Seabirds as monitors of marine plastic pollution

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Declaration

This thesis reports original research that I conducted under the auspices of the FitzPatrick Institute of African Ornithology, University of Cape Town. All assistance received has been fully acknowledged. This work has not been submitted in any form for a degree at another university.

Table of contents

Abstract

Small buoyant plastic items are one of the most pervasive and abundant marine pollutants. They pose significant environmental impacts, including threatening the health of marine life through plastic ingestion, necessitating efforts to reduce plastic leakage into the sea. To evaluate the effectiveness of mitigation strategies, it is essential to understand trends in marine plastic densities, types, and sources, which requires a reliable baseline for repeated assessments. While sea-surface net trawls are commonly used to monitor trends in small floating plastics at sea, they face several challenges. Seabirds, particularly petrels and albatrosses (order Procellariiformes), offer a practical alternative to net sampling as they often ingest and retain buoyant plastics encountered while foraging at sea, making them valuable indicators of this type of plastic pollution. However, few studies have thoroughly tested their utility.

Larger species, such as albatrosses and giant petrels, typically ingest macroplastics items like bags, bottle lids, and fishery-related debris, which can often be traced back to specific sources. In contrast, smaller petrels, including storm petrels, prions, and shearwaters, tend to ingest smaller items like industrial pellets and fragments of larger plastic objects, whose sources are more challenging to identify. Due to their high propensity for ingesting plastics and their tendency to consume larger volumes, these smaller petrels may be particularly well-suited for monitoring ingested plastic loads over time.

In Chapter 2, I assess trends in litter items collected at the nests of albatrosses and giant petrels breeding on Marion Island in the southwestern Indian Ocean, from 1996 to 2018. Temporal variation in litter composition and amounts were compared to data on Patagonian toothfish *Dissostichus eleginoides* fishing intensity in the area. Fishery-related litter abundance peaked during industry's height, declining in the following two decades. Other litter items increased over the last decade, when the most frequently recorded identifiable litter items were drink bottle lids from Indonesia. Long-distance drift of buoyant plastic items from Southeast Asia, mainly Indonesia, is a major source of litter to the western Indian Ocean.

In Chapter 3, I assess the use of an indirect method to sample plastics ingested by seabirds by examining regurgitated Brown Skua *Catharacta antarctica* (Stercorariidae) pellets containing prey remains of petrels at Inaccessible Island in the central South Atlantic Ocean. I compare the size of plastics in skua pellets to those collected directly from seabird carcasses, to assess the validity of this method. I also compare the composition of plastics ingested within each seabird taxon to small buoyant plastics sampled with a neuston net, to understand how the ingested plastic compares with that found in the environment. I found that as a community, petrels reflected the composition of small buoyant plastics at sea, providing support for their use as biomonitors of marine plastic pollution.

In Chapter 4, I assess how plastic loads in four petrels have changed from 1987 to 2018 in roughly decadal time periods and years. More than 3 700 regurgitated Brown Skua pellets, each containing the remains of a single petrel, indicated fluctuations in plastic loads between periods and years, but no overall clear trend was evident in any species. The number and proportions of industrial pellets among ingested plastics decreased over the study period, indicating that industry initiatives to reduce pellet leakage have been at least partly successful.

In Chapter 5, I assess whether the size, mass, and polymer types of ingested plastic items have changed over the study period (1987 – 2018) to help interpret the results from Chapter 4. I found little change in the size and mass of ingested plastics since the 1980s. The ratio of polypropylene to polyethylene has increased consistently among hard fragments of user items over time. Overall, the limited change in plastic characteristics is consistent with the absence of clear trends in plastic loads over time (Chapter 4).

In Chapter 6, which also serves as my synthesis, I investigate whether plastics sampled on beaches along the southern Cape coastline of South Africa from 1984 to 2023 exhibit the same trends in composition as small buoyant plastics ingested by petrels from 1987 to 2018. The findings show minimal changes in beached hard fragment sizes, with a recent increase in industrial pellet mass due to two major spills at sea off South Africa in 2017 and 2020. Polymer ratios in hard fragments mirrored those ingested by seabirds in the South Atlantic, indicating common influencing variables. More data are needed to understand the increase in the ratios of polypropylene to polyethylene over time, and how this may influence retention rates of plastics on the sea surface.

In summary, this thesis demonstrates that sampling plastics ingested by seabirds provides a comprehensive assessment of marine litter composition and sources. Seabirds offer valuable insights into temporal trends in plastic loads and characteristics which align with variations observed in beached plastics. The lack of clear patterns in plastic loads over time suggests that initiatives to reduce the influx of plastics, and remove existing litter, may be preventing a rapid increase in the density of floating plastics at sea, despite the ongoing increase in global plastic production. However, the possible egestion of plastics by seabirds while out at sea, may also account for the lack of clear trends. More empirical data are needed to assess this, and how turnover rates of floating plastics will change under different plastic emission scenarios, to help interpret patterns in the loads and sizes of plastics in the marine environment. These insights are crucial for assessing the efficacy of mitigation strategies to reduce plastic waste leakage into the marine environment.

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Published chapters and author contributions

The following chapters are based on published manuscripts. All papers were adapted to fit the flow and format of the thesis, and some additional data analyses were included.

Chapter 2:

Perold, V., Schoombie, S., Ryan, P.G., 2020. Decadal changes in plastic litter regurgitated by albatrosses and giant petrels at sub-Antarctic Marion Island. Mar. Pollut. Bull. 159, 111471.

Author contributions: PGR, SS and VP collected plastic litter from boluses on Marion Island, with assistance from field personnel based on the island each year. VP conceptualised the study, processed all the plastics, performed the FTIR spectroscopy, analysed the data and wrote the manuscript. SS and PGR assisted with manuscript review and editing.

Chapter 3:

Perold, V., Connan, M., Suaria, G., Weideman, E.A., Dilley, B.J., Ryan, P.G. 2024. Regurgitated skua pellets containing the remains of South Atlantic seabirds can be used as biomonitors of small buoyant plastics at sea. Mar. Pollut. Bull. 203, 116400.

Author contributions: PGR, MC, and BJD collected skua pellets on Inaccessible Island in 2018. PGR processed whole skua pellets and VP processed all the ingested plastics. VP sampled and processed surface plastics collected at sea with assistance from EW, GS, BJD and MC. GS conducted FTIR spectrometry on some of the plastics collected using neuston nets. VP conceptualised the study, analysed all the data and wrote the manuscript with guidance from PGR. PGR, MC, GS, BD and EW assisted with manuscript edits.

Chapter 4:

Perold, V., Ronconi, R.A., Moloney, C.L., Dilley, B.J., Connan, M., Ryan, P.G. 2024. Little change in plastic loads in South Atlantic seabirds since the 1980s. Sci. Tot. Environ. 950, 175343.

Author contributions: PGR, RAR, CLM, BD and MC collected skua pellets on Inaccessible Island. PGR processed whole skua pellets to remove and count all plastics. VP conceptualised the study, analysed all the data, and wrote the manuscript with guidance from PGR. PGR, RAR, CLM, BD and MC assisted with manuscript review and editing.

Chapter 1

General introduction - marine plastic pollution and seabirds

Marine litter washed ashore at Inaccessible Island, Tristan da Cunha archipelago, South Atlantic Ocean, in 2018 (photo Peter Ryan).

Introduction

Marine plastic pollution is now recognized as an environmental crisis although initial reports garnered limited attention (Carpenter and Smith, 1972; Andrady, 2011; Ryan, 2015a). Impacts range from economic (e.g. cost of cleaning street and beach litter, impacts on shipping and fishing operations) to ecological (e.g. entanglement of and ingestion by marine biota, transport of organisms) and often have far-reaching implications (Gregory, 2009; Kühn et al., 2015). Instances of ingestion were documented as early as the 1950s in turtles (Cornelius, 1975), the 1960s in seabirds (Kenyon and Kridler, 1969; Rothstein, 1973; Harper and Fowler, 1987), and industrial pellets appearing in coastal waters and on beaches were noted in the early 1970s (Carpenter and Smith, 1972; Carpenter et al., 1972; Gregory, 1978). However, it wasn't until the 1980s that most impacts associated with marine litter were documented, prompting international efforts to prevent plastic entering marine ecosystem (e.g. MARPOL Annex V, www.imo.org; Ryan, 2015a). This was followed by a period of waning interest, until recent decades when the issue re-entered mainstream conversations, fuelled by reports of vast "plastic garbage patches" floating at sea (Moore et al., 2001; Law et al., 2010; van Sebille et al., 2012; Ryan, 2015a) and heightened public awareness driven by striking visual imagery and extensive media coverage of plastic litter impacts on biota (e.g. Fig. 1.1). Public interest further surged with proliferating reports of microplastics and their impacts on both human and marine life (Thompson et al., 2009; Bouwmeester et al., 2015; Galloway and Lewis, 2016), leading to a significant increase in research effort on this subject (Barboza and Gimenez, 2015; Ryan, 2015a; Provencher et al., 2020).

The ubiquitous nature of marine plastic pollution is now well documented as plastics have been recorded on the most remote oceanic islands (Ryan and Watkins, 1988; Lavers and Bond, 2017), the deepest trenches of the sea (Chiba et al., 2018), and in both Arctic and Antarctic waters, sea ice and snow (Barnes et al., 2010; Kanhai et al., 2020; Kelly et al., 2020; Aves et al., 2022; Bergmann et al., 2022). Plastics have also been ingested by a staggering diversity of marine species (Kühn and van Franeker, 2020), ranging from the tiniest zooplankton (Cole et al., 2013) to some of the largest marine megafauna (de Stephanis et al., 2013; López-Martínez et al., 2023). A comprehensive understanding of the nature of plastic pollution, the pathways into the marine environment, and its ultimate fate is essential for accurately measuring the extent of the issue and to monitor and mitigate future spread and impacts.

Fig. 1.1 Graphic images like this carcass of a Black-footed Albatross *Phoebastria nigripes* chick filled with plastics (A, photo Dan Clark/USFWS), this Olive Ridley Sea Turtle *Lepidochelys olivacea* with a plastic straw stuck in its nasal passage (B, source plasticpollutioncoalition), and this White Stork *Ciconia ciconia* entangled in a plastic bag (C, photo John Cancalosi), all heightened public awareness and concern regarding the impacts of plastic pollution, and ignited the drive to ban single-use plastics such as shopping bags, straws and disposable cutlery.

Origin, composition and growth of the plastics industry

Plastics made its debut in the early 20th century as a by-product of the fossil fuel industry. Characterised as a versatile, light weight and inexpensive material, it has a wide range of applications. It is produced by blending monomers with various chemical additives, yielding approximately 20 categories of synthetic polymers (Thompson et al., 2009). Each polymer possesses distinct properties that dictate its utility, durability, and buoyancy. The main buoyant polymers are polyethylene (PE) (both high-density [HDPE] and low-density polyethylene [LDPE]) and polypropylene (PP), while those that sink in seawater include polyethylene terephthalate (PET), polystyrene (PS [unexpanded]) and polyvinyl chloride (PVC) (Gewert et al., 2015). Industrial-scale plastic production only began in the 1950s (Ryan, 2015a; Geyer et al., 2017). From a modest base of around 2 million metric tonnes (Mt) per year, production quickly grew as the versatility, durability and convenience in disposable applications were realized and expertly marketed (Anon., 1955), to a staggering 460 Mt per year by 2019 (Fig.

1.2). An estimated 9 500 Mt of plastics had been produced by 2019, and following a business-as-usual trajectory, this is predicted to increase to 42 000 Mt by 2050 (Thompson et al., 2009; Geyer et al., 2017; OECD, 2022). Of all the waste plastic produced to date, it is estimated that only 9% has been recycled, 12% incinerated and 79% either gone to landfill, or been dumped or littered in the environment (Geyer et al., 2017), which raises serious concerns about the cumulative impacts of plastic waste accumulating in the environment (Lebreton and Andrady, 2019; Eriksen et al., 2023). An additional concern is the fate of waste exported from developed countries to poorer nations that lack the capacity to manage it effectively. For instance, Lebreton and Andrady (2019) found that Africa has the highest rate of improper waste disposal, despite its low levels of resin production compared to more developed nations, which is largely attributed to this unfair practice.

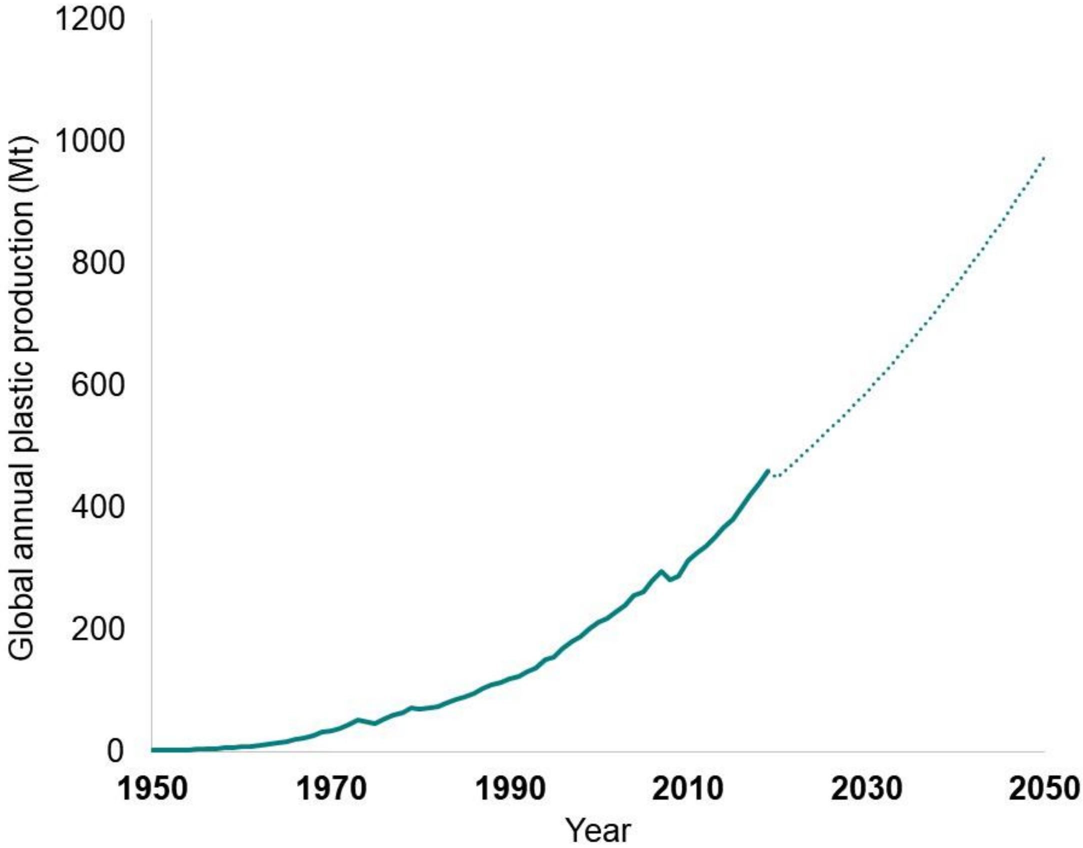


Fig. 1.2 Global annual plastic production (million metric tonnes (Mt)) from 1950 to 2019 (thick line) and estimated production from 2020 to 2050 (dashed line) following a business-as-usual trajectory (data sources: Geyer et al., 2017; OECD, 2022).

Sources and sinks of marine plastics

Marine plastics are discarded plastic litter items found in the marine environment, such as cooldrink lids, bottles, straws and user items, as well as fishing-related items such as trays, fishing line, floats, nets and ropes. Although a universally agreed upon size classification is still contested (GESAMP, 2019; Hartmann et al., 2019), the most commonly recognised size categories for plastic items are: megaplastics (> 1000 mm), macroplastics (25-1000 mm), mesoplastics (5-25 mm), microplastics (< 5 mm) and nanoplastics (< 0.001 mm; GESAMP, 2019). Once plastic enters the environment, it starts to break down into smaller fragments due to ultraviolet (UV), mechanical or microbial abrasion (Andrady, 2011; Eriksen et al., 2014; Gewert et al., 2015; Kaandorp et al., 2021). Plastics that are purposefully manufactured to be small, such as industrial plastic pellets and microbeads in cosmetic products, are referred to as primary microplastics, whereas fragments originating from larger items are referred to as secondary meso- and microplastics.

Marine plastics stem from mismanaged plastic waste, which is waste that has not been recycled, re-used or disposed of responsibly in a landfill or formal waste management facility. Marine plastics are either transported from land-based sources via littering beach-goers, wind, rivers and stormwater drains into the marine environment (Fig. 1.3), or they can also come from ship-based sources due to accidental loss at sea (e.g. during fishing operations or rough seas) or dumping at sea (Fig. 1.3) (Vauk and Schrey, 1987; Ryan, 2023a). Once buoyant mismanaged waste enters the marine environment, it often ends up in sinks such as beaches, floating in coastal waters or the open ocean, or can be ingested by marine animals (Ryan et al., 2009a; Cózar et al., 2014; Eriksen et al., 2014; Jambeck et al., 2015; Lebreton et al., 2019), while non-buoyant plastics are likely sedimented onto the seabed (Galgani et al., 1995; Galgani et al., 2000) (Fig. 1.3). Buoyant plastics that break into smaller fragments can be lost from the sea surface as a result of size-mediated biofouling (Fazey and Ryan, 2016). Biofouling occurs on the surface of floating items, and the smaller items become, the greater their surface area to volume ratio (Ryan, 2015b; Fazey and Ryan, 2016). By comparison, buoyancy is a function of volume thus as plastic items fragment they are likely to sink from the sea surface. Small plastic fragments may either sink the ocean floor, or oscillate vertically between the surface and mid-ocean layer (Ryan et al., 2009a; Reisser et al., 2015; Kooi et al., 2017; Kaandorp et al., 2021), highlighting the dynamic and complex flux of plastic litter at sea (Fig. 1.3).

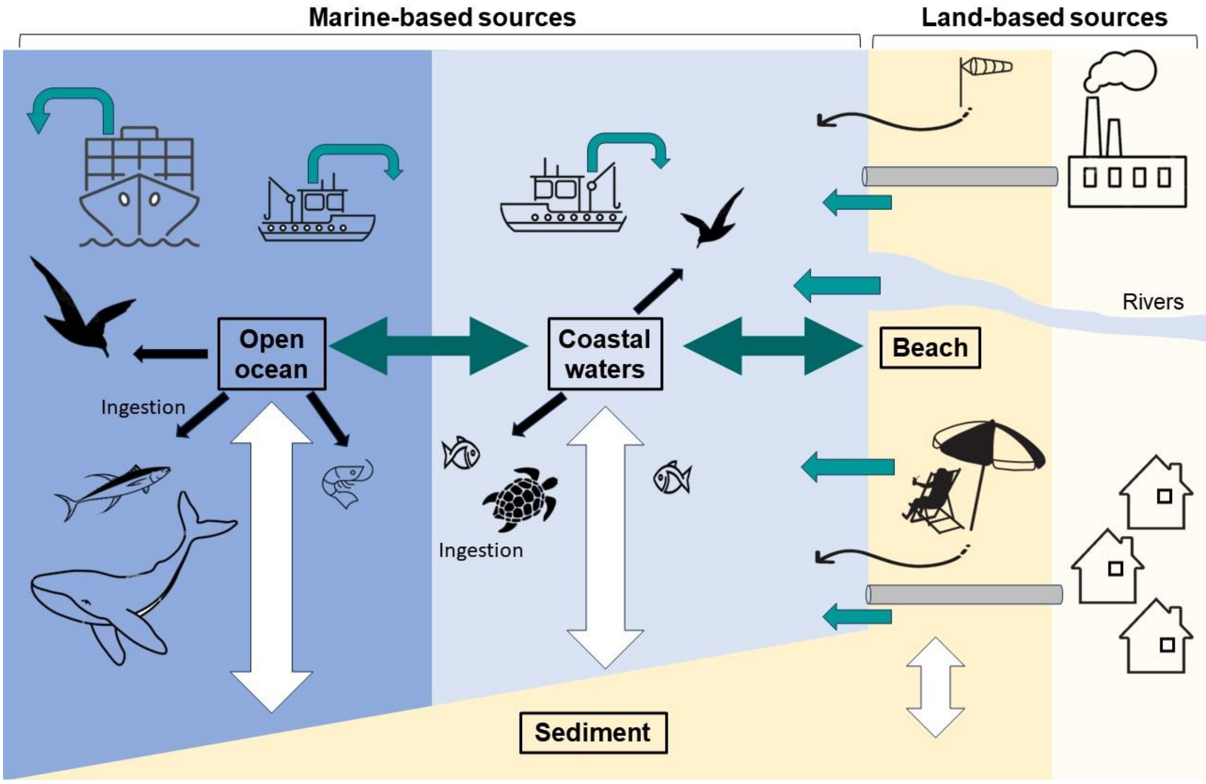


Fig. 1.3 Main sources (marine-based and land-based), sinks (open ocean, coastal waters, sediments and beaches) and movement pathways (arrows) for plastics in the marine environment. Curved black arrows depict wind-blown litter, green arrows water-borne litter, and white arrows vertical movement through the water column (including burial in sediments). Black arrows indicate ingestion by marine biota (schematic adapted from Ryan et al., 2009a).

