Wilcox *et al.*, 2013; Camedda *et al.*, 2014). Despite this, quantitative research on mortality rates is lacking and a large knowledge gap exists in terms of implications for global sea turtle populations (Matsuoka *et al.*, 2005). Our literature search returned just nine peer-reviewed publications directly referring to marine debris entanglement and turtles (Bentivegna, 1995; Chatto, 1995; Lopez-Jurado *et al.*, 2003; Casale *et al.*, 2010; Santos *et al.*, 2012; Jensen *et al.*, 2013; Wilcox *et al.*, 2013, 2014; Barreiros and Raykov, 2014) and of these, seven are related to ghost fishing gear. For individual turtles, the effects of entanglement are injuries, such as abrasions, or loss of limbs; a reduced ability to avoid predators; or forage efficiently due to drag leading to starvation or drowning (Gregory, 2009; Barreiros and Raykov, 2014; Vegter *et al.*, 2014). From a welfare perspective, entanglement may cause long-term suffering and a slow deterioration (Barreiros and Raykov, 2014). In some cases, injuries are so severe that amputation or euthanasia are the only options for rehabilitators (Chatto, 1995; Barreiros and Raykov, 2014).

Ghost nets—mostly consisting of synthetic, non-biodegradable fibres, such as Nylon—may persist in the marine environment for many years, indiscriminately "fishing" an undefinable number of animals (Bentivegna, 1995; Wilcox *et al.*, 2013, 2014; Stelfox *et al.*, 2014). Some nets, which may be several kilometres long, drift passively over large distances (Brown and Macfadyen, 2007; Jensen *et al.*, 2013), eventually becoming bio-fouled by marine organisms and attracting grazers and predators, such as turtles (Matsuoka *et al.*, 2005; Gregory, 2009; Jensen *et al.*, 2013; Stelfox *et al.*, 2014). Although this widespread problem is not unique to turtles, as a taxon, they appear to be particularly vulnerable. For example, a study by Wilcox *et al.* (2013) reported that 80% of the animals found in lost nets off the Australian coast were turtles. It may be, however, that the physical attributes of marine turtles mean they are more persistent in these nets. For example, their robust carapaces are likely to degrade more slowly and could be easier to identify than carcasses of other marine animals.

More recently, Wilcox *et al.* (2014) found that nets with large mesh sizes but smaller twine sizes are more likely to entangle turtles, and larger nets seemed to attract turtles, further increasing their catch rates.

Aside from lost or discarded fishing gear, turtles may become trapped in debris from land-based sources. For example, a juvenile loggerhead was found off the island of Sicily trapped in a bundle of polyethylene packaging twine (Bentivegna, 1995) and a juvenile flatback turtle stranded in Australia after becoming trapped in a woven plastic bag (Chatto, 1995). Reports of such incidences in scientific literature are scarce and it is likely that many individual cases of entanglement are never published (BJG, pers. obs.). Thus, the rates of entanglement in debris, such as sheet plastic and Nylon rope, from land-based sources may be greatly underestimated.

There are few investigations into the susceptibility of the various life stages, but one study found that for olive ridleys, the majority of trapped animals were subadults and adults (Santos *et al.*, 2012). There could be several reasons for this. First, the smaller size of young juveniles enhances their ability to escape. Second, it may be that their carcasses are more readily assimilated into the environment through depredation and decomposition and therefore the evidence of their entanglement is less likely to be discovered. Lastly, it may be that nets are impacting migrating or breeding areas rather than juvenile habitats. The lack of published literature means that the scale of entanglement-induced mortality is unknown, as are the population level impacts of such mortality.

