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Plastic ingestion by fish in the Southern Hemisphere: A baseline study and review of methods

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ABSTRACT

Plastic ingestion is well documented among marine birds and sea turtles but fewer studies have investigated ingestion in fish, particularly in the Southern Hemisphere. We investigated the frequency of plastic ingestion in 21 species of fish and one species of cephalopod. The overall occurrence of plastic ingestion was 0.3%. Two micro-plastic items were recovered from the gastrointestinal tract of a single Antarctic toothfish (*Dissostichus mawsoni*). Ingestion rates were similar to other studies of fish conducted in both the Northern and Southern Hemispheres, however comparisons across species and locations are challenging due to the lack of consistency in the identification and classification of plastic debris. In response, we propose a standardised sampling protocol based on the available literature to provide a stronger basis for comparisons among existing and future studies of plastic ingestion in fish.

1. Introduction

Plastic pollution is widespread throughout the world's marine environments (Eriksen et al., 2014; Thompson et al., 2004). Current production, use, and disposal of plastic materials is not sustainable and presents significant concerns in terms of its introduction and subsequent accumulation in the global oceans (Thompson et al., 2009). Marine plastic debris originates from land and sea, entering the ocean as a result of both deliberate and accidental actions. Research suggests there are five trillion plastic items, weighing more than 243,978 million metric tonnes (MT), currently floating at the ocean's surface (Eriksen et al., 2014; Jambeck et al., 2015). Once present in the marine environment, plastic items are dispersed via oceanic currents and wind patterns, resulting in their global manifestation which extends throughout the water column (Barnes et al., 2009; Lebreton et al., 2012). In addition to this ubiquitous distribution, there are regions where debris is known to accumulate in substantial concentrations, most notable are the five oceanic gyres located in each of the major ocean basins (Eriksen et al., 2014). Of great concern is that these same regions often exhibit increased abundance of wildlife due to associated upwelling processes and biological productivity (Jantz et al., 2013).

Plastic debris presents a significant threat to marine biota (Gall and Thompson, 2015; Vegter et al., 2014). Negative encounters between wildlife and marine plastic pollution have increased from 267 species in 1997 (Laist, 1997) to 693 species in 2015 (Gall and Thompson, 2015), demonstrating an increase of nearly 75% in less than two decades. Major threats to marine life are from entanglement, or the direct and sub-lethal effects of ingestion, exposing wildlife to pollutants absorbed to the surface of plastic particles (Chua et al., 2014; Lavers and Bond, in press; Lavers et al., 2014; Tanaka et al., 2013). Throughout the water column, plastic objects exist in a variety of colours, shapes, sizes and densities (Reisser et al., 2014). These items degrade slowly in the marine environment, persisting for long periods of time, and are subsequently available for entry into the marine food web via ingestion by zooplankton (Cole et al., 2013), invertebrates (Graham and Thompson, 2009), fish (Davison and Asch, 2011), sea turtles (Di Beneditto and Awabdi, 2014), birds (Lavers et al., 2014) and marine mammals (Gall and Thompson, 2015).

Once ingested, plastic debris can contribute to a wide range of impacts including internal blockages and disrupted digestion (Hjelmeland et al., 1988; Jackson et al., 2000), biomagnification of harmful chemicals associated with plastics up the food web (Farrell and Nelson, 2013; Teuten et al., 2009), and a growing list of sub-lethal effects including morbidity (Lavers et al., 2014), liver toxicity (Rochman et al., 2013), endocrine disruption (Rochman et al., 2014) and neurotoxic effects (Oliveira et al., 2013). There is conflicting evidence in the literature regarding the retention times of plastic in the stomachs and intestines of marine wildlife (Hoss and Settle, 1990; Ryan, 2015) and the ability of fish to pass plastic items through their digestive tract (Hoss and Settle, 1990; Van Noord et al., 2013).

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Ingestion of marine plastic by fish was first reported by Carpenter et al. (1972). The majority of studies since then have been undertaken during the past five years focusing almost exclusively on Northern Hemisphere species (Anastasopoulou et al., 2013; Boerger et al., 2010; Davison and Asch, 2011; Foekema et al., 2013; Jantz et al., 2013; Lusher et al., 2013; Romeo et al., 2015). Far less research has been conducted on Southern Hemisphere species (Cliff et al., 2002; Di Beneditto and Awabdi, 2014; Ramos et al., 2012) as well as in freshwater environments (Faure et al., 2012; Sanchez et al., 2014). Of the handful of plastic ingestion studies conducted in the Southern Hemisphere (Appendix 1), none have investigated fish in Australian waters.