Impacts of marine plastic pollution

Marine plastic pollution has a broad range of impacts. Economic impacts include the costs of cleaning efforts both on beaches and in streets to ensure that beachfronts, which are a significant revenue source for coastal regions, remain safe and attractive to tourists (Ballance et al., 2000; Newman et al., 2015). Remote small Island states are often faced with exuberant financial burdens to collect, remove and return litter accumulated from distant sources, with little international support (Burt et al., 2020). The shipping and fishing sectors incur losses from entangled fishing gear and damage to propellers and rudders, resulting in costly removal and operational delays (McIlgorm et al., 2011; Newman et al., 2015; Rodolfich et al., 2023). Plastic bags can also block vessels' water intakes, resulting in engine damage, which poses severe financial and safety concerns, especially in remote locations (McIlgorm et al., 2011; Newman et al., 2015). Environmental impacts include habitat destruction and structural damage to already stressed coral reefs, which exacerbate disease risk (Lamb et al., 2018;

Pinheiro et al., 2023). Macroplastic items can smother the seabed, causing anoxic conditions, reducing primary productivity and invertebrate density and diversity (Green et al., 2015; Kühn et al., 2015). Plastic debris may also serve as a vector dispersing invasive species across oceans, while also transporting pathogens that threaten human and animal health (Gregory, 2009; Rodrigues et al., 2019; Rasool et al., 2021). Entanglement of marine biota is perhaps the most visually distressing impact, affects a wide array of species and often leads to a slow, agonising death (Gall and Thompson, 2015; Ryan, 2018; Jepsen and de Bruyn, 2019; Fig. 1.1). However, plastic ingestion poses the greatest risk to marine ecosystems because of the often-widespread nature of ingestion (Ryan, 2016; Kühn and van Franeker, 2020).

Seabirds are particularly prone to plastic ingestion and of the species that have been checked, 78% contained plastics (Ryan, 2016). When plastic is ingested, the response is species and size specific and potentially affects the fitness of the animal. Larger items may be regurgitated or retained, while smaller items may be excreted (Ryan, 2016; Provencher et al., 2018). If retained, larger plastic items can block or injure the digestive tract, or create a sense of false satiation leading to starvation. They can also release adsorbed toxic compounds and additives such as colorants, plasticisers and flame retardants into the gut (Ryan et al., 1988; Gall and Thompson, 2015; Roman et al., 2020). Ultrafine particles (< 1mm) may not always be excreted as Rivers-Auty et al. (2023) found microplastics embedded in organs. As research advances, reports of other health implications for seabirds have surfaced. For example, Charlton-Howard et al. (2023) reported a plastic-induced fibrotic disease they termed “plasticosis” in Flesh-footed Shearwaters *Ardenna carneipes*, a species that frequently ingests plastic (Lavers et al., 2021). These economic and ecological impacts are a major driving force behind the movement to reduce plastic inputs and mitigate impacts, and systematic monitoring of marine plastic pollution forms the basis of these efforts.

Monitoring marine plastic pollution

Monitoring the inputs and density of marine plastics at sea started as early as the 1970s (Carpenter and Smith, 1972; Carpenter et al., 1972; Colton et al., 1974; Ryan, 1988a), and more recent estimates suggest that plastic leakage into the ocean (Jambeck et al., 2015; Lebreton et al., 2017) far outweigh estimates of the amounts floating at sea (Eriksen et al., 2014). For example, Eriksen et al. (2014) estimated that there are 5 trillion pieces of plastics weighing over 250 000 metric tonnes afloat at sea while Jambeck et al. (2015) estimated that 4.8 to 12.7 Mt enters the ocean annually from land-based sources. Eriksen et al. (2023) updated the data to 2019 and estimated the global abundance of small buoyant plastics to now be 82-358 trillion pieces weighing between 1.1 and 4.9 Mt, but these estimates still reveal a gross mismatch, and suggests a severe "loss" of floating plastics from the ocean surface (Cózar et al., 2014; Eriksen et al., 2014). This mismatch may be attributed in part to overestimates of plastic leakage from land-based sources into the sea (Ryan, 2020; Verster and Bouwman, 2020). However, it could also result in part from the challenges associated with estimating and monitoring marine plastic densities. For example, the large spatial and temporal heterogeneity in the amounts and distribution of plastic litter, as well as the intricate dynamics governing the resurfacing, recirculation, transport and sedimentation of plastics (Fig. 1.3) in the marine environment, requires large sample sizes and areas to overcome these challenges (Ryan et al., 2009a; Ryan et al., 2020a).

Marine plastics can be monitored in several ways: beach litter surveys are the most frequently used as beaches are usually accessible and inexpensive to sample. Methods used include either standing stock or accumulation surveys (Ryan et al., 2009a; GESAMP, 2019; Ryan et al., 2020a), but there are some challenges associated with the interpretation of these data. For example, debris turnover rates on beaches are still largely unknown, and episodic events such as storms may uncover buried items, further complicating the interpretation of temporal trends in litter abundance (Ryan et al., 2020a). Monitoring can also be done at source; for example monitoring waste dispersal from ships, inputs from rivers and runoff from storm water drains (Moore et al., 2011; Lebreton et al., 2017; Weideman et al., 2020). Monitoring is also often done at sea and includes visual surveys of floating macroplastics (Ryan, 2014; Ryan et al., 2014; Suaria and Aliani, 2014; Connan et al., 2021), collecting water samples (Suaria et al., 2020a) or using surface net trawls (Fig. 1.4) (Morét-Ferguson et al., 2010; Cózar et al., 2014; Suaria et al., 2020b; Suaria et al., 2023; Weideman et al., 2023). As technology advances, new methods such as using satellite-based monitoring of litter from source to sink are being developed (Cózar et al., 2024).

Over time, changes have been recorded in the density of marine plastics at sea estimated from surface net trawls, but contrary to expectations, a consistent increasing trend is not commonly found (Law et al., 2010; Law et al., 2014; van Franeker and Law, 2015; Galgani et al., 2021). Another widely applied method used to monitor marine plastics is sampling plastics ingested by marine biota (Provencher et al., 2017). This has been achieved for various species including turtles (Thibault et al., 2023) and fish (Beer et al., 2018), but the most frequently used taxa are seabirds (Ryan, 2016; Clark et al., 2023; Rodríguez et al., 2024).

Fig. 1.4 A neuston net being slowly trawled alongside the *S.A. Agulhas II* research vessel to sample floating meso- and microplastics off Gough Island, central South Atlantic Ocean in 2019 (photo Vonica Perold).

Buoyant plastics often concentrate in oceanic convergence zones such as fronts or eddies (Reisser et al., 2015). These same processes that aggregate plastics at surface convergence zones also concentrate prey of surface-feeding seabirds. As a result, seabirds are susceptible to ingesting floating plastics that are confused with food items (Bost et al., 2009; Scales et al., 2014; Clark et al., 2023). Seabirds of the order Procellariiformes exhibit particularly high plastic loads because they usually retain indigestible items in their gizzard (ventriculus) (Furness, 1985; Ryan, 1987a; Wilcox et al., 2015; Ryan, 2016; Roman et al., 2019a). The petrels generally have higher plastic loads compared to albatrosses (particularly those from the Southern Hemisphere; Furness, 1985; Ryan, 1987a; Ryan et al., 2016; Roman et al., 2019a) due to a narrow and acutely angled pyloric sphincter that traps plastics in the ventriculus (Furness, 1985; Ryan, 2015c). Consequently, these seabirds have been used as ecological indicators and to monitor temporal trends in marine plastic pollution at sea (Ryan, 1988b; Moser and Lee, 1992; Ryan, 2008; van Franker and Law, 2015; Petry and Benemann, 2017; Cartraud et al., 2019; Provencher et al., 2019; Phillips and Waluda, 2020; Lavers et al., 2021;

Rodríguez et al., 2024). For example, plastic loads in the Northern Fulmar *Fulmarus glacialis* have been adopted as an indicator of marine litter in the northeast Atlantic Ocean, where an Ecological Quality Objective for marine litter states that no more than 10% of fulmars should contain more than 0.1 g of plastic (OSPAR, 2010). van Franeker et al. (2021) proposed that this be updated to data spanning a period of at least five consecutive years where no more than 10% of at least 100 fulmars may exceed 0.1 g of plastics in their stomachs. Although several studies have proposed seabirds as bioindicators of marine plastic pollution, few have compared their findings with available environmental plastics and therefore test their utility in this regard.

Seabird body size generally determines the size of marine litter items they ingest, and therefore the taxon used to sample ingested plastics can determine their utility as biomonitors of marine plastics over time (Ryan, 1987a; Wilcox et al., 2015; Roman et al., 2019b). For example, larger procellariiformes such as albatrosses and giant petrels tend to ingest large items, that can often be assigned to source, such as plastic bags, fishing-related items such as snoods (the lines used in longline fisheries to attach hooks to longlines) and hooks, bottle lids and even plastic water bottles (Jiménez et al., 2015; Phillips and Waluda, 2020; Roman et al., 2021). In contrast, petrels and other small procellariiform seabirds, including storm petrels, prions and shearwaters (hereafter referred to as petrels), are generally limited to smaller items such as industrial pellets and secondary meso-and microplastic fragments (Ryan, 1987a; Ryan, 2008; Wilcox et al., 2015; Roman et al., 2019b; Robuck et al., 2022) that are often difficult to assign to source. Therefore, using seabird communities of varying sizes to monitor temporal and spatial variation in plastic loads and sources, provides a holistic overview of a range of litter items compared to relying on a single taxon.

Plastics ingested by seabirds can be sampled through various methods (Provencher et al., 2017; Provencher et al., 2019). Some techniques such as using emetics to sample breeding adults or chicks is invasive and can cause mortalities (Bond and Lavers, 2013). Sampling the stomachs of beached birds potentially biases towards larger plastic loads if ingestion contributes to seabird mortality (Provencher et al., 2019). However, van Franeker and Meijboom (2002) found that the age of the beach-sampled Northern Fulmars was the only variable affecting ingested plastic loads, but other studies have reported higher loads in beach-washed than road-killed birds (e.g. Rodríguez et al., 2018; Lavers et al., 2021). Beached birds may also introduce greater variation related to differences in the age and breeding status of birds sampled, making difficult to control for these factors (Rodríguez et al., 2018). Indirect approaches, such as collecting plastics from regurgitated boluses at nests or regurgitated predator pellets containing seabird prey remains have also been used (Ryan and Fraser, 1988; Ryan, 2008; Hammer et al., 2016; Philips and Waluda, 2020; Bond et al., 2021) and offer an alternative to direct sampling from seabird carcasses. In this thesis, I employ such methods to sample ingested plastics, and assess their validity in this regard. I use plastics recovered from boluses at the nests of four albatrosses and giant petrels (Fig. 1.5) breeding on Marion Island (46°52′S, 37°41′E; Fig. 1.6) sampled between 1996 and 2018, as well as regurgitated Brown skua *Catharacta antarctica* pellets containing the remains of four petrels (Fig. 1.7) that breed on Inaccessible Island (37°2′S, 12°4′W; Fig. 1.6) in the South Atlantic Ocean, sampled from 1987 to 2018. I evaluate the utility of these methods to use seabirds as monitors of marine plastic pollution by comparing the characteristics of ingested plastics to those sampled at sea within the seabirds' foraging ranges. I also assess how plastic loads, types and characteristics have changed over time across these taxa that differ in size and foraging ranges. Using a range of procellariiform seabird taxa as bioindicators of marine plastic may sample for larger and more diverse types of items than typically captured with surface nets. I use this information to interpret trends in temporal and spatial variations in reported plastic densities and composition at sea and to gauge the success of past mitigation efforts aimed at reducing plastic inputs at source. Lastly, to assess if trends in plastic composition are influenced by seabird selectivity, I compare ingested plastics to small buoyant plastics sampled on beaches along the southwestern coastline of South Africa from 1984 to the present. Consistent trends in biotic and abiotic compartments may indicate a common variable influencing these patterns.

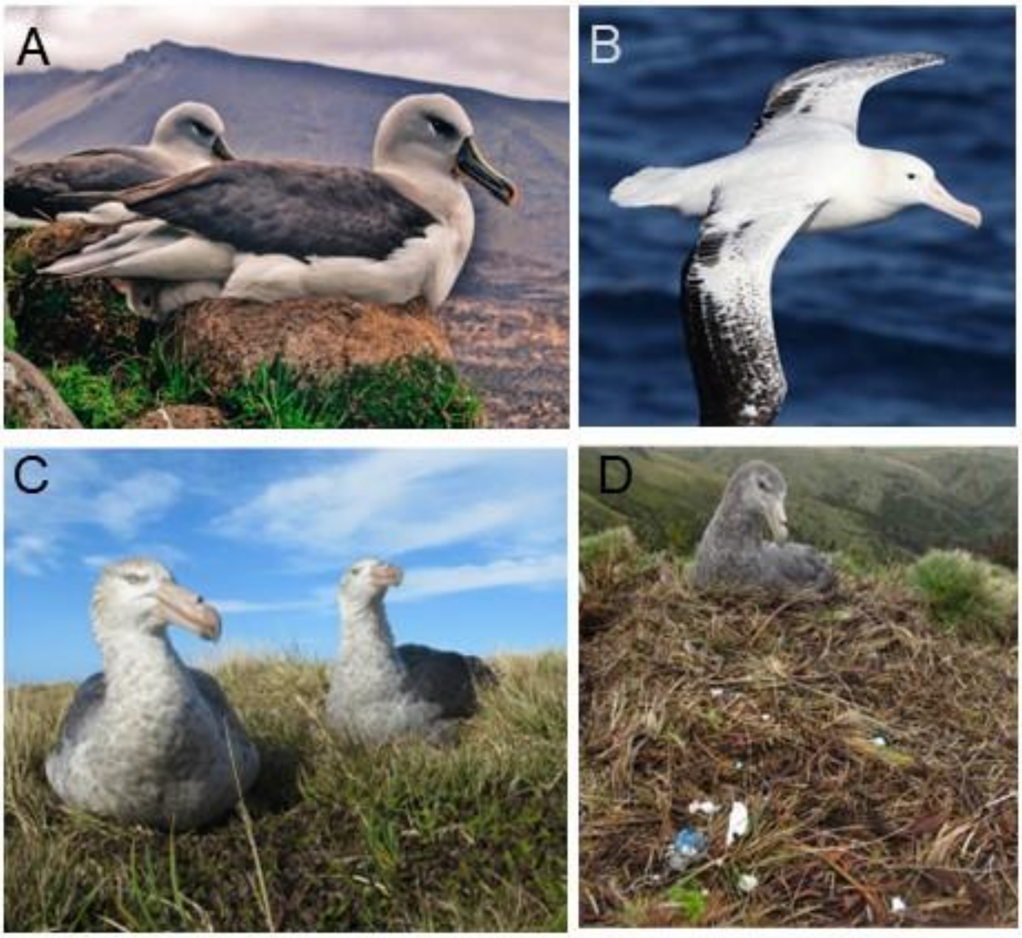


Fig. 1.5 The four large procellariiform seabird species that were used to sample plastics in the southern Indian Ocean: Grey-headed Albatrosses *Thalassarche chrysostoma* (A); Wandering Albatross *Diomedea exulans* (B), Northern Giant Petrel *Macronectes halli* pair (C) and Southern Giant Petrel *Macronectes giganteus* (D) with the remnant of a bolus, containing 124 plastic items including an Asian water bottle lid, found at the nest (photos Vonica Perold).

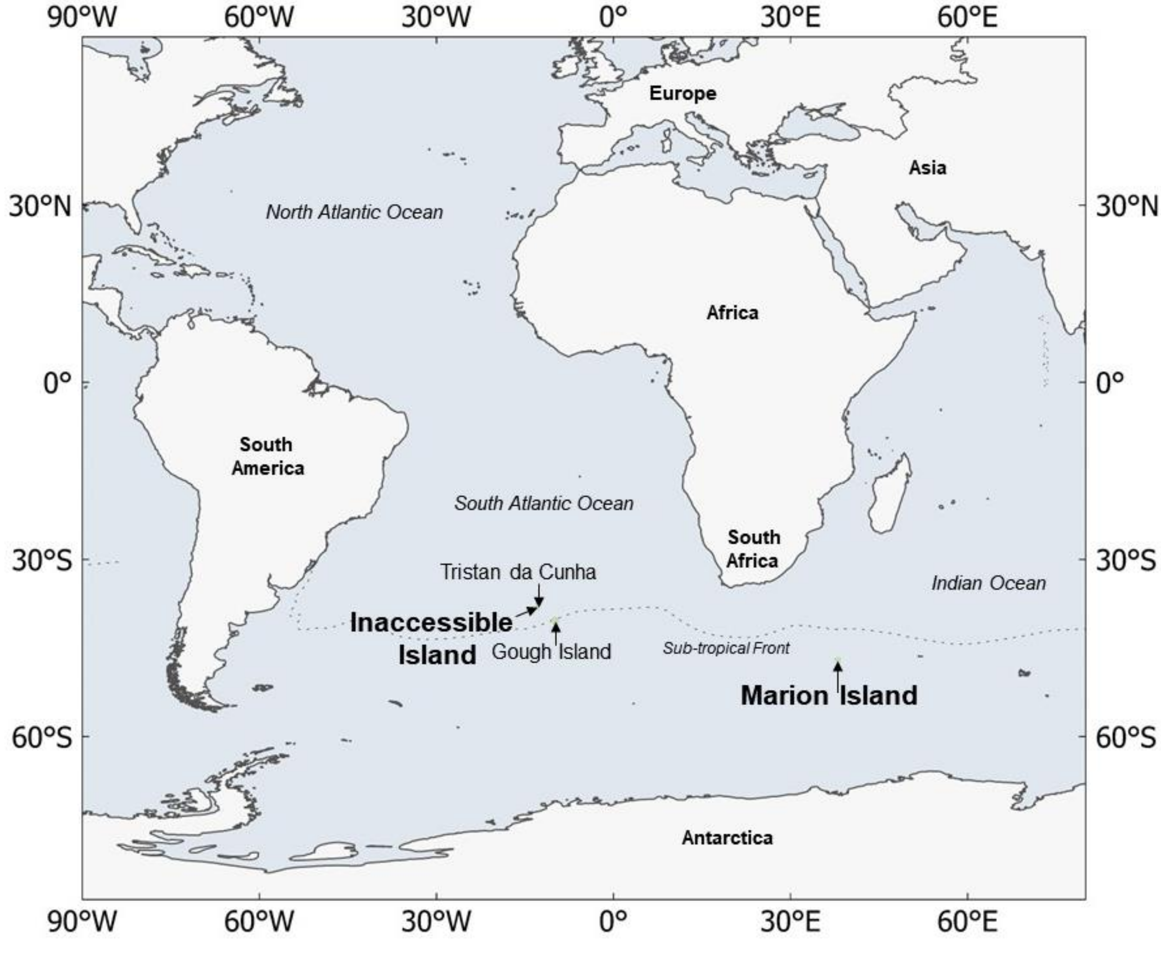


Fig. 1.6 Locations of the sampling sites: Inaccessible Island (part of the Tristan da Cunha archipelago) in the South Atlantic Ocean and Marion Island (part of the Prince Edward Island Group) in the southwestern Indian Ocean. The approximate location of the Sub-tropical Front is indicated with a dashed line.