Impacts on nesting beaches

Nesting beaches are extremely important habitats for marine turtles and are already under pressure from issues such as sea-level rise and coastal development (Fuentes *et al.*, 2009). Sandy shorelines are thought to be sinks for marine debris whereby litter, after becoming stranded, is eventually trapped in the substrate or is blown inland (Poeta *et al.*, 2014). As such, various sizes and types of plastic accumulate on marine turtle nesting beaches (Ivar do Sul *et al.*, 2011; Turra *et al.*, 2014). Developed or remote beaches may experience similar levels of contamination but inaccessible beaches, which are not cleaned may experience greater densities of plastic pollution (Figure 2d; Özdilek *et al.*, 2006; Ivar do Sul *et al.*, 2011; Triessnig, 2012). From large fishing nets to tiny microscopic particles, this debris presents a threat to nesting females, their eggs, and emerging hatchlings (Ivar do Sul *et al.*, 2011; Triessnig, 2012; Turra *et al.*, 2014), further limiting and/or degrading the amount of habitat available for reproduction.

Female marine turtles are philopatric, returning to their natal region to lay eggs in the sand (Bowen and Karl, 2007). Large debris obstacles may impede females during the nest site selection stage, causing them to abort the nesting attempt and return to the sea without depositing eggs (Chacón-Chaverri and Eckert, 2007). Alongside this, entanglement is a risk when debris, such as netting, monofilament fishing line, and rope, is encountered (Ramos *et al.*, 2012). Additionally, macro-plastic within the sand column itself may prevent hatchlings from leaving the egg chamber, trapping them below the surface (Authors', pers. obs.).

On emergence from the nest, hatchlings must orient themselves towards the sea and enter the water as quickly as possible to avoid depredation and desiccation (Tomillo *et al.*, 2010; Triessnig, 2012). The presence of obstacles may act as a barrier to this frenzy crawl, not only trapping and killing the hatchlings but increasing their vulnerability to predators and causing them to expend greater amounts of energy (Özdilek *et al.*, 2006; Triessnig, 2012).

The physical properties of nesting beaches, particularly the permeability and temperature, are known to be altered by the presence of plastic fragments and pellets (Carson *et al.*, 2011). These authors found that adding plastic to sediment core samples significantly increased permeability, and sand containing plastics warmed more slowly, resulting in a 16% decrease in thermal diffusivity (Carson *et al.*, 2011). This, and the fact microplastics have been found up to 2 m below the surface (Turra *et al.*, 2014), indicates potential ramifications for turtle nests. Hatchling sex-ratios are temperature-dependent; consequently, eggs that are exposed to cooler temperatures produce more male hatchlings than females within the clutch (Witt *et al.*, 2010; Carson *et al.*, 2011; Vegter *et al.*, 2014). Eggs buried beneath sediment containing a high plastic load may also require a longer incubation period to develop sufficiently (Carson *et al.*, 2011). Increased permeability may result in reduced humidity which could in turn lead to desiccation of the eggs (Carson *et al.*, 2011). Other possible impacts include sediment contamination from absorbed persistent organic pollutants or leached plasticizers (Oehlmann *et al.*, 2009; Carson *et al.*, 2011; Turra *et al.*, 2014). For example, the physiological processes of normal gonad development in red-eared slider turtles (*Trachemys scripta*) at male-producing incubation temperatures were altered by PCB exposure, resulting in sex ratios that were significantly biased towards females (Matsumoto *et al.*, 2014).

Wider ecosystem impacts

Marine turtles utilize a variety of aquatic habitats that are both neritic and oceanic (Bolten, 2003), but the presence of marine plastics may reduce productivity and cause detrimental changes in ecosystem health (Richards and Beger, 2011). Here, we outline the possible impacts of plastic pollution on two key types of habitats.