Recent estimates suggest plastic pollution is present in substantial quantities in Southern Hemisphere marine environments (Eriksen et al., 2014) and is therefore available for ingestion by fish and other species. Exceptionally high rates of plastic ingestion by seabirds foraging in the Tasman Sea off eastern Australia (Lavers et al., 2014), along with sporadic reports from southern opah (*Lampris immaculatus*) in the Southern Ocean (Jackson et al., 2000), and tiger sharks (*Galeocerdo cuvier*) in northern Australia (Stevens and McLoughlin, 1991) provide further evidence of plastic in Australian waters. However, observations of ingestion were incidental to the primary aims of most of these studies and as a result, the frequency of plastic ingestion and its associated impacts on Australian fish remains largely unknown.

One of the challenges to assessing the impacts of plastic in marine environments is the lack of standardised methodologies used across studies, making comparisons among them problematic. For example, there are inconsistencies in the size classes used to describe plastic items. While mega- (>100 mm), macro- (>20 mm), meso- (>5 mm) and micro-plastic (<5 mm) classifications are generally accepted in most studies (Barnes et al., 2009; Romeo et al., 2015; Ryan et al., 2009; Sanchez et al., 2014), others have used varying size categories (Dantas et al., 2012; Eriksen et al., 2014; Romeo et al., 2015).

The primary aims of this study were twofold: to address the paucity of quantitative and qualitative information regarding plastic ingestion by fish in Australian waters by describing the frequency of occurrence, size, and types of plastic ingested; and to develop a standardised sampling protocol from the available literature that will maximise the value of data collected in future studies as well as facilitating direct comparisons among them.

2. Materials and methods

2.1. Specimen collection and analysis

Twenty-one species of fish and one species of cephalopod were collected for this study (Table 1). The majority of fish were wild-caught with a small subset obtained from Australian fish markets. Most marine specimens were sampled from southeast Australian waters, Nichol's lanternfish (*Gymnoscopelus nicholsi*) were sourced from the Southern Ocean (Fig. 1), as were Antarctic toothfish (*Dissostichus mawsoni*), which were recovered from illegal gillnets deployed off the Banzare Banks (Fig. 1). The freshwater species Shannon galaxias (*Paragalaxias dissimilis*) was collected from the Great Lake in the central northern region of Tasmania. Fish were primarily caught during 2010–2015, however *G. nicholsi* specimens were sampled during the 1990 Australian Antarctic Division's KHIPPER cruise. Freshly caught fish were frozen after capture, while samples of *G. nicholsi* and *P. dissimilis* were preserved in 10% formalin and 70% ethanol, respectively.

To ensure a sterile working environment free of plastic contamination, all laboratory surfaces and equipment were cleaned using 100% ethanol and then visually inspected for the presence of plastic fragments. Necroscopies were undertaken in a laminar flow cabinet to prevent airborne contamination. The majority of fish were whole, allowing measurements of total length (TL), a straight line measure (not measured over the curve of the body) from the tip of the snout to the longest lobe of the caudal fin (cm), body weight (g), girth (maximum length between the ventral and dorsal sides; cm), sex (male, female or immature, where determinable) and general body condition, determined by the presence of physical injury and/or parasites. *Platycephalus bassensis* were provided to the project as stomachs only and associated biological data were unavailable. Inspection of the intestinal contents could not be undertaken for species where only stomachs or stomach contents were provided, including *P. bassensis* and *Conger verreauxi*.

Where intact fish were available, the entire gastrointestinal tracts were dissected from the tip of the oesophagus to the vent. Visual inspection was undertaken as per Di Beneditto and Awabdi (2014) to determine if any ulcerations, perforations, or obstructions were caused due to ingested plastic items. Full stomachs were weighed using an electronic balance (precision ± 0.0001 g). Contents of the digestive tract were washed into a clean petri dish and empty stomachs were re-weighed to determine content mass. Plastics were identified via visual inspection and buoyancy tests in deionised water (Hidalgo-Ruz et al., 2012). Contents were passed through a series of Tyler sieves (0.33, 1.00, and 4.75 mm) and carefully examined under a dissecting microscope to determine their likely nature (e.g., prey or plastic).