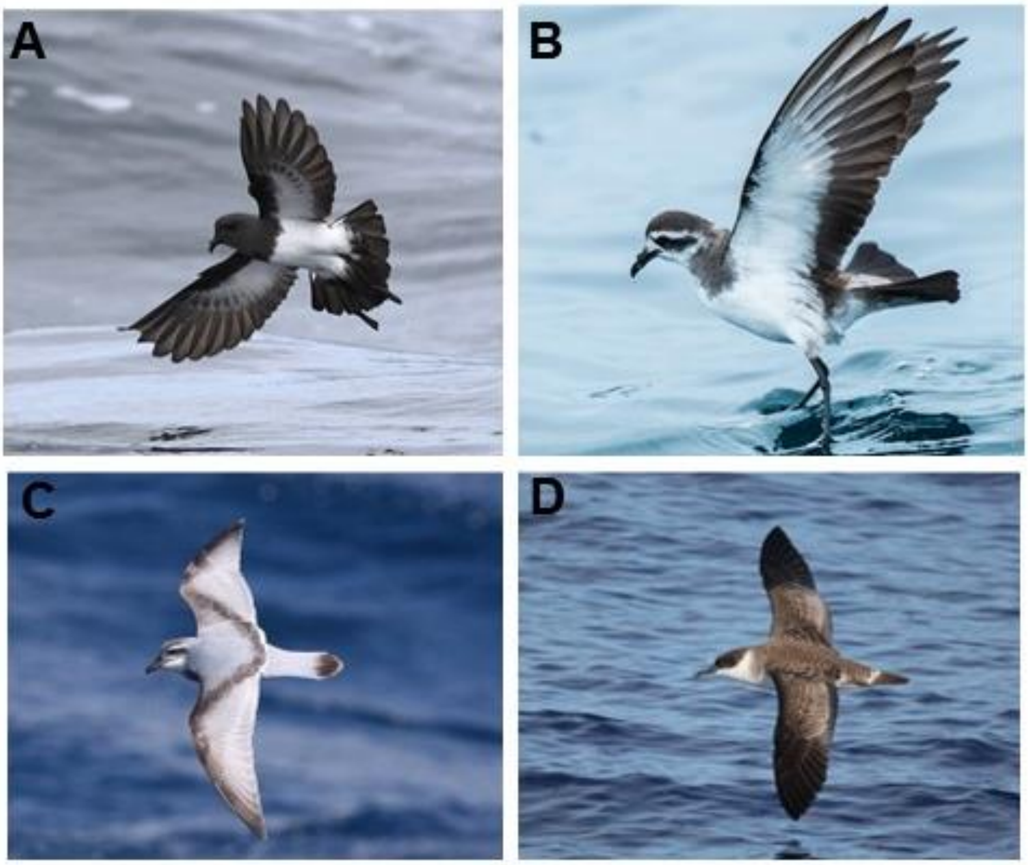


Fig. 1.7 The four petrel taxa used to sample plastics in the South Atlantic Ocean from Brown Skua prey remains: white-bellied form of the Black-bellied Storm Petrel *Fregetta tropica* (A), one of the two *Fregetta* species breeding on Inaccessible Island (photo Peter Ryan), White-faced Storm Petrel *Pelagodroma marina* (B, photo Philip Griffin), Broad-billed Prion *Pachyptila vittata* (C) and Great Shearwater *Ardenna gravis* (D, photos Peter Ryan).

Overall aims and thesis structure

The main aim of this thesis was to analyse the amounts and composition of marine plastics ingested by seabirds over time to investigate temporal and spatial variation in marine plastic pollution at sea. By quantifying the loads and characteristics of ingested plastics in both large and small procellariiform seabirds, foraging in different areas of the ocean, and relating it to what is available in the marine environment, I assess their use as monitors of temporal and spatial trends in marine plastic pollution. More specifically, the aims can be broken down into six interlinked topics (Fig 1.8).

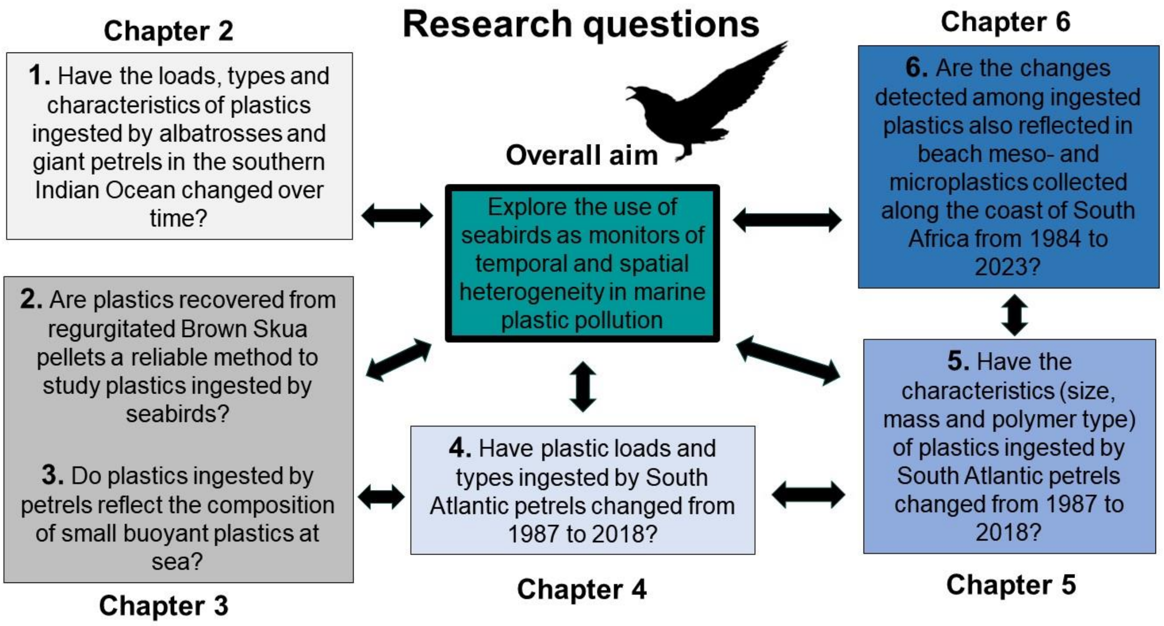


Fig. 1.8 Diagram of the linkages between the main research questions and aims.

Aims:

1) To investigate if the amounts, types and characteristics of plastics ingested by albatrosses and giant petrels breeding on Marion Island in the southern Indian Ocean have changed from 1996 to 2018.
2) To assess if using regurgitated Brown Skua pellets is a reliable method to study plastics ingested by petrels.
3) To assess if plastics from Brown Skua pellets containing the remains of petrels breeding on Inaccessible Island in the central South Atlantic Ocean reflect the composition of small buoyant plastics at sea.
4) To investigate if plastic loads and types ingested by petrels breeding on Inaccessible Island have changed from 1987 to 2018.
5) To investigate if the size, mass and polymer types, of ingested plastics in petrels breeding on Inaccessible Island have changed from 1987 to 2018.
6) To assess whether the changes detected among ingested plastics are also reflected in beach meso- and microplastics collected along the coast of South Africa from 1984 to 2023.

Thesis structure:

The main body of work of this thesis consists of four data chapters (Chapters 2–5). Chapters 2–4 were structured as stand-alone research articles in order to facilitate the publication process, and were published at the time of the final submission of the thesis. I have added a detailed section describing the roles of each co-author on page vii. I have amended the published versions for my chapters to reduce repetition of common elements in the methods and to improve the flow of the thesis.

Chapter 1: General introduction - Marine plastic pollution and seabirds

This introductory chapter provides a general overview of the history, sources, impacts and monitoring methods of marine plastic pollution over time, as well as the use of seabirds as biomonitors of plastic pollution. I set the narrative for the forthcoming chapters and introduce the main research aims.

Chapter 2: **Albatrosses and giant petrels as monitors of marine litter in the southern Indian Ocean**

In this first data chapter, I assess whether the amounts and types of plastics collected from boluses recorded at the nests of albatrosses and giant petrels breeding on Marion Island in the southwestern Indian Ocean have changed from 1996 to 2018. I assign litter items to source, and country of origin if possible, and compare the findings with historical data on fishing intensity in the area. I assess whether albatrosses and giant petrels are useful bioindicators of marine litter in the southwestern Indian Ocean, and evaluate their use as biomonitors of plastic litter over time (Aim 1).

Chapter 3: Investigating the use of skua pellets containing the remains of petrels to monitor small buoyant plastics at sea

In this second data chapter, I assess whether plastic collected from regurgitated skua pellets is a reliable tool to sample ingested plastics to track temporal changes in the characteristics of marine plastics ingested by four petrel taxa in the South Atlantic Ocean. I firstly assess if Brown Skua pellets underestimate plastic size by comparing plastics collected from petrel carcasses to those from regurgitated skua pellets in 2018. I then investigate similarities or differences in the characteristics between ingested plastic and plastic available in the marine environment, collected with surface net trawls in the South Atlantic and southwest Indian Oceans during 2016–2019, to assess the extent to which seabirds select ingested plastics (Aims 2 and 3).

Chapter 4: **Exploring temporal and spatial variation in plastic loads in South Atlantic petrels**

In my third data chapter, I analyse plastics collected from regurgitated skua pellets between 1987 and 2018, to investigate if plastic loads and types ingested by four petrel taxa (Fig. 1.7) have changed over time. I also relate my findings to temporal variation in global plastic production, the types of plastics recorded at sea and discuss this in light of historic mitigation efforts (Aim 4).

Chapter 5: Have the characteristics of plastics ingested by petrels in the South Atlantic Ocean changed since the 1980s?

In my fourth data chapter, I test whether the size, mass and polymer type of the ingested plastics in Chapter 4 have changed from 1987 to 2018 in four near-decadal time periods. I contextualise the findings in light of the dynamic flux of plastics through the marine environment, in order to untangle and comprehend any discernible trends found in Chapter 4 (Aim 5).

Chapter 6: Synthesis - A comparison of meso- and microplastic trends in seabirds and on beaches

In my last chapter, I synthesise the key findings from Chapters 2 to 5. I introduce further data which investigates temporal variation in beached litter (~2-25 mm in size) from 1984 to 2023, collected along the southern Cape coastline, South Africa, to determine if the patterns in the types and characteristics of ingested plastics continue across both biotic and abiotic compartments (Aim 6). I conclude my thesis by critically evaluating the use of seabirds as monitors of marine plastic pollution, discussing the observed trends and provide recommendations for future directions.

Chapter 2

Albatrosses and giant petrels as monitors of marine litter in the southern Indian Ocean

A Wandering Albatross *Diomedea exulans* chick on its nest at sub-Antarctic Marion Island (photo Vonica Perold).

Abstract

Plastic ingestion by seabirds is a tool used to monitor marine plastics. I report temporal variation in the characteristics of marine litter regurgitated by albatrosses and giant petrels on sub-Antarctic Marion Island between 1996–2018. Both fishery and other litter peaked during the height of the Patagonian Toothfish fishery around the island (1997–1999). Comparing the two subsequent decades of reduced fishing effort (1999–2008 and 2009–2018), fishing litter decreased while other litter increased across all species. Litter increased most in Grey-headed Albatrosses, followed by giant petrels and Wandering Albatrosses. Similar ranked responses were found in the same species at South Georgia, but non-fishery-related litter has increased faster in the Indian Ocean than in the southwest Atlantic, possibly indicating regional changes in litter growth rates. However, growing population sizes of these seabirds on Marion Island over the last decade may actually indicate a reduction in litter rates. These seabirds' regurgitations provide an easy, non-invasive way to track changes in oceanic litter rates and sources in a remote area that is otherwise difficult to monitor.

Introduction

As top predators, seabirds can be useful biomonitors of changing ecological conditions including marine plastic loads and composition (Furness and Camphuysen, 1997; Vlietstra and Parga, 2002; Hazen et al., 2019; Phillips and Waluda, 2020; Lavers and Bond, 2021; Rodríguez et al., 2024). Petrels and storm petrels are particularly useful bioindicators for studying temporal trends in plastic ingestion in seabirds (e.g. Ryan, 1987a; Ryan, 2008; van Franeker and Law, 2015; Youngren et al., 2018). They have longer retention times than many other seabird species due to the angled isthmus juncture between the proventriculus and ventriculus, preventing the bird from regurgitating the ventricular contents (Furness, 1985; Chapter 1). Plastic items, along with the hard indigestible prey remains like cephalopod beaks therefore accumulate in the ventriculus until they are broken down into pieces small enough to pass into the intestine (Ryan, 2015c). Albatrosses and giant petrels have a much less constricted isthmus juncture, allowing them to spontaneously regurgitate indigestible matter such as fish bones, cephalopod beaks and plastics in a bolus, which can be found within their breeding colonies usually close to their nests (Ryan, 1987a; Ryan, 1988b; Nel and Nel, 1999; Phillips et al., 2010; Ryan, 2015c). Ingested litter items can also be transferred from adults to chicks during feeding and egested by the chick in a bolus with other indigestible prey remains, usually shortly before fledging (Young et al., 2009; Phillips and Waluda, 2020).

Albatrosses and giant petrels are large procellariiform seabirds (e.g. Wandering Albatross *Diomedea exulans* = 7-10 kg, Grey-headed Albatross *Thalassarche chrysostoma* = 3-4 kg; Northern Giant Petrel *Macronectes halli* = 3-6 kg; and Southern Giant Petrel *Macronectes giganteus*= 2-6 kg; Ryan, 2023b) and are capable of ingesting large litter items (e.g. plastic bottle lids, fishing snoods and hooks) that can sometimes be assigned to source (Phillips et al., 2010; Phillips and Waluda, 2020), thus aiding in interpreting the origins of marine plastics (Chapter 1). Collecting regurgitated plastic and other litter from boluses is a non-invasive method to obtain data on marine litter from remote oceanic areas (O'Hanlon et al., 2017) and has been used in a variety of seabird species, including albatrosses and giant petrels (Ryan, 1987a; Nel and Nel, 1999; Young et al., 2009; Phillips and Waluda, 2020), shearwaters (Bond et al., 2021), boobies (Tavares et al., 2016), skuas (Ryan, 2008; Hammer et al., 2016), gulls (Witteveen et al., 2017) and terns (Hays and Cormons, 1974). Routine collection since 1993/94 of man-made litter regurgitated at albatross and giant petrel nests on Bird Island, South Georgia has revealed interesting patterns in the types, amounts and sources of litter regurgitated by different seabirds (Huin and Croxall, 1996; Phillips et al., 2010; Phillips and Waluda, 2020). Huin and Croxall (1996) found a six-fold increase in the incidence of fishery-related litter associated with albatrosses and giant petrels from 1992/93 to 1993/94, linked to a doubling in local fishing effort. Phillips and Waluda (2020) reported on the type and abundance of marine litter (excluding fishery-related items) recorded between 1994–2019 and reported that Wandering Albatrosses and giant petrels mostly ingested food-packaging while Grey-headed Albatrosses mostly ingested rigid plastic items. They also found long-term increases in the incidence of marine litter at Grey-headed and Black-browed Albatross *Thalassarche melanophris* nests over the 26-year period (Phillips and Waluda, 2020).

Sub-Antarctic Marion Island hosts a similar suite of breeding seabirds as South Georgia, and regurgitated plastics have been found at the nests of these species since the 1980s (Ryan, 1987a). However, routine collection of nest pollutants only started in 1996, when the amount of litter increased substantially, associated with the start of a largely unsanctioned long-line fishery for Patagonian Toothfish *Dissostichus eleginoides* around the islands (Nel and Nel, 1999). Nel and Nel (1999) found that the incidence of fishing gear and other litter increased greatly from that recorded in 1996/97 to 1997/98. The following year (1998/99) delivered even higher rates of litter items, with a strong correlation between fishing effort and number of fishery-related items recovered (Fig. 2.5). Fishing effort for toothfish started to decrease in 1999, as catch rates declined, and by 2010 only two toothfish longline vessels were licensed to fish in the waters around the Prince Edward Islands (Ryan et al., 1997; Nel et al., 2000; Nel et al., 2002a; Nel et al., 2002b; Tuck et al., 2003; CCAMLR, 2017).

In this chapter, I address the first aim of my thesis, which is to investigate if the amounts, types and characteristics of plastics ingested by albatrosses and giant petrels breeding on Marion Island in the southwestern Indian Ocean have changed from 1996 to 2018. I characterise plastic and other litter items found in regurgitated boluses in or around the nests of albatrosses and giant petrels during routine monitoring on sub-Antarctic Marion Island. I compare the types of pollutants, their colour, polymer composition, potential sources, size and mass among species recorded in 1996/97 (start of toothfish fishing around Marion Island), 1997/98–1998/99 (peak of toothfish fishing around Marion Island) and two decadal periods thereafter (1999/2000–2008/09 and 2009/10–2018/19) when there was a marked decrease in fishing effort. I expected a reduction in the amount of fishery related items over the decadal periods, resulting from the decrease in fishing activity around the Prince Edward Islands. I also expected an increase in the frequency of other plastic litter items, driven by the steady increase in global plastic production (Geyer et al., 2017) from 1950s to present (Fig. 1.2). Finally, I compare the trends at Marion Island to those observed at South Georgia to interpret how regional differences in the amounts of plastics ingested by albatrosses and giant petrels are changing over time.

Methods

Study area and data collection

Sub-Antarctic Marion Island (290 km^2) is the larger of the two Prince Edward Islands. It lies in the southwestern Indian Ocean (Fig. 1.6) between the Sub-tropical- and Antarctic Polar Front (Lutjeharms and Ansorge, 2008). Since annexation by South Africa in 1947, a small team of researchers has been based on the island each year. Since the 1980s, populations of Wandering Albatrosses, Grey-headed Albatrosses, and Northern and Southern Giant Petrels have been studied each year (Ryan et al., 2009b; Fig. 1.5). All four species are censused annually during island-wide incubation and fledgling counts. The number of nests ranged around ~1 600 for Wandering Albatrosses, ~6 000 for Grey-headed Albatrosses, ~1 500 for Southern Giant Petrels and ~350 for Northern Giant Petrels, but recent population estimates show an increase in all species over the last decade (Ryan and Connan, unpubl. data; Stevens et al., in press). Grey-headed Albatrosses and Southern Giant Petrel pairs breed in colonies which typically are counted from a distance with binoculars to limit disturbance, therefore there is less chance of recovering plastics from these nests. In addition, there are intensive study colonies (ca. 100-300 pairs per species) of all species except Southern Giant Petrel, where all breeding adults and chicks are ringed, and researchers check nests every 7–15 days from incubation to fledging (island wide fledgling counts not done for giant petrels). During these surveys, field biologists collect any items of anthropogenic origin. Each item is labelled with the date, location and species (Fig. 2.1). Items were unfortunately not always recorded per bolus or per nest, and therefore I could only use the total number of items sampled per year for each species in the analyses. Only items that could be associated with a specific species (i.e. found in or next to a nest) were included in this study. Most litter items recovered from boluses were assumed to come from chicks shortly before fledging as it is still largely unknown if albatrosses and giant petrels, apart from pre-fledging chicks (Young et al., 2009), offload ingested plastics while at sea or on land (Ryan, 2015c).

Fig. 2.1 Examples of marine litter items egested by albatrosses and giant petrels on Marion Island in the southwestern Indian Ocean from 1996 to 2018.

Samples were collected between 1996 and 2018, starting in May and ending in April the following year, to align with the summer-breeding seasons of most seabirds on Marion Island (i.e. 1996 refers to the 1996/97 sampling year). Sampling effort increased for Grey-headed Albatrosses from 2015, following the discovery that introduced House Mice *Mus musculus* were attacking Grey-headed Albatross chicks in April 2015 (Dilley et al., 2016). Since then, field biologists have checked these colonies more intensively for evidence of mouse attacks, entering colonies formerly only scanned from a distance, and thus increasing the likelihood of finding regurgitated plastic items. As a result, I restricted the comparisons of decadal changes in the amounts of litter for Grey-headed Albatrosses to the period up to 2015/16.