Neritic foraging habitats

Coral reefs are relied upon by turtles for food, shelter from predators, and the removal of parasites by reef fish at "cleaning stations" (León and Bjorndal, 2002; Blumenthal *et al.*, 2009; Sazima *et al.*, 2010; Goatley *et al.*, 2012). Richards and Beger (2011) found a negative correlation between the level of hard coral cover and coverage of marine debris as it causes suffocation, tissue abrasion, shading, sediment accumulation, and smothering; all of which may lead to coral mortality (Matsuoka *et al.*, 2005; Brown and Macfadyen, 2007; Richards and Beger, 2011). Additionally, high densities of marine debris appear to impact both the diversity and functioning of coral reef communities, which may lead to a further reduction in biodiversity (Matsuoka *et al.*, 2005; Richards and Beger, 2011). Furthermore, scleractinian corals have been shown to ingest and assimilate microplastics within their tissues, suggesting that high microplastic concentrations could impair the health of coral reefs (Hall *et al.*, 2015). For turtles, changes to these assemblages may lead to a reduced availability of food, a greater predation risk, and an increase in epibiotic loads, such as barnacles (Sazima *et al.*, 2010).

Sea grass beds and macroalgae communities are important foraging habitats for the herbivorous green turtle but are sensitive to habitat alterations; the impacts of which are often observed in the form of reduced species richness (Santos *et al.*, 2011). As highly competitive species become dominant, some marine herbivores are forced to consume less-preferred algal species which in turn reduces the dietary complexity of those organisms (Santos *et al.*, 2011). Balazs (1985) found that this resulted in reduced growth rates of juvenile turtles.

Oceanic fronts

As previously discussed, features such as mesoscale thermal fronts and smaller coastal eddies act as foraging hotspots for many marine organisms and are an important micro-habitat for pelagic or surface feeding coastal turtles (Scales *et al.*, 2014, 2015). However, these features are likely sink areas for both macro and microplastics which degrade the quality of these critical habitats, not only in terms of increasing the risk of direct harm through ingestion and entanglement, but by indirectly altering the abundance and quality of the food available (González Carman *et al.*, 2014). Small particles of plastic are known to affect the reproduction and growth rates of low trophic level organisms, for example, zooplankton (Cole *et al.*, 2013). Finally, there is a possibility that the accumulation of such plastic debris can inhibit the gas exchange within the water column, resulting in hypoxia or anoxia in the benthos, which in turn can interfere with normal ecosystem functioning and alter the biodiversity of the seabed (Derraik, 2002).

Future research

There are many worthy lines of investigation that would further aid our understanding of the expanding issue of marine plastic pollution and its impact on marine turtles. These are discussed below and summarized in Table 3.

Ingestion

Given the variability in the scale and extent of plastic pollution within the marine environment, there is a clear need to improve our knowledge of relative risk. To achieve this, we advocate for further research to better understand the species, populations, and size classes that have either high likelihood of exposure or high consequences of ingestion. There are a number of biases that need to be eliminated in our knowledge base.

Geographic

Studies from the Atlantic are as many as those from all other oceans combined. There clearly needs to be much further work from the Indo-pacific.

Species

Although the relative distribution of studies in some way maps to the overall research effort across species, there clearly needs to be more work on species other than loggerhead and green turtles. Of particular interest are hawksbill, leatherback, and olive ridley turtles, given their cosmopolitan distribution and the largely oceanic nature of the latter two species. For Kemp's ridleys and flatbacks, despite their limited geographic range, there is clearly room for a better understanding of this problem, especially given the conservation status of the former.

Life stage

It is suggested that young turtles residing in or transiting convergence zones, where high densities of plastics are known to occur, are at greater risk from ingesting plastic debris. As such, these areas could act as a population sink (Witherington, 2002; Witherington *et al.*, 2012; González Carman *et al.*, 2014). As the development and survivorship of young turtles is critical for species persistence, it must be emphasized to generate greater understanding of the impacts of plastics for this life stage and therefore future population viability. Further sampling of frontal zones and knowledge concerning the oceanic developmental stage or "lost years" is also needed. Particularly as the detectability of mortality rates in these post-hatchling turtles is likely to be low (Witherington, 2002; Witherington *et al.*, 2012).