2.2. Plastic analysis

Potential plastic items, including unidentified and miscellaneous objects, removed from the gastrointestinal tracts of fish were rinsed gently to remove organic materials, dried, and weighed using an electronic balance. Each item was examined under a dissecting microscope and categorised by colour, type, degree of degradation, malleability, and provenance wherever possible. The longest and widest dimensions were recorded using vernier callipers. Items were analysed by Fourier Transform Infrared Spectrometry (FT-IR) at the University of Tasmania's Central Science Laboratory to determine polymer type. FT-IR analyses were performed using a Bruker Vertex 70 Spectrometer with a DLaTGS room temperature detector. The larger sample was run using Zinc Selenide Attenuated Total Reflectance (ZeSe ATR) at 4500–600 wavenumbers (cm^{-1}) with a resolution of four wavenumbers (cm^{-1}). Thirty-two scans were performed for the background and the sample. Microscopic samples were run with a Bruker Hyperion 3000 microscope using a 20× ATR Germanium objective and an MCT detector (liquid nitrogen cooled) at 4000–500 wavenumbers (cm^{-1}) with a four wavenumber (cm^{-1}) resolution. One-hundred and twenty-eight scans were performed for the background and samples. Spectral processing included atmospheric compensation, cutting (e.g., from 4000 to 3500 wavenumbers cm^{-1}), and an extended ATR correction. All output spectra were compared to the FT-IR Raman spectral library to determine the identities of the samples using the first derivative search function.

2.3. Development of standardised approach

A search of the available literature was performed using databases Scopus, Web of Science, and ScienceDirect. The key words used for each database search included combinations of “fish”, “plastic”, “marine debris”, “polyethylene”, “packaging”, “synthetic”, and “litter”. Information on the type of study, sampling procedure, laboratory analyses, plastic identification processes and plastic categorisation systems was extracted from each article. Each report was critically reviewed and data collated to allow for a comparative analysis and general overview of project approaches.

3. Results

3.1. Presence of plastic marine debris

A total of 342 fish from 21 species, representing 17 fish families of class Actinopterygii (ray-finned fishes) were examined (Table 1). Five cephalopod specimens were also analysed (Table 1). Of the 347 samples, plastic was present in one individual (0.3%). Two micro-plastic

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Table 1
Fish and mollusc species examined including weight and length where applicable. Percent frequency is the proportion of individuals that were found to have ingested plastic, calculated as the number of fish containing plastic, divided by the number of fish sampled and including those with empty digestive tracts. Species nomenclature follows Froese and Pauly (2014).

Species	*n*	Average weight ± SD (g)	Average total length ± SD (cm)	Ind. with plastic (% freq)
Class Cephalopoda				
Family Loliginidae				
Sepioteuthis australis, Southern calamari squid	5	236.5 ± 60.2	17.7 ± 2.0	–
Class Actinopterygii				
Family Arripidae				
Arripis species, Australian salmon	9	505.5 ± 73.7	34.6 ± 2.2	–
Family Carangidae				
Trachurus declivis, Greenback horse mackerel	23	228.0 ± 24.4	29.2 ± 2.5	–
Family Clupeidae				
Hyperlophus vittatus, Sandy sprat	10	5.2 ± 0.7	9.2 ± 0.4	–
Family Congridae				
Conger verreauxi, Conger	25	NA	NA	–
Family Engraulidae				
Engraulis australis, Australian anchovy	10	6.0 ± 0.9	10.3 ± 0.6	–
Family Galaxiidae				
Paragalaxias dissimilis, Shannon galaxias	20	0.8 ± 0.2	4.8 ± 0.3	–
Family Gempylidae				
Thyrsites atun, Snoek	2	504.1 ± 40.3	55.7 ± 1.0	–
Family Hemiramphidae				
Hyporhamphus melanochir, Southern sea garfish	66	133.2 ± 37.7	31.4 ± 3.3	–
Family Labridae				
Notolabrus tetricus, Blue-throated wrasse	7	494.7 ± 67.8	30.3 ± 1.5	–
Family Mugilidae				
Aldrichetta forsteri, Yellow-eye mullet	17	443.3 ± 61.0	37.9 ± 1.4	–
Mugil cephalus, Flathead grey mullet	43	246.5 ± 61.2	30.8 ± 2.6	–
Family Myctophidae				
Gymnoscopelus nicholsi, Nichol's lanternfish	39	22.4 ± 4.8	14.6 ± 1.0	–
Family Nototheniidae				
Dissostichus mawsoni, Antarctic toothfish	10	31400.0 ± 10823.8	145.2 ± 20.5	1 (10.0)
Family Odacidae				
Haletta semifasciata, Blue weed whiting	2	455.3 ± 25.3	37.6 ± 0.3	–
Family Platycephalidae				
Platycephalus bassensis, Sand flathead	10	NA	31.0 ± 1.0	–
Platycephalus laevigatus, Black flathead	7	256.8 ± 75.9	34.6 ± 3.5	–
Family Scombridae				
Katsuwonus pelamis, Skipjack tuna	1	1629.3	49.0	–
Scomber australasicus, Blue mackerel	4	370.8 ± 26.9	33.8 ± 0.9	–
Family Scorpaenidae				
Scorpaena jacksoniensis, Eastern red scorpionfish	19	228.7 ± 69.3	21.7 ± 2.2	–
Family Sillaginidae				
Sillaginodes punctatus, Spotted sillago	8	122.1 ± 15.9	27.8 ± 1.0	–
Sillago flindersi, Flinders' sillago	10	47.8 ± 8.2	17.8 ± 1.0	–
Total	347			1 (0.3)