Sample processing

All nest pollutants found by field biologists on Marion Island between 1996–2018 were recorded, but only half of the items (751 of 1 486) were retained and available for detailed processing. These are referred to as the subsampled items throughout the manuscript and include samples across all sampling years and species. Each item (n = 1 486) was classified into one of six categories to help assign their potential source: 1) fishery-related items (hooks, fishing line, squid jigs, rope), 2) rigid plastic items (lids, hard fragments), 3) plastic bags, 4) food packaging, 5) other flexible plastic items, and 6) miscellaneous items (rubber gloves, expanded polystyrene pieces, pill packets, etc.) (Fig. 2.1; Tables 2.1 and S2.1). Within these broad categories, items were assigned to a range of sub-categories (Table S2.1). Each item was scored into one of eight colour groups following recommendations by Provencher et al. (2017): white/clear, red/pink, blue/purple, green, orange/brown, grey/silver, yellow and black. To compare my results with that of Phillips and Waluda (2020) at South Georgia, and to see if they accord with Thayer's law of counter-shading (Santos et al., 2016), I also grouped individual colours into light and dark groups (Table S2.2).

To determine the predominant polymer types of marine litter egested by albatrosses and giant petrels on Marion Island, a subset of rigid plastic litter items (n = 400) from the subsampled items (90%) spanning across all years and species was identified. Only rigid items were selected for analysis, as they were the most numerous items ingested by seabirds across all taxa, and therefore most likely the best representation of the predominant polymer types ingested by these seabirds. Items were randomly drawn from a bag until the desired number of samples were identified. Items were processed with ATR-FTIR using an iD7 ATR Nicolet iS5 (ThermoScientific) instrument under absorbance mode with spectral region from 400 to 4000

cm^{-1} and at a resolution of 4 cm^{-1}. OMNIC Spectra Software with Hummel Polymer- and HR Nicolet Sampler Libraries were used for automated identification of polymers and only assigned if there was ≥ 70% match. I also attempted to identify the country of origin of items based on any identifiable cues such as markings, labels or writing by using similar methods to Ryan et al. (2021) and Ryan et al. (2024).

For subsampled items only, dimensions (maximum length, width and thickness) were measured to the nearest 0.01 mm using digital callipers. Length was the longest dimension of the item and thickness the maximum depth perpendicular to the maximum length and maximum width plane (Roman et al., 2019b). Flexible items were extended to their maximum extent before measuring. I classified items into three commonly used size classes for marine litter (GESAMP, 2019) based on the maximum length of an item: macrolitter (> 25 mm), mesolitter (5–25 mm) and microlitter (< 5 mm). Items were weighed using a precision digital balance to the nearest 1 mg. Buoyancy was tested by placing each item in fresh water and noted as either buoyant (floats) or non-buoyant (sinks).

Data analyses

The total number, mass and proportion of each litter category and sub-category were summarised for each species and overall, pooling data for the two giant petrels due to small sample sizes from these species. Differences in the frequencies of categories were tested with chi-square goodness-of-fit tests. Categories were pooled (for low counts) to ensure that no more that 20% of expected counts were < 5, resulting in four categories: 1) fishery-related, 2) rigid plastics, 3) other flexible plastics and 4) all other litter. To assess if species preferred specific colours, I compared the colour frequency distributions of subsampled buoyant items within the eight colour groups for each species and tested for differences with chi-square goodness-of-fit tests. Grey/silver, yellow and black items were grouped as “other” to ensure that no more that 20% of expected counts were < 5. I limited the analyses to buoyant items ($n = 655$) as non-buoyant items were likely ingested secondarily (e.g. in fishing offal).

I used correspondence analysis and a contribution biplot to visualize differences in the main litter categories and colours (rows) ingested by the three seabird taxa (columns) using the *“FactoMineR”* and "*factoextra*" packages in R (Lê et al., 2008; Kassambara and Mundt, 2020; R Core Team, 2023). The relative distance between litter categories (or colours) of litter items sampled from boluses or at nests indicated how characteristic that litter type (or colour) is for that taxon. The more acute the angle of the arrows is between rows and columns, the stronger the relationship is. Temporal variation in polymer composition of rigid plastics was tested between species and decadal periods (1999–2008 and 2009–2018) with chi-square goodness-of-fit tests. I excluded polystyrene from the analyses as it constituted < 1% of samples.

Differences in the length, width, thickness and mass of subsampled items were pooled as follows: 1) rigid items (including hooks), 2) all flexible items (including fishing line and bags), and 3) all items combined, and compared among species. All data were assessed for normality using Shapiro-Wilk normality tests and as data were not normally distributed (all $P < 0.001$), Kruskal-Wallis tests were used to test for differences among species. A post-hoc Dunn's test with a Bonferroni correction (to decrease the chance of committing a Type 1 error) was used to show which species differed. To investigate if item size was related to species, I compared the maximum length to mass for all flexible items (e.g. flexible plastics, fishing line and rope) and the maximum length and width to mass ratio for all rigid items (e.g. hard plastic, hooks and squid jigs). I included the maximum width of rigid items because it could limit the swallowability of an item (determined by gape width, which roughly correlates with body size), although most studies only compare length to mass. Some studies have suggested that the size of buoyant hard fragments found at sea are decreasing over time (e.g. Morét-Ferguson et al., 2010). To evaluate if the size of hard fragments ingested by albatrosses and giant petrels has changed over time, I compared the lengths of items recorded between 1999–2008 and 2009–2018 using Mann-Whitney U tests. This was only done for Wandering Albatrosses because they were the only taxon with large enough sample sizes of hard fragments in both periods to perform a temporal comparison.

The relationship between the number of fishery-related litter items collected per year and annual toothfish yield (including estimates of illegal, unreported and unregulated catches; CCAMLR, 2017) was investigated using Spearman's rank correlation, with a lag of one year between catches and litter in seabird regurgitations due to the partial mismatch in sampling periods (calendar year for fish catches, and May-April for seabird litter). I compared the number of fishery-related items and all other litter items collected per year between the start of Patagonian Toothfish fishing operations around the island (1996/97), the height of toothfish fishing (1997/98 and 1998/99) and two decadal periods after 1999, which saw a decrease in fishing effort during the first decade to only two vessels operating in 2009/10–2018/19. I expressed changes as an index from -1 (decrease) through 0 (no change) to 1 (increase) calculated as $1 - (n \cdot year^{-1}_{smaller} / n \cdot year^{-1}_{larger})$, where $n \cdot year^{-1}$ is the average number of litter items of a given type per species per year. All statistical tests were conducted in the R computing environment (R Core Team, 2023).

Results

Characteristics of regurgitated litter

A total of 1 486 litter items was recorded, 56% from Wandering Albatrosses, 24% from Grey-headed Albatrosses and 20% from giant petrels (Table 2.1). Across all species, most items were rigid plastics (56%) and fishery-related items (36%), with small numbers of food packaging, plastic bags, other flexible plastic and miscellaneous items (Table 2.1). Of the rigid plastics, 77% were secondary hard fragments broken off from larger user items and 20% were lids or lid rings (Table S2.1). Most fishery-related items were 'snoods' (41%) (Fig. 2.1; Table S2.1). Categories of litter (fishery-related, rigid plastics, other flexible plastics and other litter) differed in the frequencies of items recorded among taxa ($\chi^2 = 256.8$, df = 6, $P < 0.001$). Rigid plastics comprised most of the litter items in Grey-headed Albatrosses (86%) and giant petrels (58%) (Table 2.1). Fishery-related items were most important in Wandering Albatrosses (51%), followed by giant petrels (28%) and Grey-headed Albatrosses (8%) (Table 2.1). The correspondence analysis showed that Wandering Albatrosses had a stronger association with fishery-related items and food packaging compared to giant petrels, which had a greater tendency to ingest plastic bags, polystyrene and miscellaneous items like rubber gloves (Fig. 2.2). Grey-headed Albatrosses had a greater propensity for ingesting rigid items such as hard fragments compared to the other taxa (Fig. 2.2).

Table 2.1 Main categories of regurgitated plastic and other litter items recorded at the nests of albatrosses and giant petrels on Marion Island between 1996–2018.

Category	Wandering Albatross % (n)	Grey-headed Albatross % (n)	Giant petrels % (n)	Total % (n)
Fishery related	51% (423)	8% (29)	28% (84)	36% (536)
Rigid plastics	39% (324)	86% (301)	58% (175)	54% (800)
Polyethylene bags	1% (12)	< 1% (1)	4% (13)	2% (26)
Food packaging	3% (26)	< 1% (1)	3% (9)	2% (36)
Other flexible plastic	2% (20)	4% (15)	3% (8)	3% (43)
Miscellaneous	3% (28)	1% (5)	4% (12)	3% (45)
Total	**56% (833)**	**24% (352)**	**20% (301)**	**1 486**

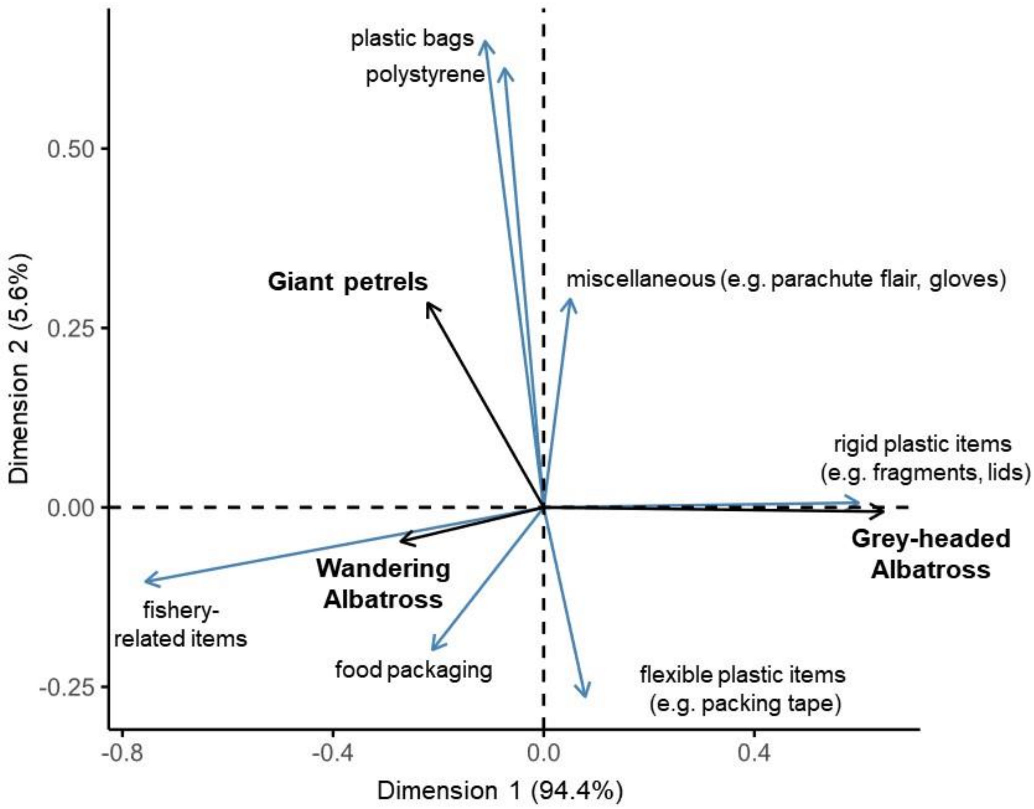


Fig. 2.2 Correspondence analysis indicating the relationship amongst the main litter categories egested by albatrosses and giant petrels on Marion Island.

Most subsampled litter items were buoyant (86%), and most non-buoyant items were fishery related items (95%), especially hooks, fishing lines and some ropes. Of the 655 buoyant litter items, most were white/clear (32%), red/pink (20%) and blue/purple (18%) (Table S2.2). Black items were seldom found (all species ≤ 1%; Table S2.2). The colour of litter items differed among taxa (χ^2 = 151.8, df = 10, P < 0.001) across 6 pooled colour categories (Table S2.2). Wandering Albatrosses showed a strong association with white/clear or blue/purple items, Grey-headed Albatrosses with red/pink and orange/brown and giant petrels with green and grey/silver items (Fig. 2.3). Most items were light (87%) compared to dark (13%).

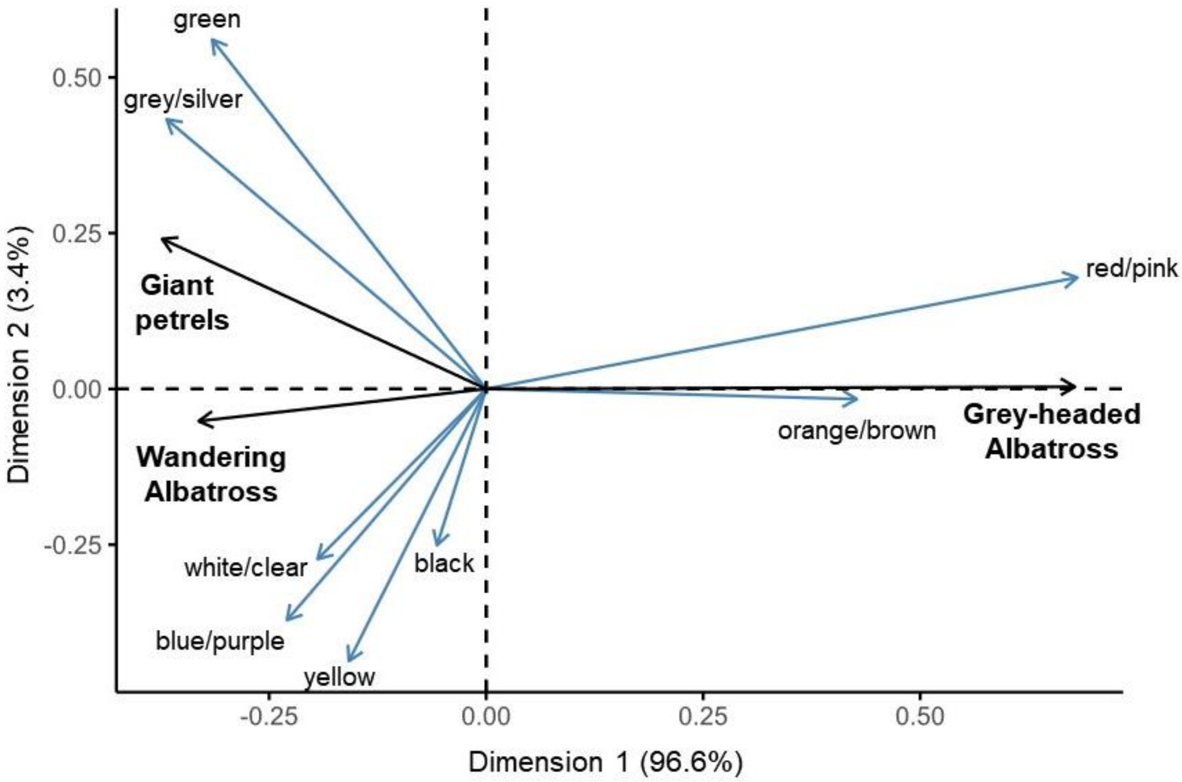


Fig. 2.3 Correspondence analysis indicating the relationship amongst the main colours (of buoyant items) egested by albatrosses and giant petrels on Marion Island.

Of the 400 rigid plastic pieces analysed using ATR-FTIR, 92% were identified to polymer type. Most were PE (55%) or PP (44%), with only two unexpanded PS items (< 1%; Table S2.3). There were no significant differences in the frequency of PE and PP (excluding PS from the analyses due to small sample size) among taxa (χ^2 = 3.90, df = 2, P = 0.14) or between the two decadal periods (all taxa combined: χ^2 = 0.11, df = 1, P = 0.64). I was able to confidently assign origins to only 22 items (3% of subsampled items). All these items were plastic and most originated from Asia (55%), followed by South America (18%), Africa and Europe (both 14%; Table S2.4). The most common items from Asia were lids from 'Danone Aqua' water bottles (50%), the largest bottled water company in Indonesia, followed by Asian food packaging (25%), and Malaysian drink bottle lids (17%). All items originating from South America were drink bottle lids, while items from Africa and Europe were mostly food packaging (67%; Table S2.4).

Most subsampled litter items (82%) were macrolitter (> 25 mm) (Table 2.2). The remainder (18%) were all mesolitter (5–25 mm) except for one microlitter (< 5 mm) item found at a Wandering Albatross nest: a small piece of rigid plastic possibly broken off from a larger ingested piece. Fishery-related items, food packaging and plastic bags were all macrolitter, as were most flexible plastic items (90%), with the highest proportions of mesolitter among rigid plastics and miscellaneous items (Table 2.2). Giant petrels had a lower percentage of macrolitter items (69%) than Grey-headed (74%) and Wandering Albatrosses (88%) (Table 2.2) because all of the rigid plastic pieces from their nests (nearly half of all subsampled items recorded for giant petrels) were mesolitter.

Table 2.2 Percentage (and number) of items classified as macrolitter (> 25 mm) in the main categories of the subsampled (n = 751) regurgitated plastic and other litter items recorded at the nests of albatrosses and giant petrels on Marion Island between 1996–2018.

	Wandering Albatross	Grey-headed Albatross	Giant petrels	Overall
Fishery related	100% (200)	100% (10)	97% (32)	100% (242)
Rigid plastic	77% (215)	71% (196)	0% (40)	71% (451)
Polyethylene bags	100% (6)	100% (1)	100% (3)	100% (10)
Food packaging	100% (17)	100% (1)	100% (2)	100% (20)
Other flexible plastic	83% (12)	100% (8)	100% (1)	90% (21)
Miscellaneous	67% (3)	50% (2)	100% (2)	71% (7)
Total	**88%** (453)	**74%** (218)	**69%** (80)	**82%** (751)

There were significant differences in subsampled items among species in the dimensions and mass of all items combined (length: $\chi^2 = 82.83$, $P < 0.001$; width: $\chi^2 = 42.16$, $P < 0.001$; thickness: $\chi^2 = 8.08$, $P < 0.05$; mass: $\chi^2 = 53.98$, $P < 0.001$), as well as flexible items (length: $\chi^2 = 15.51$, $P < 0.001$; width: $\chi^2 = 13.80$, $P < 0.001$; thickness: $\chi^2 = 11.68$, $P = < 0.05$; mass: $\chi^2 = 15.01$, $P < 0.001$) and rigid items (length: $\chi^2 = 32.47$, $P < 0.001$; width: $\chi^2 = 45.64$, $P < 0.001$; thickness: $\chi^2 = 10.44$, $P < 0.05$; mass: $\chi^2 = 38.98$, $P < 0.001$; df = 2; Table S2.5). Across all categories, Wandering Albatrosses tended to regurgitate larger, wider, thicker and heavier items than Grey-headed Albatrosses and giant petrels (Table S2.5). The relationship between the length to mass ratio of flexible items showed considerable overlap between species, but Wandering Albatrosses ingested larger and heavier flexible items than Grey-headed Albatrosses and giant petrels (Fig. 2.4A). Similarly, the relationship between the length and width to mass ratio of rigid items showed that Wandering Albatrosses generally ingested longer (apart from a toothbrush and a large user item fragment ingested by Grey-headed Albatrosses; Fig. 2.4B), wider (Fig. 2.4C) and heavier items than the other two taxa, although again there was considerable overlap (Fig. 2.4B and C). Between the two time periods, the length of hard plastic fragments ingested by Wandering Albatrosses increased from a median (±SE) of 29 ± 2.0 (range: 7.7-114.0) mm in 1999–2008 to 37.0 ± 1.7 (13.5-114.0) mm in 2009–2018 ($Z = 2.27$, $P < 0.05$). Item thickness increased ($Z = 3.25$, $P < 0.05$) from 2.7 ± 0.5 (0.4-30.1) mm to 2.8 ± 0.3 (0.7-23.9) mm. Item mass also increased from 0.9 ± 0.1 (0.03-6.0) g to 1.1 ± 0.1 (0.1-5.6) g ($Z = 2.1$, $P < 0.05$).