We found only one study that compared ingestion between the sexes, the results of which showed that the frequency of occurrence of debris ingestion was significantly higher in females. Further studies are needed to investigate whether this pattern is observed elsewhere and if so, whether this sex-based difference in plastic ingestion is biologically significant (Bjorndal *et al.*, 1994).

In terms of practical methods for identifying temporal and spatial patterns of plastic ingestion by turtles, Schuyler *et al.* (2014) found necropsy to be the most effective method. Its application, however, is constrained by small sample sizes because data collection is limited to dead animals. Therefore, every opportunity to examine by-caught and stranded individuals should be utilized (Bjorndal *et al.*, 1994). Alongside gut contents from necropsied turtles, faecal and lavage samples from live specimens should also be analysed. Although not currently a commonly used practise, this may

Table 3. Summary of recommended research priorities.

Topic	Methods
Ingestion	Experiments and field-based studies to investigate selectivity (by size, polymer type, colour) and cues leading to ingestion
	Targeted efforts to necropsy more widely to address demonstrated geographic, species, life stage, sex, and negative-results biases. Incorporate body condition indices. This would be facilitated by a global database
	Analyse faecal and lavage samples from live specimens with targeted efforts to sample pelagic life stages
	Compare data for differences in frequency, amount, type, shape, colour of plastic. Use standardized methods to catalogue debris for comparable results
	Create risk maps by assessing exposure to and consequences of ingestion, *i.e.*, utilizing satellite tracking, oceanographic and niche modelling in combination with empirical data, *i.e.*, from necropsies for ground-truthing
	Understand distribution of plastic by size and type in the water column and benthic habitats and develop three-dimensional oceanographic models to understand transport and sink areas for microplastics
	In situ investigation of plastic passage time and breakdown in turtle gut
	Health studies focusing on short- and long-term impacts of plastic debris ingestion
	Investigate role as secondary consumers including dietary analysis using molecular and isotope techniques. Sample wild invertebrate prey species for the presence of microplastics. Mesocosm experiments in a controlled laboratory setting
	Further investigation of potential for plastic consumption to lead to secondary contamination and methods to detect exposure
	Develop methods for the quantification of microplastics in turtle gut content
	Develop risk frameworks for species and populations, including detection of vulnerable life stages
Entanglement	Develop a global online database that records incidents of exposure according to entanglement, debris type, species, and life stage
	Increase reports and understanding of entanglement in plastic debris from land-based sources
	Creating risk maps utilizing satellite tracking, oceanographic and niche modelling, and data from fisheries layers such as VMS. Ground-truthing and investigation of consequences using empirical data, *i.e.*, necropsies
	On encountering debris, record the presence/absence and decomposition state of any entangled turtles
	For live strandings, gather information on health status and post-release mortality
Impacts on nesting beaches	Record observations of encounters with beach debris for females and hatchlings
	Establish baseline surveys for occurrence of plastic debris on beaches with global online database
	Sample sand-cores to investigate subsurface plastic distributions/densities
	Investigate effects on eggs and hatchlings (*e.g.*, sex ratios, embryo development, and fitness)
	Use oceanographic modelling to forecast how and when key coastal areas are likely to be impacted by plastic pollution
Ecosystem effects	Monitor key turtle habitats to generate baseline data. Mesocosm experiments. Collaborate with other research disciplines and industries
	Develop methods to detect and quantify trophic transfer of plastic, associated toxins, and bioaccumulation
	Explore the impact of plastics on the process of bentho-pelagic coupling

offer insights into survival, partial or total digestion, and comparisons with dead turtles with plastic loads (Witherington, 2002; Hoarau *et al.*, 2014). Integrating body condition indices into necropsy practices will generate a better understanding of the sublethal impacts of plastic ingestion, such as malnutrition and the absorption of toxins (Bjorndal *et al.*, 1994; Gregory, 2009; Labrada-Martagón *et al.*, 2010). It may also be useful to record conditions such as the presence of fibropapillomatosis or epibiotic loads (such as barnacles) as they are also often used as indicators of health (Aguirre and Lutz, 2004; Stamper *et al.*, 2005).