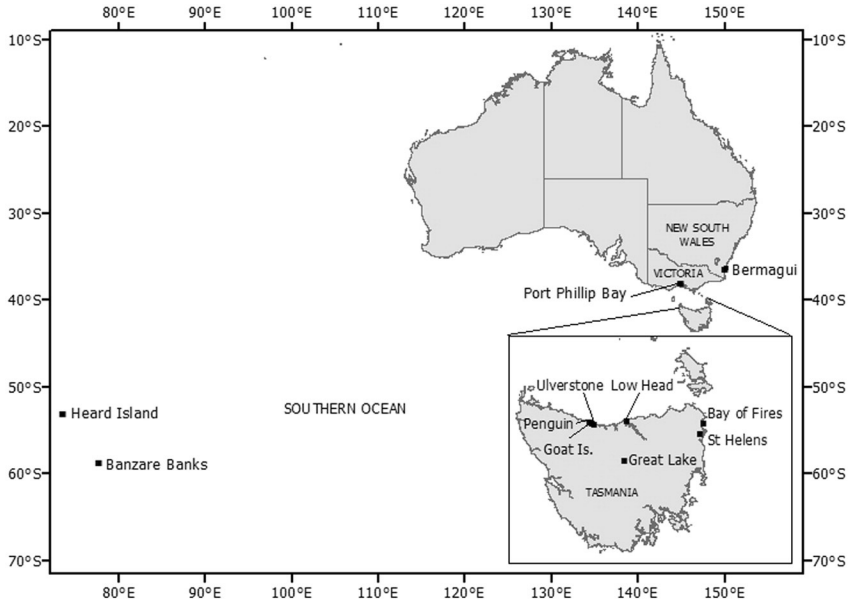

Fig. 1. Fish sampling locations for this study.

items (Fig. 2) with a total mass of 0.0001 g were found in the gastrointestinal tract of a female *D. mawsoni* (TL 152.0 cm, weight 32.0 kg). Plastic fragments were green-brown in colour, hard and brittle. FT-IR identified the samples as an acrylic resin. Provenance could not be determined. Gut contents of the *D. mawsoni* specimen also included unknown species of fish eggs and parasitic nematodes.

3.2. Method review

Thirty-four references were initially identified from the literature review for their potential usefulness in the development of a standardised sampling protocol for studying plastic ingestion in fish. Of the 34 references identified, 19 were excluded due to their lack of information regarding the number and type of items ingested (the primary aims of these studies were not plastic ingestion, so few details were recorded by the authors). The remaining 15 studies predominantly investigated debris ingestion in fish as their primary objective (see Appendix 1). Eighty percent of the literature included in the review explicitly distinguished the ingestion of plastic items from other debris materials. The remaining studies did not stipulate the recovery of plastic objects from debris, reporting ingestion of plastic with other non-plastic items.