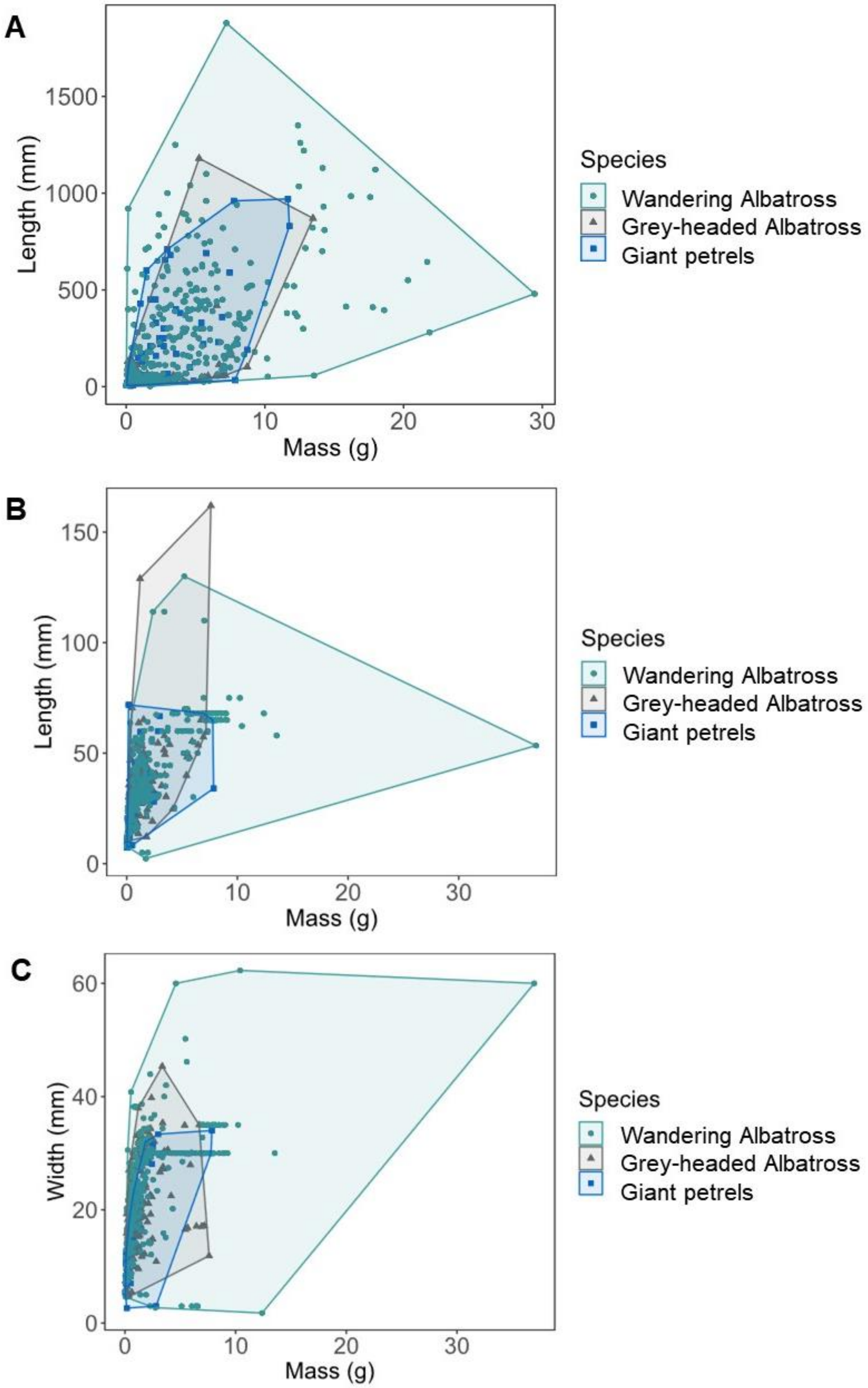


Fig. 2.4 Biplots of the relationships between A) the maximum length (mm) and mass (g) of all subsampled (n = 751) flexible items (including fishing line), B) the maximum length (mm) and mass (g) of all rigid items (including fishing hooks and squid jigs), and C) the maximum width (mm) and mass (g) of all rigid items (including fishing hooks and squid jigs) regurgitated by albatrosses and giant petrels on Marion Island.

Trends in amounts of litter

Compared to the start of the toothfish fishery (1996/97), the numbers of both fishing gear and other litter items increased greatly during the height of operations (1997/98–1998/99) (Table 2.3; Fig. 2.5). There was a strong correlation between the number of fishery-related items recorded each year and estimated toothfish catches ($r_s = 0.72$, df = 21, $P < 0.05$; Fig. 2.5). As fishing effort dwindled from 1999, the number of fishery-related items decreased across all species, and in the decade from 2009–2018, it was close to the level recorded at the start of toothfish fishing operations (Table 2.3). The number of other litter items collected each year decreased sharply during 1999–2008, but then increased during 2009–2018 (Table 2.3). However, the increases differed among species, with little change in Wandering Albatrosses while Grey-headed Albatrosses and giant petrels nearly doubled. The index of change in litter between the two decadal periods (1999–2008 and 2009–2018) showed a decrease in the frequency of fishery-related items (-27%), mostly driven by strong decreases in Grey-headed Albatrosses (-44%) with more modest decreases in Wandering Albatrosses (-26%) and giant petrels (-22%) (Fig. 2.6). By comparison, the mean number of other litter items (e.g. hard plastic pieces, food packaging, bags, other flexible plastics and miscellaneous items) increased overall (111%), driven mainly by a nearly two-fold increase in Grey-headed Albatrosses (190%, even excluding data from after 2015/16) and giant petrels (151%) (Fig. 2.6). Increases over this period were minimal in Wandering Albatrosses (101%) (Fig. 2.6).

Table 2.3 Rate ($n.year^{-1}$) of fishery-related and other litter items recorded for albatrosses and giant petrels during the start of the toothfish fishery (1996/97), the height (1997/98–1998/99) of operations and the following two decades: 1999–2008 and 2009–2018.

	1996/97	1997/98–1998/99	1999–2008	2009–2018
Fishery-related				
Wandering Albatross	10	81.5	14.4	10.6
Grey-headed Albatross	1	9.0	0.3	0.2*
Giant petrels	1	25.5	1.8	1.4
Other litter				
Wandering Albatross	11	39.0	16.0	16.1
Grey-headed Albatross	15	30.5	2.9	5.5*
Giant petrels	14	50.0	4.1	6.2

*only up to 2014/15

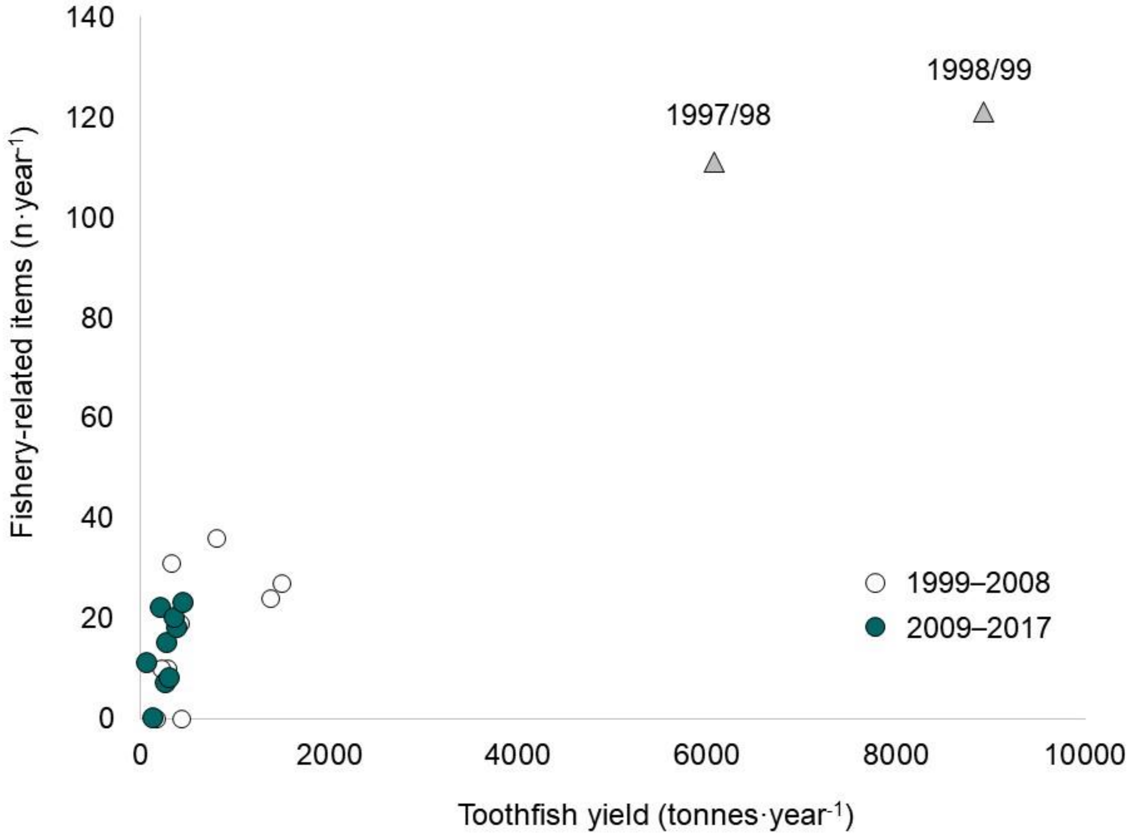


Fig. 2.5 The relationship between fishery-related litter items (n·year^{-1}) recorded in nests of albatrosses and giant petrels on Marion Island and the annual toothfish yield (including estimates for illegal, unreported and unregulated efforts; from CCAMLR, 2017) within the EEZ around the Prince Edward Islands (1996–2017).

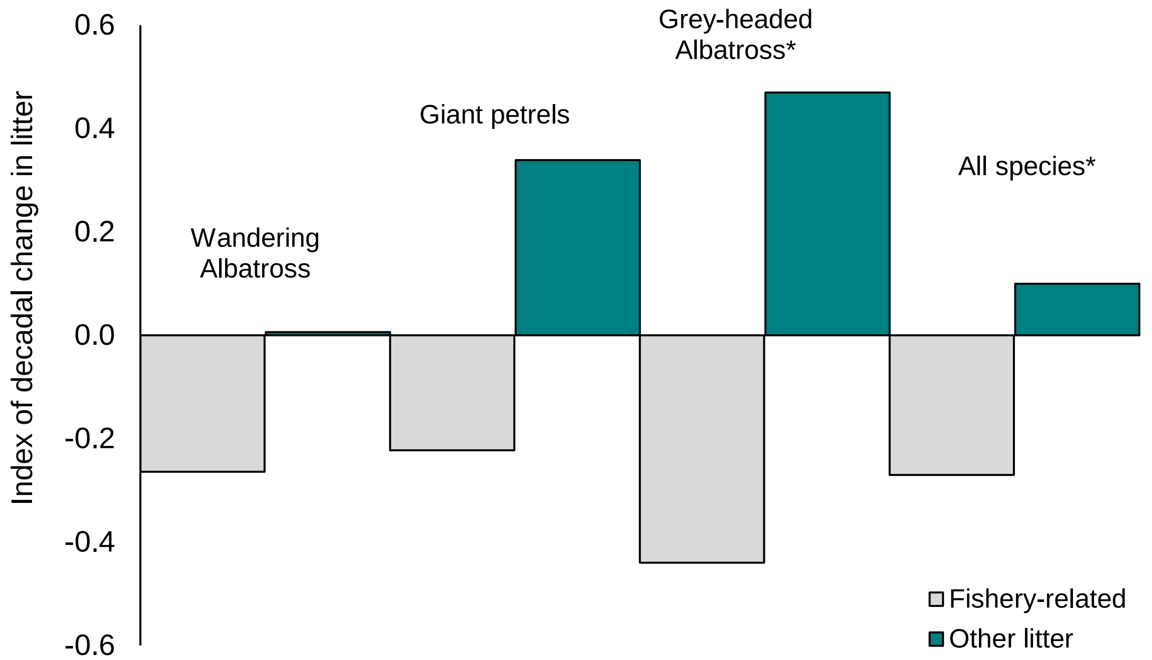


Fig. 2.6 Decadal changes in the mean number of litter items recorded after the height of toothfish fishing operations (1999–2008 and 2009–2018) around Marion Island for fishery-related and other litter items collected at Wandering Albatross, Grey-headed Albatross and giant petrel nests. All species represents combined data across all species (* = excluding Grey-headed Albatross data recorded from 2015/16 onwards).

Discussion

Seabirds may ingest litter items either directly while foraging at sea (primary ingestion) or in contaminated prey (secondary ingestion; Ryan, 2016). One third of items recorded in this chapter were fishery-related, most of which were non-buoyant (hooks attached to snoods or monofilament line; Fig. 2.1) and were likely ingested secondarily in non-target fish or heads of targeted fish that were discarded with the hooks still in them. Most non-fishery litter items were buoyant and probably too large to have been ingested by prey, suggesting that primary ingestion was likely the route of intake. However, the intestines of larger fish discarded by fishing vessels might contain larger litter items that could have been ingested secondarily (Justino et al., 2023). Albatrosses and giant petrels often scavenge prey at the water surface, and probably ingest floating plastic and other litter items because they are perceived as potential food items (Ryan, 1987a; Santos et al., 2016).

There were distinct differences between the types of litter items among taxa. Wandering Albatrosses regurgitated mostly fishery-related items, whereas giant petrels and Grey-headed Albatrosses regurgitated more rigid plastic items. Wandering Albatrosses and giant petrels showed a greater proclivity towards food packaging, plastic bags and miscellaneous items than Grey-headed Albatrosses, reflecting their tendency to scavenge behind vessels (Griffiths, 1982; Otley et al., 2007; Weimerskirch et al., 2020). The same pattern was found for these species at South Georgia (Phillips and Waluda, 2020). All four species' foraging ranges during breeding extend north of the Sub-tropical Front (Carpenter-Kling et al., 2020), into waters where there is considerable long-line fishing effort for tunas (Ryan et al., 1997; Nel et al., 2001, Nel et al., 2002b; Tuck et al., 2003). Here they are also more likely to encounter higher densities of floating litter as Suaria et al. (2020b) found a nearly 10-fold greater density of macrolitter items and a nearly 50-fold greater mean density of meso-litter items (mostly small rigid plastic pieces) north of the Sub-tropical Front (Fig. 1.6) compared to the Southern Ocean. Connan et al. (2021) also reported similar trends in the Indian Ocean, where only 4 floating anthropogenic items were recorded south of the front, compared to 1 604 items to the north.

I found similar results in the size of regurgitated items as on South Georgia, where albatrosses ingested larger and heavier items than giant petrels (Phillips and Waluda, 2020). The relationship between the width of rigid items and bird size confirms that larger species are capable of ingesting larger items, but the considerable overlap in the size of ingested litter among species shows that body size does not necessarily determine the lower limit of items ingested. Wandering Albatrosses and giant petrels exhibited similar colour preferences, ingesting mostly white/clear, blue/purple and green items, whereas Grey-headed Albatrosses mostly regurgitated red/pink items. Ryan (1987a) found similar colour preferences for these species at Marion Island in the 1980s, and pale items (87%) also were preferred over dark colours (13%) by the same species at South Georgia (light = 93 %, dark = 7 %; Phillips and Waluda, 2020). This accords with Thayer's Law of counter-shading, which suggests that surface feeding seabirds should ingest paler floating items because they contrast with the darker water (Santos et al., 2016; Phillips and Waluda, 2020). However, this might also be due to a lower detection probability by field biologists for darker items as these colours probably are less conspicuous than bright or pale items next to nests. Furthermore, it is difficult to assess colour preferences in ingested litter without prior knowledge on background availability in the marine environment (Kain et al., 2016). Further studies addressing the composition and colour of floating litter at-sea within the foraging ranges of these species are needed to draw firm conclusions.

Sources of ingested litter

Fishery-related items undoubtedly originated from fishing vessels, supported by the high frequency recorded during the height of toothfish fishing operations around the islands (Fig. 2.5). Virtually all fishing effort within the ranges of these seabirds, while breeding on Marion Island, is long-line fishing (Nel et al., 2000; Nel et al., 2002a; Nel et al., 2002b). Some of the hooks and snoods could have been lost accidentally, but most probably are discarded in fish heads and non-target species, indicating the need for better controls on offal management. Some of the snoods probably were ingested with hooks attached, and the hooks digested (Ryan, 2015c). Most of the food packaging and PE bags manufactured in Asia, Africa and Europe were also likely discarded from ships, either accidentally or deliberately because such items entering the ocean from land-based sources seldom remain adrift long enough to be encountered by albatrosses and giant petrels. These items have large surface area to volume ratios and biofouling reduces their buoyancy within weeks, limiting their dispersal ability (Fazey and Ryan, 2016). As a result, very few bags and flexible food-packages are recorded floating in the Southern Ocean (e.g. Ryan et al., 2014; Ryan, 2015b; Suaria et al., 2020b). The notion that such items are likely discarded or lost from ships is also supported by the high rates of non-fishery litter recorded in 1997/98 and 1998/99, when many fishing vessels operated around the islands. Phillips and Waluda (2020) recorded similar patterns in the same seabird species at South Georgia, and concluded that South American fishing fleets were most likely a major source of these types of litter.

Identifying the origins of rigid plastic items floating in the open ocean is challenging, as many are fragments of larger items (longer lasting user or single-use plastic items) with few identifiable markings. Some may come from land-based sources as they are capable of drifting farther compared to less buoyant flexible packaging (Fazey and Ryan, 2016). Some may also originate from ship-based sources e.g. plastic drink bottles, pieces of fishing trays, plastic buckets, plastic hard-hats, abandoned Fish Aggregating Devices (FADs) and other user items lost at sea. Items that have identifiable markings like logos or brand names can often be traced to the country of manufacture (Ryan et al., 2019). Most of the identifiable items regurgitated by albatrosses and giant petrels on Marion Island were from Asia, with Indonesian water bottle lids making up the bulk (Fig. 2.1). Duhec et al. (2015) found that most lids washed up on beaches in the Seychelles were from the same brand, suggesting that HDPE drink bottle lids from southeast Asia drift long distances across the Indian Ocean. Ryan et al. (2024) found that the majority of foreign plastic drink bottles (with lids) and loose drink bottle lids that washed ashore on South African beaches were manufactured in Asia. The drink bottles with lids, mostly made in China, Malaysia/Singapore and UAE, were recently manufactured and in good condition, and therefore likely dumped from vessels operating in the surrounding waters (see also Ryan et al., 2019), whereas most loose bottle lids were made in Indonesia and were in poor condition, and likely drifted across the Indian Ocean before stranding on South African beaches (Ryan et al., 2024). The drink bottle lids recovered from boluses on Marion Island showed signs of prolonged weathering (Fig. 2.1), indicating that they likely drifted from Asia before being encountered by albatrosses and giant petrels in the southern Indian Ocean. Larger identifiable litter items such as drink bottle lids ingested by these seabirds at sea therefore provide valuable information to help us understand the sources and transport pathways of marine litter. These findings can be applied to monitor the effectiveness of waste-management and source-to-sea prevention strategies over time.

Temporal variation in litter abundance

Overall, the amount of fishery-related litter recorded at seabird nests on Marion Island decreased after 1997/98–1998/99, coinciding with the marked decrease in fishing effort for Patagonian Toothfish around the Prince Edward Islands (Fig. 2.5). A further decrease was noted between the decadal periods as in 2010, the number of toothfish vessels legally permitted to fish around the islands were restricted to two (CCAMLR, 2017). The greatest decrease was recorded in Grey-headed Albatrosses, and less so for Wandering Albatrosses and giant petrels, likely caused by their tendency to follow ships and forage closer to the African continent, within tuna longline fishing grounds (Nel et al., 2002a; Nel et al., 2002b; Tuck et al., 2003), as compared to Grey-headed Albatrosses, thus increasing their exposure to fishery-related litter.