When surveying the literature on plastic debris and marine turtles, it must be emphasized to recognize that published studies do not necessarily represent a randomized sample of the rates of interactions between marine turtles and plastic debris. It is unlikely that researchers who find no evidence of plastic in their study (either in habitats or during necropsies) report negative findings—we found only two studies that did so (Flint *et al.*, 2010; Reinhold, 2015). Data on the absence of marine turtle interactions with plastic debris form an important complement to other datasets, and will facilitate a better understanding of spatio-temporal trends in rates of interactions. We strongly encourage researchers to publish both positive and negative results related to plastics and marine turtles.

We suggest that the endeavours above would be greatly facilitated by a global open access database of necropsy results with regard to plastics. At its simplest, this would be date, location, species, size, state of decomposition, likely cause of death, and some basic descriptors of presence or absence of plastic ingestion or entanglement with associated metadata. This way, workers with a single or small number of cases could still contribute to the global endeavour. Currently, seaturtle.org hosts a Sea Turtle Rehabilitation and Necropsy Database, STRAND, which allows users to upload gross necropsy reports.

To complement this, it will be important to investigate the passage of plastics through the gut, their degradation, and in addition the transport and bioavailability of bioaccumulative and toxic substances (Campani *et al.*, 2013). Few studies have been conducted on the bioaccumulation and trophic transfer of microplastics. Most have focused on invertebrates in controlled laboratory experiments and none focus on the higher trophic level organisms such as marine turtles (Wright *et al.*, 2013). Future studies should sample turtle prey species for the presence of microplastics, examine trophic transfer from prey species containing microplastics, and test for the presence of the contaminants associated with these particles in tissues of necropsied turtles.

To ensure data are comparable, the measurements used to quantify plastic abundance should be standardized. Currently, a variety of metrics are employed, making comparisons among studies difficult. The most common approach is to record total numbers and/or size of fragments. There is a possibility, however, that plastic may break down within the gut or become compressed to appear smaller. Therefore, it is more accurate and comparable to record the total dry weight once extracted (Schuyler *et al.*, 2012; Camedda *et al.*, 2014). Additionally, a wider, more global application of the European Marine Strategy Framework Directive (MSFD) "toolkit" for classification would allow a better comparison of the properties and types of ingested plastics. Furthermore, although not currently included in the MSFD toolkit, efforts to classify colour and/or shape would aid selectivity studies and offer insights as to whether these properties influence the levels of ingestion by turtles (Lazar and Gračan, 2011; Hoarau *et al.*, 2014). The colour and shape should then be compared with those of plastic pieces found in the environment of the species/life stage investigated. Systematic collection of photos with a scale bar could allow computer-based analytical techniques to be used to classify plastics and compare data across studies.

Debris–turtle interactions often occur in remote locations, far from human habitation and the chronic effects of plastic ingestion may present themselves long after the items were first encountered (Witherington, 2002; Ivar do Sul *et al.*, 2011; Schuyler *et al.*, 2014). The use of tracking technologies, such as satellite telemetry, has already been successfully employed to identify foraging habitats and migration corridors for all sea turtle species. Such data are now being used to develop niche models that can offer a synoptic view of the distribution of a whole segment of a population by season (Pikesley *et al.*, 2013) and can help predict where these ranges may be in the future (Pikesley *et al.*, 2014). Combining such data with plastic debris concentrations using remote sensing methods may identify threat hotspots leading to more effective conservation recommendations (Barnes *et al.*, 2009). At present, the tracking devices used on subadult and adult turtles are not yet available for hatchlings, but technological advances mean they will most likely be available soon as small turtles are now being tracked (Abecassis *et al.*, 2013; Mansfield *et al.*, 2014). In the interim, direct sampling of juveniles *in situ* with subsequent assessment of plastic loads during a period of captivity would seem a reasonable approach. Alternative methods, such as ocean circulation modelling, can be used to predict the migratory trajectories of hatchling turtles to understand their movements in the open ocean (Putman *et al.*, 2012). Additionally, such methods could also be employed to simulate marine debris dispersal. The development of sophisticated three-dimensional oceanographic models will enable substantial improvements to our understanding of debris transport and turtle movements.