Fish location and capture method were reported by 12 of the studies to varying degrees. Capture methods varied, but fish were primarily caught by trawling and netting using a range of different mesh sizes. Sample sizes ranged from 41 to 15,666 specimens per study, with the number of species ranging from 1 to 2741. Length, mass, age and sex were the most frequently reported characteristics, however no single parameter was reported across all studies. Infrequently reported parameters included stomach content weight (a measure of fullness) and fish girth.

A range of techniques were used to identify plastic in fish. Microscopy (53%) and the visual examination of gut contents, including all plastic items in the visible spectra (47%) were the most frequently used techniques in the reviewed literature. Other methods used to isolate plastic from gut contents included solutions or stains to degrade or highlight organic from synthetic materials, FT-IR, and sieving or filtering. Foekema et al. (2013) and Lusher et al. (2013) were the only studies to include FT-IR to categorise polymer type once the presence of plastic had been detected. Twenty-seven percent of the studies analysed the full gastrointestinal tract of the fish and 27% examined only the stomach. The remaining papers did not specify this information, however the terminology employed in these papers suggests only the stomach was examined for the majority of the studies. Contamination of plastic in the laboratory was identified in two separate studies, likely resulting in underestimated frequencies of occurrence due to the exclusion of all contaminants and potential contaminants (Davison and Asch, 2011; Foekema et al., 2013) while other studies which did not account for contamination may have reported inflated results.

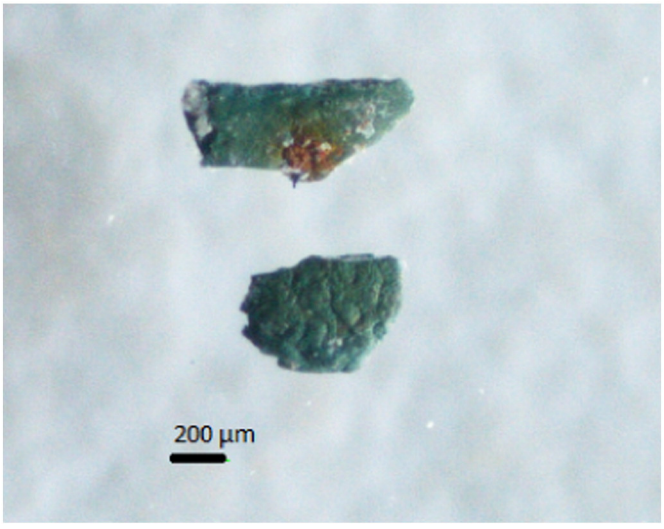

Fig. 2. Photograph (2.5× magnification) of two green-brown acrylic resin items (longest dimensions top: 845.7 µm, bottom: 583.2 µm) recovered from the gastrointestinal tract of a *Dissostichus mawsoni*. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

Categories used to describe plastics varied widely among studies. Most provided some information on the type, colour and size of items ingested, few provided information on weight, shape, degradation stage, pliability or provenance (Jantz et al., 2013; Romeo et al., 2015). Colour categories were also variable across studies, with a general lack of consistent approach to differentiating between categories such as white and clear (Boerger et al., 2010; Choy and Drazen, 2013; Di Beneditto and Awabdi, 2014) or transparent (Davison and Asch, 2011; Romeo et al., 2015). Inconsistencies were also apparent in the classifications used for the types of plastics recovered, in particular the irregular use of fibre, filament, line, net, rope, fragment, nylon and strap. These terms were used by various studies to describe what the European Union Marine Strategy Framework Directive (MSFD) define as thread or fragment (Hanke et al., 2013). Under MSFD guidelines, thread is used to describe threadlike materials, such as pieces of nylon wire and net fragments whereas fragment would be more commonly used to describe thicker type plastics (Hanke et al., 2013). Some studies reported the average size, or the size range of plastics ingested, but most did not provide detail on the mass or dimensions of individual items.

4. Discussion

The present study contributes baseline data on the ingestion of marine plastic debris by ray-finned fishes and cephalopods in Australian waters. Acrylic resin fragments were recovered from the gastrointestinal tract of a single *D. mawsoni* specimen. Overall findings suggest that bony fish sampled from the southeast Australian and Southern Ocean region are not ingesting substantial quantities of visible plastics (0.3% frequency of occurrence). Other studies have similarly recorded zero to low plastic recovery from species used in the present study and from regions in the Southern Hemisphere (Appendix 1). It is therefore likely that the low rates of ingestion exhibited in the present study are related to the reduced vessel traffic, less-populated coastlines, and consequently lower levels of marine plastic pollution available in the Southern Hemisphere (Anastasopoulou et al., 2013; Barnes et al., 2009; Eriksen et al., 2014).