The high rate of other litter items recorded between 1997/98, compared to 1996/97 and the decades following 1998, suggests that many of these items also were linked to toothfish fishing operations, and discarded from vessels close to the island. Apart from Wandering Albatrosses, the number of other litter items recorded at the start of toothfish fishing operations (1996/97) was more than double the annual number in 2009–2018. However, the relatively large number collected in 1996/97 might also have been inflated by litter accumulated over several years, because routine collection of nest pollutants only started in 1996 (Table 2.3). In contrast to the decrease in fishery-related litter, the frequency of other litter recorded at seabird nests on Marion Island nearly doubled between the two decades, driven by marked increases in giant petrels and especially Grey-headed Albatrosses. This exceeds the growth in global plastic production, which increased by roughly two-thirds between 1996–2004 and 2010–2018 (Geyer et al., 2017; OECD, 2022). The increase in the size of hard fragments ingested by Wandering Albatrosses among the two decadal periods might further suggest that greater proportions of larger hard plastic fragments are available on the sea surface now, than compared to earlier decades. This however needs further investigation, as the complex dynamics behind fragmentation rates and the longevity of plastics on the sea surface are still largely unknown, and influenced by several factors (Ryan, 2015b; Kaandorp et al., 2021; Andrady, 2022). The results from Marion Island contrast with other studies, which show little change in floating plastics at sea or in biota over the last 30 years (e.g. Ryan, 2008; van Franeker and Law, 2015; Beer et al., 2018; Chapters 4 and 5). Analysing comparative data for the same bird species at South Georgia in the same was as was done here (total number of items sampled per year) shows slight decreases in Wandering Albatrosses (-38%) and giant petrels (-0.02%), but an increase in Grey-headed Albatrosses (121%) between 1999–2008 and 2009–2018 (Phillips and Waluda, 2020). The similar ranked responses in the pattern of litter loads at the two islands (Grey-headed Albatrosses > giant petrels > Wandering Albatrosses) suggest that there are consistent species-specific differences across the Southern Ocean, with Grey-headed Albatrosses more sensitive to changes in the amount and composition of plastic at sea than the other two taxa.

The greater increases in non-fishery litter across all species at Marion Island compared to South Georgia presumably reflects regional differences in the growth of plastic litter at sea, but may also reflect contrasting population trends. Population sizes of all three taxa on Marion Island have increased over the last decade (Ryan and Connan, unpubl. data; Stevens et al., in press), while albatross populations at South Georgia have decreased, Northern Giant Petrels have increased slowly and Southern Giant Petrels have remained stable (Philips and Waluda, 2020). The amount of litter collected annually in my study was reported as totals rather than as a rate per nest checked. Therefore, with growing populations, we might expect an increase in the total litter amounts per year. This means that the increase in litter items over the last decade may simply reflect the growing population sizes rather than an actual increase in litter amounts. To prevent this in future, I recommend that plastics should be recorded per nest, and rates analysed as number of items per nest per year. Changing the protocol as such will account for growing population sizes when calculating temporal trends in litter densities. However, the albatrosses and giant petrels breeding at Marion Island forage in the outer litter accumulation zone of the Indian Ocean (Carpenter-Kling et al., 2020), which has higher concentrations of floating litter than the outer accumulation zone in the southwest Atlantic Ocean (Cózar et al., 2014; Eriksen et al., 2014; Connan et al., 2021). This region derives most of its long-distance litter from Asia (Lebreton et al., 2012; Duhec et al., 2015; Irfan et al., 2024), as shown by the composition of litter items found on Marion Island (Table S2.4). My results suggest that litter in the Indian Ocean gyre may have increased over the last few decades, in contrast to that in the South Atlantic Ocean. Beach litter surveys at South Georgia show only a slight increase in the numbers of stranding litter items, and a decrease in the total mass of plastic stranding over the last few decades (Waluda et al., 2020), which is consistent with the modest changes in non-fishery litter in seabirds at this site. Although the Southern Ocean is still the cleanest of the world's oceans (Barnes et al., 2010; Suaria et al., 2020b), the rapid growth in non-fishery litter at Marion Island highlights regional differences in the exposure of sub-Antarctic seabirds to elevated plastic loads in adjacent temperate waters that lie within their foraging ranges.

Conclusions

This chapter underscores the value of using albatrosses and giant petrels as a monitoring tool to track regional changes in the types, amounts and sources of marine litter in the open ocean (Ryan et al., 2009a; GESAMP, 2019). Their ability to sample large litter items that can be assigned to source, and thus evaluate the efficacy of mitigation measures aimed at reducing the inputs of readily identifiable litter items such as drink bottle lids over time, furthers their utility as important monitors of marine plastic pollution. They are also useful monitors of fishing practices in remote oceanic areas. This monitoring data can aid decision making processes aimed at reducing the amount of litter available to seabirds at sea. The continued long-term monitoring of plastic litter items ingested by albatrosses and giant petrels on Marion Island using this non-invasive and simple method, is therefore recommended.

Supplementary materials: Chapter 2

Table S2.1 Number and percentage of the main categories and types of regurgitated plastic and other litter items recorded at the nests of albatrosses and giant petrels on Marion Island between 1996–2018.

Category	Wandering Albatross n	Grey-headed Albatross n	Giant petrels n	Total n
Fishery related	**51% (424)**	**8% (29)**	**28% (84)**	**36% (537)**
Rope snoods*	219	17	65	301
Hook and line	85	2	7	94
Hook and rope	69	0	4	73
Monofilament line	30	0	6	36
Squid jigs	5	8	1	14
Hook	13	0	1	14
Float	1	2	0	3
Net	2	0	0	2
Rigid plastics	**39% (324)**	**86% (301)**	**58% (175)**	**54% (800)**
Fragments	242	226	149	617
Lid (drink)	52	37	14	103
Lid (other)	20	25	6	51
Lid ring	1	3	0	4
Toy	0	4	2	6
Button	1	1	0	2
Rawl plug	1	1	0	2
Plastic jar	2	0	0	2
Toothbrush	1	1	0	2
Balloon holder	1	0	0	1
Earbud stick	0	0	1	1
Molten plastic	1	1	0	2
Plastic spoon	1	0	0	1
Toothpaste tube	1	0	0	1
Clothes peg	0	1	2	3
Pen	0	1	1	2
Polyethylene bags	**1% (12)**	**< 1% (1)**	**4% (13)**	**2% (26)**
Plastic bags	12	1	13	26
Food packaging	**3% (25)**	**< 1% (1)**	**3% (9)**	**2% (35)**
Food wrap	23	1	7	31
Sweet wrap	1	0	1	2
Vegetable net bag	1	0	1	2
Other flexible plastic	**2% (20)**	**4% (15)**	**3% (8)**	**3% (43)**
Pieces	14	9	5	28
Packing strip	2	2	1	5
Insulation tape	1	2	0	3
Lid liners	1	1	1	3
Plastic ribbon	1	1	0	2
Cable tie	1	0	0	1

Plastic sticker	0	0	1	1
Miscellaneous	**3% (28)**	**1% (5)**	**4% (12)**	**3% (45)**
Rubber gloves	15	2	4	21
Expanded polystyrene	8	3	3	14
Foil	1	0	2	3
Pill packet	2	0	0	2
Parachute flare	0	0	1	1
Material	0	0	1	1
Elastic band	0	0	1	1
Silicon	1	0	0	1
Leather belt	1	0	0	1
Total	**833**	**352**	**301**	**1 486**

*lines used to attach hooks to longline mainlines

Table S2.2 Proportion of individual colours recorded per species for all subsampled buoyant items (n = 655). Items are separated into two groups: light (white/clear, blue/purple, red/pink, grey/silver, orange/brown, yellow), and dark (green and black) as in Phillips and Waluda (2020).

Colour	**Wandering Albatross** n = 369	**Grey-headed Albatross** n = 215	**Giant petrels** n = 71	**All species** n = 655
Light				
White/clear	36%	25%	34%	32%
Red/pink	10%	41%	11%	20%
Blue/purple	21%	11%	18%	18%
Orange/brown	5%	18%	4%	9%
Grey/silver	9%	-	10%	6%
Yellow	3%	< 1%	1%	2%
Dark				
Green	15%	5%	21%	13%
Black	< 1%	-	-	< 1%

Table S2.3 Frequencies of plastic polymers polyethylene (PE), polypropylene (PP) and polystyrene (PS) recorded for rigid plastic pieces (number of items in brackets) between 1999–2008, 2009–2018 and overall (1999–2018) around nests and in colonies of albatrosses and giant petrels on Marion Island. Only items with a polymer match of ≥ 70% are presented. The total number identified (with number analysed in brackets) is also included.

Polymer	**Wandering Albatross**	**Grey-headed Albatross**	**Giant petrels**	**Total**
1999–2008				
PE	52% (32)	20% (1)	80% (4)	51% (37)
PP	47% (29)	80% (4)	20% (1)	47% (34)
PS	1% (1)	0% (0)	0% (0)	1% (1)
2009–2018				
PE	57% (64)	52% (86)	68% (19)	55% (169)
PP	42% (47)	48% (80)	32% (9)	44% (136)
PS	1% (1)	0% (0)	0% (0)	1% (1)
1999–2018				
PE	55% (96)	51% (87)	70% (23)	54% (206)
PP	44% (76)	49% (84)	30% (10)	44% (170)
PS	1% (2)	0% (0)	0% (0)	< 1% (2)
Total identified	**174 (184)**	**171 (177)**	**33 (39)**	**378 (400)**

Table S2.4 Origin of subsampled plastic and other litter items (n = 22) including the litter type and count recorded in albatross and giant petrel colonies on Marion Island between 1996–2018.

Origin	Litter type (n)	Species	%
Asia			**55%**
Indonesia	Lid (drink) (6)	Wandering Albatross	
Japan	Lid (drink) (1)	Grey-headed Albatross	
Malaysia	Lid (drink) (2)	Wandering Albatross	
Asian	Food packaging (3)	Wandering Albatross	
South America			**18%**
Argentina	Lid (drink) (1)	Wandering Albatross	
Argentina	Lid (drink) (1)	Northern Giant Petrel	
Brazil	Lid (drink) (1)	Wandering Albatross	
South American	Lid (drink) (1)	Wandering Albatross	
Africa			**14%**
Mauritius	Food packaging (1)	Wandering Albatross	
South Africa	Food packaging (1), plastic bag (1)	Wandering Albatross	
Europe			**14%**
United Kingdom	Food packaging (1)	Wandering Albatross	
Scotland*	Lid liner (1)	Grey-headed Albatross	
Spain	Food packaging (1)	Southern Giant Petrel	

*Lid liner from a bottle of Scottish whisky (widely distributed).

Table S2.5 Summary of the mean ± SD (range) length, width, thickness (mm) and mass (g) of all subsampled items for all rigid items (including hooks), flexible items (including monofilament line) and all items combined that were regurgitated by seabirds at their nests on Marion Island (1996–2018). Dunn's test results to indicate differences between taxa are also included. WA = Wandering Albatross; GHA = Grey-headed Albatross; GPs = Giant Petrels

	Wandering Albatross	**Grey-headed Albatross**	**Giant petrels**	**Dunn's test**
Length				
Rigid items	41.3 ± 19.5 (2–130)	34.1 ± 15.7 (10 – 162)	27.6 ± 19.2 (7–72)	WA ≠ GHA ($Z = 3.49$, $P < 0.05$) WA ≠ GPs ($Z = 5.21$, $P < 0.001$) GHA ≠ GPs ($Z = 3.16$, $P < 0.05$)
Flexible items	393.6 ± 300.1 (6–1879)	207.6 ± 314.1 (24–1180)	327.1 ± 264.7 (5–970)	WA ≠ GHA ($Z = 3.82$, $P < 0.001$) GHA ≠ GPs ($Z = -2.48$, $P < 0.05$)
All items	220.6 ± 281.1 (2–1879)	48.4 ± 101.2 (10–1180)	175.5 ± 240.8 (5–970)	WA ≠ GHA ($Z = 8.99$, $P < 0.001$) WA ≠ GPs ($Z = 3.70$, $P < 0.05$) GHA ≠ GPs ($Z = -2.55$, $P < 0.05$)
Width				
Rigid items	24.3 ± 9.9 (2–62)	20.0 ± 7.8 (1–45)	15.9 ± 9.4 (3–34)	WA ≠ GHA ($Z = 5.16$, $P < 0.001$) WA ≠ GP ($Z = 5.45$, $P < 0.001$) GHA ≠ GPs ($Z = 2.49$, $P < 0.05$)
Flexible items	13.7 ± 34.2 (0.4–300)	23.4 ± 22.2 (1–91)	21.1 ± 42.2 (0.5–220)	WA ≠ GHA ($Z = -3.71$, $P < 0.001$) GHA ≠ GPs ($Z = 2.80$, $P < 0.05$)
All items	18.2 ± 25.8 (0.4–300)	20.3 ± 9.7 (1–91)	18.2 ± 30.4 (0.5–220)	WA ≠ GHA ($Z = -5.83$, $P <0.001$) GHA ≠ GPs ($Z = 5.06$, $P < 0.001$)
Thickness				
Rigid items	5.3 ± 7.2 (0.4–60)	5.4 ± 5.2 (0.7–30)	3.3 ± 3.4 (0.8–14)	WA ≠ GHA ($Z = -2.44$, $P < 0.05$)

				GHA ≠ GPs (Z = 2.78, P < 0.05)
Flexible items	3.0 ± 2.7 (0–24)	1.6 ± 2.0 (0.1–8)	3.0 ± 2.0 (0.3–7)	WA ≠ GHA (Z = 3.34, P < 0.05 GHA ≠ GPs (Z = -3.09, P < 0.05)
All items	4.5 ± 6.0 (0–60)	5.2 ± 5.2 (0.1–30)	3.2 ± 2.9 (0.3–14)	GHA ≠ GPs (Z = 2.55, P < 0.05)
Mass				
Rigid items	2.7 ± 3.4 (0.01–37)	1.3 ± 1.4 (0.01–8)	1.3 ± 2.0 (0.01–8)	WA ≠ GHA (Z = 4.52, P < 0.001) WA ≠ GPs (Z = 5.25, P < 0.001) GHA ≠ GPs (Z = 2.65, P < 0.05)
Flexible items	5.4 ± 4.6 (0.01–29)	2.6 ± 3.8 (0.1–14)	3.5 ± 3.0 (0.2–12)	WA ≠ GHA (Z = 3.26, P < 0.05)
All items	3.5 ± 4.3 (0–37)	1.4 ± 1.7 (0.01 - 14)	2.1 ± 2.7 (0.01–12)	WA ≠ GHA (Z = 7.01, P < 0.001) WA ≠ GPs (Z = 3.69, P < 0.001)

Chapter 3

Investigating the use of skua pellets containing the remains of petrels to monitor small buoyant plastics at sea

Top left: Deploying a neuston net onboard the R.V. *S.A. Agulhas II* to trawl for small buoyant plastics (photo Alex Oelofse). Top right: Brown Skua *Catharacta antarctica* in flight (photo Peter Ryan). Bottom left: examples of the types and colours of small buoyant plastics sampled with a neuston net (photo Vonica Perold). Bottom right: Regurgitated Brown Skua pellets containing seabird prey remains and ingested plastics (photo Peter Ryan).

Abstract

Seabirds vary in the types of plastics they ingest, and therefore differ in the types of marine plastics they can be used to monitor over time. Using seabirds as bioindicators of marine plastic pollution requires an understanding of how the plastic retained in each species compares with that found in their environment. This is typically done using surface net trawls that mostly sample plastics < 25 mm in size. Various methods have been used to sample ingested plastics in seabirds. Some are direct and invasive, while some, like using regurgitated predator pellets containing the remains of seabird prey, have been suggested as an alternative. In this chapter, I show that Brown Skua pellets can be used to characterise plastics in four petrel taxa breeding in the central South Atlantic, even though skua pellets might underrepresent the smallest plastic items in their prey. I found that overall, *Fregetta* storm petrels ingested more thread-like plastics and White-faced Storm Petrels more industrial pellets than Broad-billed Prions and Great Shearwaters. The composition of ingested plastics (type, colour and polymer) was similar to floating plastics in the region sampled with a 200 µm net, but storm petrels were better indicators of the size of plastics than prions and shearwaters. Given this information, I propose that plastics in skua pellets containing the remains of petrels can be used to track long-term changes in small floating marine plastics.

Introduction

The global surge in plastic production has resulted in a rapid increase in the amounts of mismanaged plastic waste (Geyer et al., 2017; Lebreton and Andrady, 2019; Fig. 1.2) and governments have been pressed to implement policies and mitigation measures aimed at curbing the leakage of plastic waste into the environment (Walker, 2021). For example, the United Nations' Agenda 2030 and Sustainable Development Goal (SDG) 14.1 aim to significantly reduce marine pollution by 2025 (https://www.un.org/sustainabledevelopment/). However, in order to evaluate the effectiveness of waste reduction policies and mitigation measures, baseline data on plastic types and densities at sea must be available for comparison (Ryan et al., 2009a; Morét-Ferguson et al., 2010; Lebreton et al., 2017; Ryan et al., 2020a). This is typically done by assessing the abundance and composition of small buoyant plastics on the ocean's surface (e.g. Maes et al., 2017; Lebreton et al., 2018; González-Fernández et al., 2022; Weideman et al., 2023). However, at-sea sampling poses several challenges, and considerable effort and substantial resources are required to obtain robust results (Chapter 1).

Most studies determine the abundance of floating plastic items at sea by towing a fine-meshed net along the surface of the ocean (Fig. 1.4). The density of plastics is calculated from the number of plastic items collected within the sampled area (Ryan et al., 2009a; GESAMP, 2019). However, this is time-consuming and expensive, and each net tow only samples a tiny area (typically < 0.001 km^2) and are usually restricted to small buoyant items like user fragments and industrial pellets (Law et al., 2010; Cózar et al., 2014; Weideman et al., 2023). Ekman circulation creates local convergence patterns, causing variability and clustering of floating plastics at small spatial scales (van Sebille et al., 2020), and this, coupled with the small area sampled per tow, results in a high degree of heterogeneity in the amount of plastic sampled (Ryan et al., 2009a; Law et al., 2014). To address this uneven distribution requires large numbers of surface net tows covering extensive areas. In addition, high winds and waves cause vertical mixing of plastics at the sea surface, reducing the likelihood of sampling floating plastics with a shallow surface net (Kukulka et al., 2012; Reisser et al., 2015). Adverse weather also can prevent net deployments (Law et al., 2014), which can limit sample sizes. Together, these factors make it challenging to detect temporal or spatial changes in floating plastic abundance from surface net tows (Ryan et al., 2009a).

As discussed in Chapters 1 and 2, plastics ingested by biota such as fish, turtles, and seabirds provide an alternative to at-sea sampling as they generally forage over large areas, are abundant and inexpensive to sample (Ryan, 2008; Ryan et al., 2009a; van Franeker and Law, 2015; Bray et al., 2019; Savoca et al., 2022; Thibault et al., 2023). Albatrosses and petrels (Procellariiformes) are surface feeders that often ingest marine litter, but is thought to seldom regurgitate it, except when feeding chicks (Young et al., 2009; Ryan, 2015c; Chapter 2). Petrels however have a greater frequency of ingestion and can retain plastics for longer than larger albatrosses (Furness, 1985; Ryan, 2015c; Chapter 1). This greater frequency of ingestion and prolonged retention means that the types and quantities of small buoyant plastics in the environment are integrated over space and time, and as a result, petrels have often been used to monitor temporal variation in the amounts and types of small buoyant plastics floating at sea (Furness, 1985; Ryan, 2008; van Franeker et al., 2011; Ryan, 2015a; Ryan, 2016; Avery-Gomm et al., 2018; Rodríguez et al., 2024).