The analysis of trace elements may be used to broadly infer the locations of foraging areas and deduce possible interactions with high concentrations of plastics (López-Castro *et al.*, 2013). A study by López-Castro *et al.* (2013) tentatively identified six oceanic clusters as foraging locations for Atlantic green turtles. As it stands this method needs refinement but with further development, fine-scale mapping may become feasible, offering valuable insights in terms of the spatial overlap with plastic debris distribution.

In addition to the horizontal spatial overlap between turtles and plastics, it would also be beneficial to understand the vertical distribution of quantities and sizes of plastics as this will influence the degree to which marine biodiversity is affected, particularly for those taxa who breathe air and forage near the surface (Reisser *et al.*, 2014b).

Entanglement

In a study by Wilcox *et al.* (2013), the spatial degree of threat posed by ghost net entanglement was predicted by combining physical models of oceanic drift and beach clean data with data concerning marine turtle distributions in northern Australia. This process identified high-risk areas so that recommendations for monitoring and remediation could be made (Wilcox *et al.*, 2013). This approach could be replicated on a global scale but would only be possible where such data exist. As such, a greater research effort is urgently needed (Matsuoka *et al.*, 2005). Indeed, the MSFD Technical Subgroup on Marine Litter is developing a dedicated monitoring protocol for their next report (MSFD GES Technical Subgroup on Marine Litter, 2011). Additionally, fisheries layers, such as vessel monitoring system (VMS) data, may help outline areas of high fishing pressure (Witt and Godley, 2007). To determine the amount of time debris has drifted, Jensen *et al.* (2013) suggest recording the abundance of epibionts as well as the presence and decomposition state of any entangled turtles.

It would be beneficial to test for any variation in entanglement rates among species and life stages to better understand vulnerability (Wilcox *et al.*, 2013), particularly for small or isolated populations (Jensen *et al.*, 2013). Stranding networks, where dead or alive turtles washed up on beaches are recorded, offer an opportunity to carry out research, not only in terms of debris entanglement but for other anthropogenic issues such as fisheries bycatch and ship strike (Casale *et al.*, 2010). In obvious cases of entanglement, such data can provide valuable insights into the temporal and spatial trends in mortality. However, it can be difficult for the layperson, and even experts, to confidently determine the cause of death for accurate recording (Casale *et al.*, 2010). For those turtles that strand alive, information should be gathered on health status and post-release mortality. Currently, there are indications that species, time, depth, and severity of entanglement affect the probability of post-release survival (Snoddy *et al.*, 2009).

During our literature search, we found that the majority of publications on turtle entanglement focus on the issue of ghost fishing by lost gear and few report entrapment in other forms of marine debris, for example, those originating from land-based sources ($n = 2$ of 9). Exploration into why this may be seems a pertinent next step for research. Additionally, to overcome the lack of peer-reviewed material, efforts should be made to gather and synthesize all relevant grey literature (for example, Balazs, 1984, 1985) in a manner that is suitable for peer-reviewed publication.

As per ingestion, a global open access database of entanglements (and animals discovered without entanglement) would greatly facilitate research efforts.