The acrylic resin fragments ingested by an Antarctic toothfish, a species which inhabits depths up to 2200 m, agree with previous studies demonstrating that plastic pollution is present in even the most remote locations, such as the deep waters of the Southern Ocean (Eriksson and Burton, 2003; Eriksson et al., 2013). Unfortunately, the provenance of the plastic items could not be determined.

Studies focusing specifically on the ingestion of plastic by fish are limited, especially in the Southern Hemisphere. Consequently, identifying temporal trends and factors influencing the type or amount of plastic ingested is challenging. Current hypotheses include the inability of some marine wildlife to distinguish plastic from prey (Lavers et al., 2014), secondary ingestion via prey items (Lusher et al., 2013), feeding mode (e.g., surface seizing; Moser and Lee, 1992), and the fishes use of the water column (e.g., opportunistic feeder at the benthos; Anastasopoulou et al., 2013; Romeo et al., 2015; Sanchez et al., 2014). While the fish included in this study encompassed a diverse array of feeding modes (e.g., herbivorous (*Hyporhamphus melanochir*), carnivorous (*Scorpaena jacksoniensis*), detritophages (*Mugil cephalus*) and omnivorous (*Aldrichetta forsteri*) and included species occupying a range of different depth strata in the water column, small sample size precluded further investigation into whether feeding mode and habitat use can be used to reliably predict exposure to ingested plastic.

Guidelines suggest sampling 25–50 individuals of a given species for plastic ingestion (Hanke et al., 2013; Lusher et al., 2013), with larger

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samples required for species with variable or low ingestion rates (Lavers and Bond, in press) or for studies aimed at quantifying trends in plastic over time (Hanke et al., 2013). There are a number of possible explanations for the low frequency of plastic ingestion in the present study, including small sample size. For example, only two specimens of snoek (*Thyrsites atun*) were examined, neither contained plastic (Table 1).

Fish species and age class also likely influenced the recovery of plastic in the present study as some species of fish have been shown to be more vulnerable to ingesting plastic. In laboratory settings, juvenile mullet were found to have the highest frequency of plastic ingestion (75%) of six species examined, with some individuals containing 30 or more ingested items (Hoss and Settle, 1990). Some *M. cephalus* are able to reject plastic items, or pass items through the digestive system (Colton et al., 1974; Hoss and Settle, 1990). While fish behaviour may vary between lab and field settings, our examination of wild caught mullet suggests adults either did not ingest, or were able to pass, plastic items.

A diet study of *L. immaculatus* in the Southern Ocean recorded ingestion of a wide variety of plastic items in ten (14.5%) specimens (Jackson et al., 2000). In contrast, ingested plastic was recorded in only one specimen of *D. mawsoni* examined during our study sourced from the same area. Together, these studies suggest that ingestion rates in fish occupying similar habitats or locations can be variable, likely due to age- or species-specific colour or prey preferences as has been shown for seabirds (Lavers and Bond, 2016).

Variation in the density of plastic pollution across the ocean surface (Eriksen et al., 2014) is likely to influence ingestion in some marine species. Some fish and wildlife sampled within high plastic density "gyres" exhibit increased levels of ingested plastic (Jantz et al., 2013; Young et al., 2009). The low rates of ingestion in the present study may therefore result from the lower abundance of plastic in the environments sampled (Di Beneditto and Awabdi, 2014; Eriksen et al., 2014).

Careful monitoring of some fish species could, however, provide useful data on when, and to what extent, plastic levels in the Southern Ocean are increasing. For example, in the early 1970s, substantial quantities of polystyrene spherules were recorded in the gastrointestinal tracts of five species of fish from a region known to have high concentrations of plastic pollution (Kartar et al., 1973). In contrast, a follow-up study only a few years later found the proportion of the same species of fish containing spherules had decreased drastically, likely in response to decreased pollution in the area (Kartar et al., 1976). Other species, potentially some included in this study, may have limited capacity to function as indicators, as plastic was only recorded in a single fish examined, yet is known to be present within the collection area (Reisser et al., 2013).