Plastic items that petrels store in their stomachs are mainly ingested directly by the birds as they forage at sea, and which they presumably mistake for prey items. This is indicated by the fact that they mimic certain characteristics resembling food or prey items e.g. type, size, buoyancy, colour or conspicuousness (Ryan, 1987a; Ryan, 2016; Santos et al., 2016; Roman et al., 2019a). Small microplastics consumed secondarily with their food (e.g. fibres and microbeads) probably are excreted soon after ingestion, but retention times are still undetermined for most species (Ryan, 2015c; Provencher et al., 2018; Bourdages et al., 2021). To better understand what types of plastics are selected by seabirds at sea, we need to compare the characteristics (type, size and colour) of these plastics with what is available in their marine environment, which is typically done by sampling marine plastics using surface net trawls. Plastic sampled with surface nets are mostly < 25 mm in size (Morét-Ferguson et al., 2010; Cózar et al., 2014; Sauria et al., 2023). To determine whether the plastics ingested by seabirds reflect the size of plastics collected with surface nets at sea, we need to compare the typical size ranges of plastics ingested by each taxon with those collected by surface nets. Small buoyant plastics are not frequently ingested by larger albatrosses and giant petrels, as they generally prefer larger items (Chapter 2). Small seabirds may therefore be more appropriate to use as substitutes for net sampling, as they prefer items that are more comparable in type and size to those typically sampled with surface nets (Cózar et al., 2014; Roman et al., 2019b). However, to test their utility in this regard, we first need to assess for which types of plastics seabirds select, which in turn allows us to interpret their preferences in relation to what is available at sea. Such information can help us better evaluate the use of petrels as holistic biomonitors of temporal and spatial heterogeneity in environmental plastics (Vlietstra and Parga, 2002; Kain et al., 2016; Hidalgo-Ruz et al., 2021; Lavers et al., 2021; Shugart et al., 2023).

The frequency of occurrence (FO) of ingested plastic varies among seabird taxa (Ryan, 1987a; Kühn and van Franeker, 2020), and is influenced by factors including foraging strategies (e.g. surface seizing versus plunge diving; Ryan, 1987a; Poon et al., 2017; Roman et al., 2019a), body size (Ryan, 1987a; Roman et al., 2019b; Chapter 2), age and breeding status (Ryan, 1988b; Ryan, 2016; Tulatz et al., 2023), foraging area (van Franeker and Law, 2015; Clark et al., 2023) and retention and egestion rates (Ryan, 2015c). In addition, the sampling method (e.g. sampling from carcasses, inducing emesis or from prey remains) as well as the cause of death of dead birds could influence the number and size of plastics collected (Ryan, 1987b; Ryan and Fraser, 1988; Rodríguez et al., 2018; Lavers et al., 2021). In order to use seabirds as indicators of marine plastics, we need to understand how these factors influence the amounts and types of plastics retained in seabirds (Ryan, 2016). Plastics ingested by seabirds are collected either directly from their stomachs through dissection (e.g. Ryan, 1987a; van Franeker et al., 2011; Robuck et al., 2022), or by inducing emesis (Bond and Lavers, 2013).

Less invasive techniques include collecting regurgitated items at nests (Phillips and Waluda, 2020; Bond et al., 2021; Chapter 2) or take advantage of regurgitations of predators that include the indigestible remains of prey, including their ingested plastic (Furtado et al., 2016; Acampora et al., 2017; Diaz-Santibañez et al., 2023). Predatory seabirds like the large skuas (*Catharacta*) frequently eat seabirds at their breeding sites. Their regurgitations have been used to report plastic loads in their seabird prey (e.g. Ryan, 1987a; Ryan and Fraser, 1988; Hammer et al., 2016; Ibañez et al., 2020; Lenzi et al., 2022) and to assess temporal variation in plastic amounts and composition in multiple seabird taxa (Ryan, 2008). However, to assess how well regurgitated skua pellets represent what is ingested by their prey, a comparison between the types of plastics collected directly from prey carcasses (stomachs) to those sampled from pellets is still needed.

In this chapter, I compare the plastics ingested by petrels breeding on Inaccessible Island in the central South Atlantic Ocean with those available within their marine environment. I use regurgitated Brown Skua pellets collected in 2018 to sample plastics ingested by four petrel taxa: *Fregetta* storm petrels (both *Fregetta grallaria* and *Fregetta tropica*), White-faced Storm Petrels *Pelagodroma marina*, Broad-billed Prions *Pachyptila vittata* and Great Shearwaters *Ardenna gravis* (Fig. 1.7), and compare these to floating plastics collected with a fine-meshed surface net (Fig. 1.4) within the seabirds' foraging ranges between 2016 and 2019. I assess if skua pellets bias against certain plastics, especially smaller pieces that could be excreted by skuas or go undetected in their pellets. I then assess the characteristics of plastics ingested by the four petrel taxa, and compare them to the plastic items available in the marine environment. Finally, I evaluate the suitability of using skua pellets containing the remains of petrels as biomonitors of small buoyant plastics at sea.

Methods

Buoyant plastics at sea

Plastics were collected from the sea surface during nine oceanographic research voyages from December 2016 to November 2019 in the South Atlantic and southwest Indian Ocean (Fig. 3.1; Table S3.1). All voyages were aboard the R.V. *S.A. Agulhas II*, except the Antarctic Circumnavigation Expedition (ACE), which was on the R.V. *Akademik Tryoshnikov.* Floating plastics were collected with surface net tows following a standardised sampling procedure across all voyages and stations (Suaria et al., 2020b; Suaria et al., 2023; Weideman et al., 2023). A neuston net (Aquatic BioTechnology) with a rectangular frame (0.8 m wide x 0.2 m high) and a 2.5 m long 200 μm nylon mesh net (Fig. 1.4) was used to collect all samples. The net was lowered from the starboard side of the foredeck onto the surface of the water using a long-armed crane and positioned approximately 15 m away from the side of the ship, beyond the ship's bow wave. As soon as the net was in position, the start coordinates were recorded using a GPS (Garmin eTRex 20, USA). The net was towed at ~2 knots for 15 min, after which it was lifted out of the water and the end coordinates recorded. The area of sea surface sampled (km^2) was calculated as the width of net (0.8 m) x distance towed (estimated from the two GPS positions). After retrieval, the net was rinsed down from the outside using freshwater to ensure that all material was collected in the cod end. The contents of the cod-end were frozen in glass jars until processing. During processing, I defrosted the samples and then carefully sorted through them under a bright light to remove all visible anthropogenic litter ~ ≥ 1 mm. I stored all litter items in glass vials or aluminium foil packets until further processing.

In the laboratory, items were counted and classified into five categories: industrial pellets, and four types of user plastics: hard fragments, flexible plastics (e.g. food packaging, plastic bags), thread-like plastics (from rope, netting or fishing line) and foamed plastics (e.g. expanded polystyrene, polyurethane and other foamed plastics), and assigned to eight colour groups (white/clear, black, orange/brown, green, blue/purple, red/pink, grey/silver and yellow) following Provencher et al. (2017). For samples collected on the Gough Island Relief, SEAmester III and IV, SCALE WINTER and SPRING and Marion Island relief voyages (Table S3.1), polymer types were determined by ATR-FTIR using an iD7 ATR Nicolet iS5 (ThermoScientific) instrument under absorbance mode with spectral region from 400 to 4000 cm^{-1} and at a resolution of 4 cm^{-1}. OMNIC Spectra Software with Hummel Polymer- and HR Nicolet Sampler Libraries were used for automated identification of polymers. Samples collected on the ACE voyage, Marion Island relief 2017 and SEAmester II were characterized using a LUMOS stand-alone FTIR microscope (Bruker Optik GmbH). ATR spectra were recorded by averaging 64 scans per particle with a spectral resolution of 4 cm^{-1} (range 650–4000 cm^{-1}). Spectra were processed and analysed using the OPUS 7.5 software (Bruker) and polymer identification was performed by comparison with commercially available libraries and an additional custom library compiled within the framework of the JPI-Oceans project BASEMAN (Primpke et al., 2018). Polymer identities were assigned if there was ≥ 70% match. The maximum length of items (flexible items and thread-like plastics were straightened) was measured to the nearest 0.01 mm either with digital callipers or from digital images. I classified items into three commonly used size classes for marine litter (GESAMP, 2019) based on their maximum length: macroplastics (> 25 mm), mesoplastics (5–25 mm) and microplastics (< 5 mm). All items were weighed on a precision electronic balance to the nearest 0.1 mg.

Plastics ingested by seabirds

Plastic items ingested by seabirds were collected from regurgitated Brown Skua pellets containing the undigested prey remains (bones, feathers, etc.) of burrowing seabirds at Inaccessible Island, Tristan da Cunha, central South Atlantic Ocean. Pellets were collected from 13 September to 24 November 2018 at a large skua club at the West Point of the island, where 50–100 non-breeding skuas congregate (Ryan and Moloney, 1991; Ryan, 2008; Ryan, 2023c). Prior to processing, each pellet was air dried and then broken apart in a sorting tray to identify the prey remains. Bird prey species were identified from their bones and/or feathers, in comparison with reference sets of bones from known carcasses. Until recently, it was assumed that only White-bellied Storm Petrels *Fregetta grallaria* breed at Inaccessible Island, but genetic evidence shows that both *F. grallaria* and a White-bellied form of the Black-bellied Storm Petrel *F. tropica* breed on the island (Robertson et al., 2016). The two species are treated together here because their remains in skua pellets cannot be discriminated reliably (Ryan, 2023c). All pellets were processed on the island by the same experienced observer to reduce observer bias.

Storm petrels are usually swallowed whole, making it easy to attribute the plastics to a particular prey species. However, some pellets contained the remains of more than one bird (Ryan and Moloney, 1991; Ryan, 2023c); such pellets were discarded from this study. Pellets from prey that are too large to be swallowed whole, such as Broad-billed Prions and Great Shearwaters, sometimes result in multiple pellets per prey item (e.g. the legs in one pellet, head in another, and balls of feathers with no bony remains at all). Pellets containing bony remains were counted as representing a bird as long as they contained some additional material (i.e. they were classed as pellets rather than loose skulls or leg bones). Pellets comprised solely of feathers were not counted unless they contained indigestible stomach contents (mainly cephalopod beaks and plastics, but also pumice, seeds, fish otoliths and fish and/or cephalopod eye lenses). Pellets containing the remains of burrowing seabird chicks were readily identified by their poorly-developed bones. However, this made it hard to identify the prey species, and to tell whether a pellet contained the remains of more than one chick. Chick pellets were assumed to be from Broad-billed Prions because they are by far the most common petrel species with chicks during September to November (Ryan, 2007; Fig. S3.1). Broad-billed Prions were the only taxon to include chicks in the analyses. All plastic items ≥ 1 mm from each pellet containing the remains of only one avian prey item were stored in a labelled Ziploc bag and brought back to the University of Cape Town, South Africa. To assess if skua pellets accurately reflect plastic loads or fail to include the smaller items ingested by their seabird prey, intact carcasses of burrowing seabirds found during routine surveys were opportunistically collected and all plastic items were removed from the proventriculus and ventriculus (hereafter referred to as stomachs). Some of these birds had been predated (but not yet eaten) by skuas or Tristan Thrushes *Turdus eremita*, whereas the cause of death of others was not known. Plastics in these birds were processed in the same manner as those collected from skua pellets.

In the laboratory, I sorted and classified plastics into the same categories and colour groups as plastics collected at sea (and in Chapter 2). Due to the large number of ingested plastic items, I subsampled items for polymer analyses. As industrial pellets and hard fragments were the most numerous items ingested by all four taxa, I selected 100 industrial pellets and 100 fragments from each taxon for polymer analyses. To avoid selection bias, I blindly removed a single bag (representing an individual bird) from a bag containing all the samples collected (per taxon) within a specified month (starting with the first sample month, September). All hard fragments and industrial pellets within each bag were processed, until 100 of each were sampled per taxon. For *Fregetta* storm petrels, only 20 industrial pellets were ingested over the study period, and of these, only 18 delivered a polymer match (Table S3.4). Fewer flexible items, thread-like plastics and foamed pieces were collected, so all of these items were analysed using FTIR. Polymer types were determined with a Bruker Alpha II compact FTIR spectrometer. Samples were cleaned with 70% ethanol (to remove any residues) prior to analysis. I used absorbance mode with spectral region from 400 to 4000 cm^{-1} at a resolution of 4 cm^{-1} and 32 scans. OPUS 8.7 Spectra Software with Bruker Optics ATR-Polymer Library and the KIMW ATR-IR Polymer Library were used for automated identification and assigned a polymer if there was a ≥ 70% match. I also measured the length (longest dimension) of each item to the nearest 0.01 mm using digital calipers and flexible and thread-like plastics were straightened to record their maximum length. I also classified items into three commonly used size classes for ingested plastics (Provencher et al., 2019) as done for marine plastics. All items were weighed; most to the nearest 1 mg, but samples weighing < 2 mg were weighed to the nearest 0.1 mg.

Data analysis

The density of marine buoyant plastics was calculated per sampling station, and overall, as the number and mass (g) of items·km^{-2}. A general at-sea distribution map of the Atlantic Ocean ranges for each of the four bird taxa was produced to illustrate the potential foraging areas of each taxon by collating information from various resources (Harrison, 1983; Ryan, 2007; Ronconi et al., 2018; Schoombie et al., 2018; Jones et al., 2020; Robuck et al., 2022; Ryan and Oppel, 2022; Ryan, 2023b; Atlas of Seabirds at Sea, www.seabirds.saeon.ac.za).

To determine if all plastics ingested by seabirds were detected in skua pellets, I compared the size range of plastics from carcasses to plastics in skua pellets, using Mann-Whitney U tests and violin plots. To increase the sample size of Great Shearwater carcasses, I augmented the dataset with data from adult Great Shearwater carcasses (n = 15) killed on fishing gear in the central South Atlantic during March–April 2018 (Robuck et al., 2022). I also compared the mean mass and length of items collected in carcasses in other studies to my study. All data were assessed for normality using Shapiro-Wilks normality tests.

Chi-squared tests of independence were used to compare the ratios of plastic types (categories), colour groups and polymer types between plastic collected at sea and ingested by seabirds. Groups with few observations were pooled to ensure that no more that 20% of expected counts were < 5. I used correspondence analysis and a contribution biplot to visualize differences in the types and colour of plastics (rows) and seabirds/marine (columns) using the *"FactoMineR"* and "*factoextra*" packages in R (Lê et al., 2008; Kassambara and Mundt, 2020; R Core Team, 2023). The relative distance between types (or colour groups) of plastics ingested by seabirds (or collected at sea) tells us how characteristic that type (or colour group) is for that seabird (or marine plastics). The more acute the angle of the arrow between rows and columns, the stronger the relationship. To determine the degree of individual variation in plastic types and if the types of items ingested varied among individuals, I performed a correspondence analysis for individuals that had ingested ≥ 20 items.

I report the proportion of hard fragments and industrial pellets that were within each commonly recorded size category (micro-, meso- and macroplastics) for marine and ingested plastics (per taxon and overall). I used violin plots of mass and length (log transformed to improve visualization) to present the length and mass data among seabirds and marine plastics. Because the mass and size of ingested plastic items are generally right skewed, I report the median (and range) for mass (mg) and length (mm) of plastics overall, and within each category, unless stated otherwise. Means (± SD) are reported in the supplementary materials (Tables S3.2 and S3.3). Differences in the mass and length of plastics among seabirds and between ingested and marine plastics were assessed with Kruskal–Wallis (KW) tests, and post-hoc Dunn's tests (with Bonferroni corrections to reduce the chance of committing Type 1 errors) were used to assess which groups differed. I tested for differences in the mass and length of items between Broad-billed Prion adults and chicks with Mann-Whitney U tests. Results were considered significant where $P < 0.05$. All data analyses were performed in R version 4.2.3 (R Core Team, 2023).

Results

Floating plastics at sea

I collected 392 plastic items during 116 surface net tows that sampled a total of 0.09 km^2 of sea surface (Fig. 3.1; Table S3.1). The average density was 5 025 ± 9 001 (SD) plastic items·km^{-2} and 60 ± 154 g·km^{-2} (Table S3.1). Most marine plastics were hard fragments (92%), with small proportions of flexible pieces, industrial pellets, thread-like and foamed plastics (Table 3.1). Hard fragments occurred widely in the South Atlantic and southwest Indian Ocean (Fig. S3.2A), but all industrial pellets were found in the South Atlantic (Fig. S3.2A), and 75% of foamed and 50% of flexible items were sampled within ~200 km of the coast (Fig. S3.2B). Marine plastics were mostly white/clear, blue/purple, or black, whereas other colours (orange/brown, yellow, etc.) were seldom found (Table 3.2). Polymer type was assigned to 93% of items; 4% could not be identified due to poor spectral matches and 3% of items were either too small to analyse or crushed/lost during FTIR processing. Most hard fragments (72%), industrial pellets (100%), flexible pieces (70%) and thread-like plastics (57%) were made of PE (Table S3.5). PP was the next most common polymer (25% of hard fragments; 30% of flexible pieces; 29% of threads). PS, polypropylene/ethylene-propylene-diene monomer (PP/EPDM), PVC, polymethyl methacrylate (PMMA) and ethylene-propylene rubber (EPM), contributed 3% overall (Table S3.5). The few foamed items were composed of PS (75%) and PP/EPDM (25%). Marine plastics had a median mass of 3 mg (range 0.1–624.0 mg; Table 3.3) and length of 3.1 mm (1–435 mm; Table 3.4) but half of all of marine plastic items weighed ≤ 2 mg and a third were ≤ 2 mm in length. Most marine plastics were microplastics (78%) followed by mesoplastics (21%) and macroplastics (1%; Table S3.7).

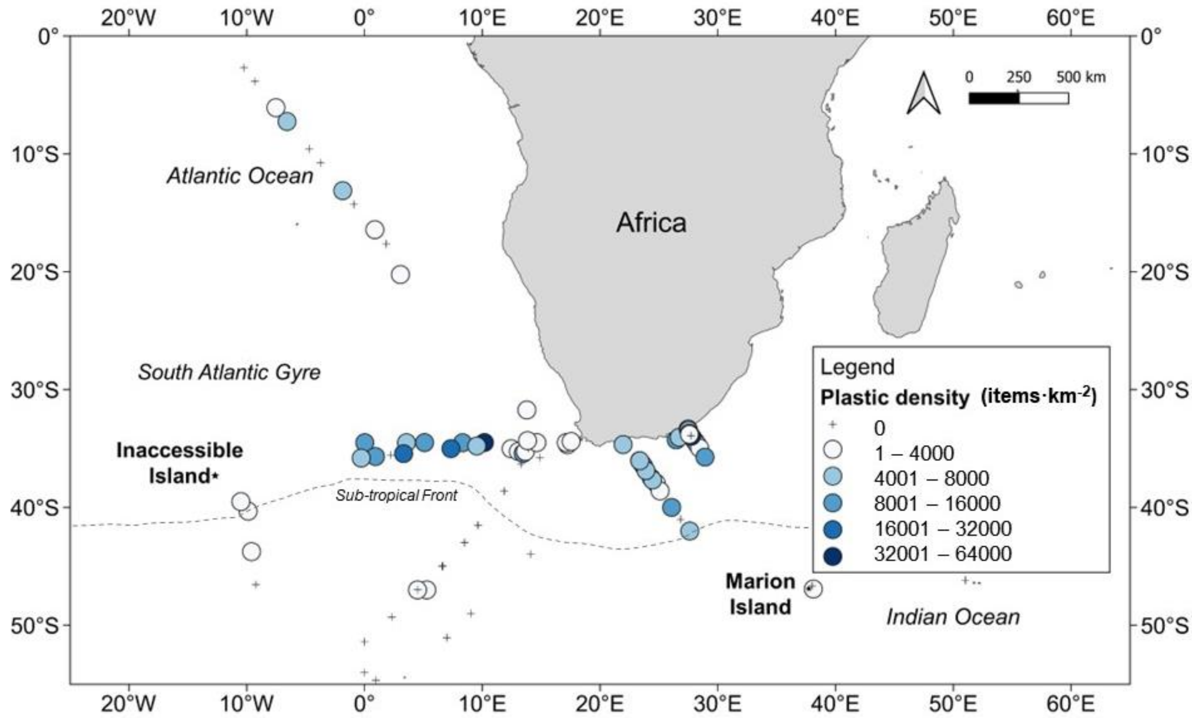


Fig. 3.1 Density of plastic (items·km^{-2}) in surface net trawls in the South Atlantic and southwest Indian Ocean relative to Inaccessible and sub-Antarctic Marion Island.