Impacts to nesting beach

Few studies exist whereby the extent of debris-induced mortality, or even interactions, for emerging hatchlings is investigated (Özdilek *et al.*, 2006; Triessnig, 2012). Observational monitoring programmes could be developed for the many conservation projects operating globally on turtle nesting beaches. This could also be applied to nesting adult females. Currently, most observations are anecdotal (Özdilek *et al.*, 2006; Triessnig, 2012). Standardized protocols for monitoring and data collection would help facilitate comparisons across studies and over time (Velander and Mocogni, 1999). Additionally, the establishment of a globally accessible database of marine debris surveys on nesting beaches would help facilitate an improved understanding of the impacts of plastics on sea turtles that use sandy beaches. Oceanographic modelling could be used to forecast how and when key coastal areas are likely to be impacted in the future.

To date, most studies on coastal microplastic distributions have focused on surface densities. As illustrated by Turra *et al.* (2014), this may lead to a mis-representation of their overall concentrations. To better quantify this, and develop a greater understanding of the potential impacts on marine turtles and their eggs, three-dimensional sampling should be carried out, investigating the distribution of microplastics at depth (Turra *et al.*, 2014).

Additionally, the relationship between marine plastics and hatchling sex ratios, both in terms of chemical contamination and nest environments, requires greater clarification. This is of interest due to the potential large-scale impacts on turtle populations, particularly as climate change is already predicted to significantly alter female to male ratios (Hawkes *et al.*, 2009).

Wider ecosystems effects

Due to the importance of marine habitats such as coral reefs, sea grass beds, and mesoscale thermal fronts for marine turtles, it is essential that we understand the scale of impact from marine debris. Data concerning the distribution and abundance of plastics within these key ecosystems will provide an environmental baseline, a method by which patterns, trends, and, potentially solutions, may be identified. As both coral reefs and seagrass beds are often frequented by divers, utilizing citizen science-based approaches, such as volunteer surveys, may be an affordable and effective method of collecting such data (Smith and Edgar, 2014). Offshore sampling at oceanic fronts may require greater resources but collaboration between research disciplines and industries may help to minimize duplication of effort and expense. As the presence of plastics within the marine environment is of concern not only for biodiversity conservation but for fisheries, tourism, and human health and well-being (through contamination of seafood, a commercially important resource), it is likely that research into this area will grow. As such, it would seem appropriate that those concerned should cooperate to tackle the issue, sharing data where possible.

To better understand the ecosystem level effects of marine plastics, micro- and mesocosm experiments are useful methods of replicating natural environmental systems in controlled conditions (Benton *et al.*, 2007). So far, the majority of such studies have looked only at single taxa, but these study systems allow for investigation into how the links between different marine environments may be affected. As such, further studies should focus on bentho-pelagic coupling to explore the impacts of plastics on the relationships themselves, providing an indication of what influences this foreign debris may have on ecosystem functioning.

Conclusion

Currently, there is little clear evidence to demonstrate that interactions with plastics cause population level impacts for marine turtles. This, however, should not be interpreted as a lack of effect (Gall and Thompson, 2015). Their widespread distribution, complicated spatial ecology, and highly mobile lifestyles make studying turtles difficult and the development of monitoring programmes that deliver statistically robust results challenging. This coupled with the diffuse nature of marine plastic pollution further exacerbates the difficulty in identifying a direct causal link to any potential impacts. In this review, we have demonstrated the widespread and diverse pathways by which plastics may affect turtles. These include ingestion, both directly and indirectly; entanglement; alterations to nesting beach properties; wider ecosystem effects. Although it is evident that this issue could have far-reaching ramifications for marine biodiversity, the lack of focused scientific research into this topic is a major hindrance to its resolution. Policy-makers require robust, comparable, scale-appropriate data (including negative results) on which to develop appropriate and effective mitigation recommendations, something which, as it stands, are severely lacking (Brown and Macfadyen, 2007). We encourage open reporting of plastic–turtle interactions and urge such observations to be submitted for peer-reviewed publication where ever possible. Furthermore, cooperation among scientists, industry, governments, and the general public is urgently needed to confront this rapidly increasing form of pollution.

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