A review of the literature indicates that the majority of plastic ingestion records for fish are incidental, arising from studies directed at dietary habits. As a consequence, limited information was available on the plastic ingested with most studies failing to provide details relating to the number, type, colour, or frequency of items ingested. In addition, some studies did not distinguish plastic from other 'debris' categories, such as glass, wood and metal (Choy and Drazen, 2013; Young et al., 1997). One study also reported net feeding bias (higher rates of ingestion due to plastic concentrating in the net when the fish were caught; Boerger et al., 2010). As a result of these limitations, comparing the frequency of plastic ingestion across studies is challenging.

To overcome this challenge, we recommend future studies follow MSFD guidelines (Hanke et al., 2013) and sample >50 specimens per species. As most studies report a measure of total fish length and/or mass, it is recommended that this continue. These data allow the amount of plastic ingested to be examined in relation to potential impacts on fish health (Foekema et al., 2013; Ramos et al., 2012). Particularly informative is a measure of stomach weight, to determine the proportion occupied by plastic versus organic (prey) items (Jantz et al., 2013). Parameters such as age, sex and location allow for insights into particular vulnerabilities, for example adult Brazilian mojarra (*Eugerres brasilianus*) have higher frequencies of ingested plastics than juveniles (Ramos et al., 2012).

Plastic ingestion studies rely heavily on visual analysis of the gut contents. However, the risk of under- or over-estimating ingestion frequency increases if visual analysis is not accompanied with other forensic techniques (Song et al., 2015). At a minimum these should include examination under a dissecting microscope, however we strongly recommend that FT-IR or equivalent be employed, especially for items <1 mm in size (see Appendix 2; Lusher et al., 2013). Microscopy and spectroscopy should also be incorporated to assess and reduce the risk of plastic contamination in the laboratory (e.g., use of a dissecting microscope to determine if any plastic is present on tools prior to examination). Other preventative measures should include working under a laminar flow cabinet and sterilisation of equipment and workspaces after each individual fish necropsy (Davison and Asch, 2011; Foekema et al., 2013). Finally, analysis of the complete gastrointestinal tract should be undertaken where possible. Exclusion of the intestines is likely to result in conservative estimates of plastic ingestion (Davison and Asch, 2011). Furthermore, complete analysis offers critical information regarding retention inside the gut and increases the ability to better compare frequencies of ingestion across studies.

Another major challenge in comparing fish ingestion studies is the variety of approaches used to describe recovered plastic items. So that trends and differences among species and locations can be detected, development of a standard system to classify and quantify ingested plastics is needed, while such studies are still in their infancy. A standardised approach will benefit future monitoring programmes (Hidalgo-Ruz et al., 2012) and assist in the management of plastic pollution. The European MSFD, developed to coordinate the protection of the marine environment, offers preliminary guidance on monitoring marine litter (Hanke et al., 2013). Although the protocol for litter ingestion in fish is still evolving, it is recommended that future studies should include, at minimum, an assessment of incidence (percent frequency of samples with plastic), abundance by number (average number of pieces per sample) and abundance by mass (Hanke et al., 2013). At a minimum, plastic items should be categorised according to the guidelines, including pellets, sheet, thread, foam, fragments, or other (Hanke et al., 2013). We recommend adopting the Eriksen et al. (2014) size classification system based on commonly used net and sieve sizes, including small microplastics 0.33–1.00 mm, large micro-plastics 1.01–4.75 mm, mesoplastics 4.76–200 mm and macroplastics >200 mm.

5. Conclusion

The results of this study indicate that many species of fish from the southeast Australian region are not ingesting substantial quantities of plastics, or if they are, plastic does not appear to reside in the gastrointestinal tract for long periods of time. The outcomes of this project are the first for this region and should be used as a baseline for monitoring the threat of plastics in this area and the global ocean. Plastic ingestion studies for fish are still in their infancy, especially in the Southern Hemisphere. It is therefore recommended that the standardised sampling protocol provided in Appendix 2 be employed across future studies to increase consistency and allow for more accurate comparisons across species and locations. This information will better inform management decisions regarding the health of our oceans.

Supplementary data to this article can be found online at http://dx.doi.org/10.1016/j.marpolbul.2016.03.057.

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