Plastics ingested by seabirds

In total, 5 310 plastic items in 569 skua pellets containing the remains of 103 *Fregetta* storm petrels (n = 210 plastic items), 147 White-faced Storm Petrels (767), 207 Broad-billed Prions (99 adults [406], 108 chicks [1347]), and 112 Great Shearwaters (2 580) were collected (Table 3.1). The frequency of occurrence of plastic ingestion varied among taxa, the most frequent being Broad-billed Prion chicks (87%, although this might be inflated by multiple chicks being sampled in some pellets), followed by White-faced Storm Petrels (75%), Great Shearwaters (73%), Broad-billed Prion adults (62%) and *Fregetta* storm petrels (38%) (Table 3.1). The median and interquartile masses of plastic items collected from Great Shearwater carcasses were almost identical to those collected from skua pellets (Fig. 3.2A), but there were fewer very small items in skua pellets, resulting in a significantly lower mass per item in carcasses ($Z = -5.47$, $P < 0.001$). The same pattern occurred in White-faced Storm Petrels, with lower mass items recorded from carcasses ($Z = -4.14$, $P < 0.001$; Fig. 3.2C). However, there was no significant difference between the length of items recorded in carcasses and skua pellets in either Great Shearwaters ($Z = -0.51$, $P = 0.61$; Fig. 3.2B) or White-faced Storm Petrels (Z =

0.47, P = 0.64; Fig. 3.2D). When compared to plastics collected from carcasses in other studies, skua pellet plastics fell well within the reported size ranges (Table S3.6). Instances where mean mass and length differed between plastics collected from skua pellets in my study, compared to stomach plastics, could be explained by the modest sample sizes from stomach samples (Table S3.6).

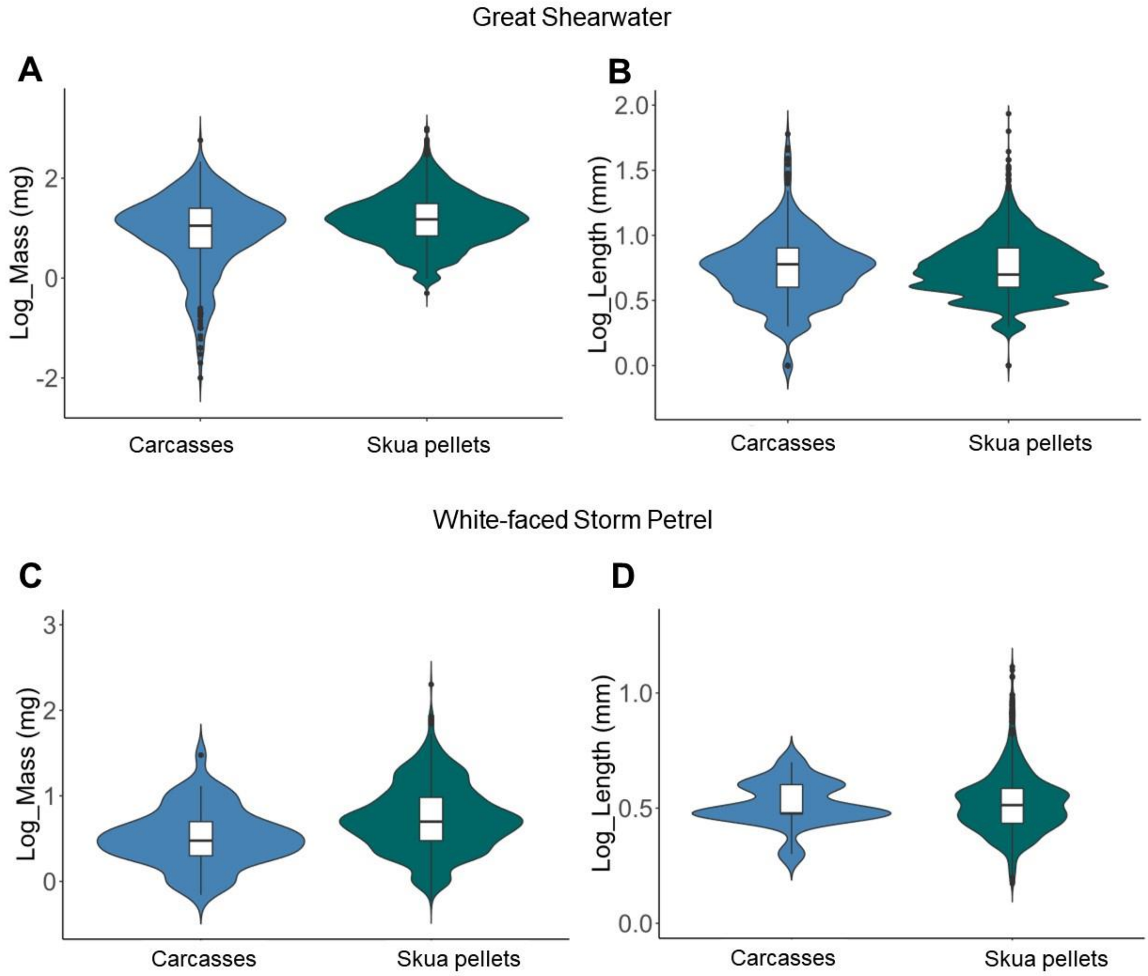


Fig. 3.2 Plots comparing the log transformed masses (mg) (A and C) and lengths (mm) (B and D) of plastic items recorded in carcasses and in skua pellets for Great Shearwaters (n = 17 carcasses (433 items), n = 112 skua pellets (2 580 items) (A and B)) and White-faced Storm Petrels (n = 2 carcasses (54 items), n = 147 skua pellets (767 items) (C and D)).The boxplot indicates the median (dark line) and interquartile range while the violin plot indicates the kernel probability density at different values where the width of each curve corresponds to the frequency of data points in that region.

Most ingested plastic items from skua pellets were hard fragments (88%) followed by industrial pellets (10%) and flexible pieces; thread-like and foamed plastics each contributed ≤ 1% (Table 3.1). Among adult seabirds, the proportions of categories of ingested plastics differed (χ^2 = 186.39, df = 6, P < 0.001; flexible pieces, thread-like plastics and foam pooled). *Fregetta* storm petrels ingested more thread-like plastics (9%) than any other species (all < 2%) and White-faced Storm Petrels consumed the greatest proportion of industrial pellets (20%) (Table 3.1). Broad-billed Prion adults and chicks did not differ in the proportions of ingested plastic categories (χ^2 = 5.05, df = 1, P = 0.08; Table 3.1). Great Shearwaters were the only species that contained foamed plastics and they ingested slightly more flexible items than other taxa (Table 3.1). These differences were reflected in the correspondence analysis (Fig. 3.3).

The most frequently ingested colour groups overall were white/clear (61%), followed by blue/purple (11%) and orange/brown (9%) (Table 3.2). The ratios of colour groups differed among seabirds (χ^2 = 165.86, df = 24, P < 0.001; yellow and grey/silver pooled), because *Fregetta* storm petrels, Broad-billed Prions and Great Shearwaters ingested more blue/purple and green items while White-faced Storm Petrels ingested greater proportions of red/pink and yellow items (Table 3.2; Fig. 3.4).

A total of 797 items (93%) sub-sampled from skua pellets delivered a polymer match (Tables S3.2 and S3.3). There was no significant difference in the ratios of polymer types among seabird species (χ^2 = 5.87, df = 3, P = 0.14; PP and other polymers pooled) and most hard fragments (7%), industrial pellets (83%) and thread-like plastics (71%) were composed of PE, followed by PP (Table S3.5). Most flexible items were composed of PP (62%; Table S3.5). PS was only recorded in industrial pellets, PP/EPDM in industrial pellets and thread-like plastics, and PU was only recorded in foamed plastics, and combined these polymers only accounted for 1% of polymer types overall (Table S3.5).

The size of items differed among adult seabird taxa (mass: KW χ^2 = 529.8, df = 3, P < 0.001; Table 3.3, Fig. 3.5A; length: KW χ^2 = 809.6, df = 3, P < 0.001; Table 3.4; Fig. 3.5B), with the two storm petrels ingesting smaller items than prions and shearwaters (Table S3.7). There was no significant difference in the mass of ingested items between the two storm petrel taxa (Z = 2.62, P = 0.05; Fig. 3.5A), but White-faced Storm Petrels ingested slightly smaller items (median = 3.3 mm, range 1.5–31.8 mm) than *Fregetta* storm petrels (3.6 mm, 1.3–104 mm; Z = 5.16, P < 0.001; Table 3.4; Fig. 3.5B), possibly due to their slightly smaller size compared to *Fregetta* storm petrels (Fig. S3.1). Adult Broad-billed Prions and Great Shearwaters did not show significant differences in the mass or length of ingested items (mass: Z = 2.17, P = 0.1; Fig. 3.5A; length: Z = -1.46, P = 0.90; Fig. 3.5B), despite their size differences (Fig. S3.1). Broad-billed Prion chicks contained slightly larger items than adults in terms of both mass (adult median = 17 mg, range 0.3–190 mg, chicks = 21 mg, range 1–719 mg; Z = -3.77, P < 0.001) and length (adults = 5.2 (1.6–99.9) mm, chicks = 5.8 (1.3–21.4) mm; Z = -4.05, P < 0.001; Tables 3.2 and 3.3). Most hard items and industrial pellets recovered from skua pellets among all seabirds were microplastics (56%) followed my mesoplastics (44%) while no macroplastics were recorded (Table S3.7).

Some variation in the types of plastic ingested between individuals of the same species was recorded, but in general, most individuals (where ≥ 20 items were recorded) ingested similar proportions of items (Fig. S3.3). For example, White-faced Storm petrels typically ingested larger proportions of industrial pellets than other taxa, and 80% of birds that had ingested ≥ 20 items (n = 5) ingested 24–38% industrial pellets (Fig. S3.3). Broad-billed Prions (≥ 20 items; n = 1 adult and 19 chicks) mostly ingested hard fragments and industrial pellets, and only three individuals included thread-like plastics (Fig. S3.3). There are a couple of considerations when using chicks for this analysis. One is the potential presence of multiple chicks in a single pellet, and the other is that they are likely fed plastics by both parents, which hinders individual comparisons. Most Great Shearwaters were similar in the types of plastics ingested, but a few individuals contained more thread-like plastics, foam and flexible pieces (Fig. S3.3). *Fregetta* storm petrels (which ingested more thread-like plastics than any other taxa) did not have any individuals containing > 20 items, but across all individuals, 16% ingested 1–2 thread-like plastics. These results indicate that among individual variation was not large, and that trends observed within taxa are characteristics of the taxon overall.

Table 3.1 Composition of floating plastic in surface net tows in the South Atlantic and southwest Indian Ocean (marine plastics) and the numbers of birds, plastic items, the proportions of plastic categories and the overall frequency of occurrence (% FO) of plastic ingested by four seabird taxa sampled in Brown Skua pellets on Inaccessible Island. Chick samples were only included for Broad-billed Prions.

Source	**n tows or birds (n plastics)**	**% hard fragments**	**% industrial pellets**	**% flexible pieces**	**% thread-like plastics**	**% foamed**	**% FO**
Marine plastics	116 (392)	92%	2%	3%	2%	1%	
All seabirds	569 (5 310)	88%	10%	1%	1%	< 1%	
Fregetta SP*	103 (210)	81%	10%	< 1%	9%	0%	38%
White-faced SP	147 (767)	79%	20%	< 1%	1%	0%	75%
Broad-billed Prion							
adults	99 (406)	87%	11%	1%	< 1%	0%	62%
chicks	108 (1 347)	87%	12%	1%	< 1%	0%	87%
Great Shearwater	112 (2 580)	92%	6%	2%	< 1%	< 1%	73%

*SP = storm petrel

Table 3.2 Proportion (%) of colour groups recorded for plastics collected at sea and those ingested by four seabird taxa. Chi-square test results compare the ratios of ingested items to marine plastics (df = 6, yellow and grey/silver items pooled due to low numbers). Chick samples were only included for Broad-billed Prions.

	white/clear	**blue/purple**	**orange/brown**	**green**	**black**	**red/pink**	**grey/silver**	**yellow**	**Significance**
Marine plastics	71%	11%	1%	4%	8%	2%	2%	1%	
All seabirds	61%	11%	9%	7%	5%	5%	1%	1%	$\chi^2 = 47.11$, $P < 0.001$
Fregetta SP*	55%	14%	5%	10%	5%	6%	4%	1%	$\chi^2 = 31.54$, $P < 0.001$
White-faced SP*	63%	5%	11%	2%	7%	9%	2%	2%	$\chi^2 = 72.95$, $P < 0.001$
Broad-billed Prion									
adult	58%	14%	10%	7%	4%	4%	2%	< 1%	$\chi^2 = 45.41$, $P < 0.001$
chicks	62%	13%	7%	7%	6%	5%	< 1%	< 1%	$\chi^2 = 43.95$, $P < 0.001$
Great Shearwater	61%	11%	9%	9%	5%	4%	1%	1%	$\chi^2 = 49.90$, $P < 0.001$

*SP = storm petrel

Table 3.3 The median (range) mass (mg) of plastic items (all items and per category) collected during surface net tows (marine plastics) and ingested by seabirds breeding on Inaccessible Island in the central South Atlantic Ocean. Significance indicates the post-hoc Dunn's test results comparing the mass of ingested plastics (all items) in adult birds to the mass of marine plastics. Chick samples were only included for Broad-billed Prions.

	All items median (range)	**Hard fragments**	**Industrial pellets**	**Flexible pieces**	**Thread-like**	**Foamed plastics**	**Significance**
Marine plastics	3 (0.1-624)	3 (0.1-624)	12 (1-22)	3 (0.2-26)	2 (0.3-6)	1 (0.1-1)	
All seabirds	14 (0.3-990)	13 (1-990)	20 (2-63)	6 (0.3-59)	2 (1-122)	7 (1-12)	
Fregetta SP*	6 (1-302)	6 (1-302)	18 (6-41)	2	2 (1-24)	–	Z = -5.32, P < 0.001
White-faced SP*	5 (1-201)	4 (1-201)	13 (2-52)	5 (2-8)	2 (1-25)	–	Z = -4.18, P < 0.001
Broad-billed Prion							
adults	17 (0.3-190)	16 (1-190)	23 (5-49)	3 (0.3-23)	6 (1-11)	–	Z = 17.07, P < 0.001
chicks	21 (1-719)	21 (1-719)	23 (3-58)	6 (3-18)	2 (1-3)	–	
Great Shearwater	15 (1-990)	14 (1-990)	21 (6-63)	7 (1-59)	2 (1-122)	7 (1-12)	Z = 20.21, P < 0.001

*SP = storm petrel

Table 3.4 The median (range) length (mm) of marine plastic items (overall and per category) collected during surface net tows in the South Atlantic and southwest Indian Oceans and those ingested by seabirds breeding on Inaccessible Island. Significance indicates the post-hoc Dunn's test results comparing the lengths of ingested plastics (all items) in adult birds to the lengths of marine plastics. Chick samples were only included for Broad-billed Prions.

	All items median (range)	**Hard fragments**	**Industrial pellets**	**Flexible pieces**	**Thread-like**	**Foamed plastics**	**Significance**
Marine plastics	3.1 (1-435)	3.0 (1-435)	3.2 (1-4)	9.4 (2-22)	20.4 (2-8)	3.8 (3-5)	
All seabirds	5.0 (1-104)	5.3 (1-33)	3.8 (2-6)	10.9 (3-44)	20.0 (3-104)	6.9 (5-7)	
Fregetta SP*	3.6 (1-104)	3.5 (1-19)	3.6 (3-5)	6.5	20.7 (3-104)	–	$Z = -2.78, P = 0.05$
White-faced SP*	3.3 (2-32)	3.2 (2-13)	3.5 (2-6)	6.6 (6-8)	7.3 (5-32)	–	$Z = 2.44, P = 0.15$
Broad-billed Prion							
adults	5.2 (2-100)	5.5 (2-17)	3 .8 (2-5)	10.9 (7-31)	56.5 (13-100)	–	$Z = 12.25, P < 0.001$
chicks	5.8 (1-21)	6.2 (1-18)	4.0 (2-6)	6.7 (6-15)	10.4 (1-21)	–	
Great Shearwater	5.5 (1-86)	5.6 (1-33)	3.9 (2-6)	12.2 (3-44)	21.1 (6-86)	6.9 (5-7)	$Z =17.35, P < 0.001$

*SP = storm petrel

Comparison between the characteristics of ingested and marine plastics

Hard fragments were the type of plastic recorded most frequently in both ingested (88%) and marine plastics (92%). However, the proportion of plastic types differed (χ^2 = 35.59, df = 3, P < 0.001; thread-like plastics and foamed pieces pooled) due to the greater proportions of industrial pellets ingested by seabirds (10%) than sampled with nets (2%; Table 3.1). Great Shearwaters ingested plastics in proportions closest to the spectrum available in the environment because they contained more flexible pieces than other species and were the only species to contain foamed plastics (Fig. 3.3).

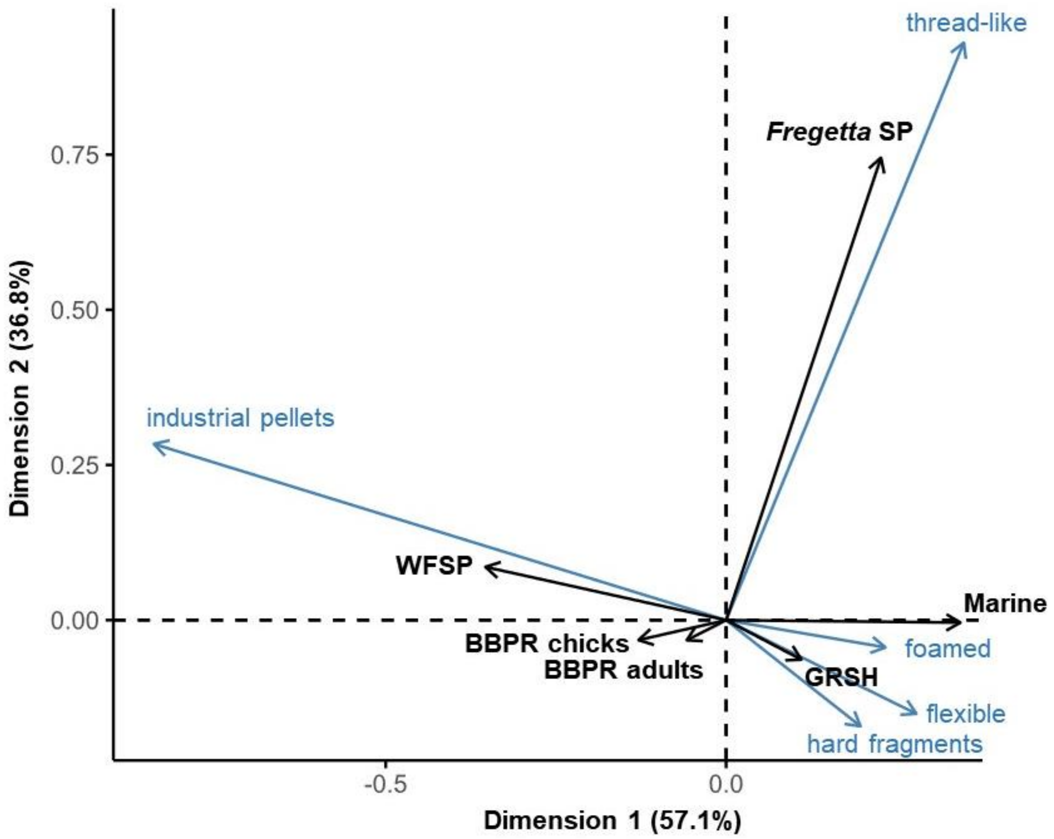


Fig. 3.3 Correspondence analysis indicating the relationships between the composition of plastic types sampled at sea (marine) and ingested by seabirds: SP = storm petrel, WFSP = White-faced Storm Petrel, BBPR = Broad-billed Prion, GRSH = Great Shearwater.

White/clear and blue/purple items were recorded more frequently than other colours in both ingested (61% and 11%) and marine (71% and 11%) plastics, but the overall proportions of colour groups differed, because seabirds ingested more red/pink, orange/brown and green items, and less white/clear/silver/grey/black items than found on the environment (Table 3.2; Fig. 3.4).

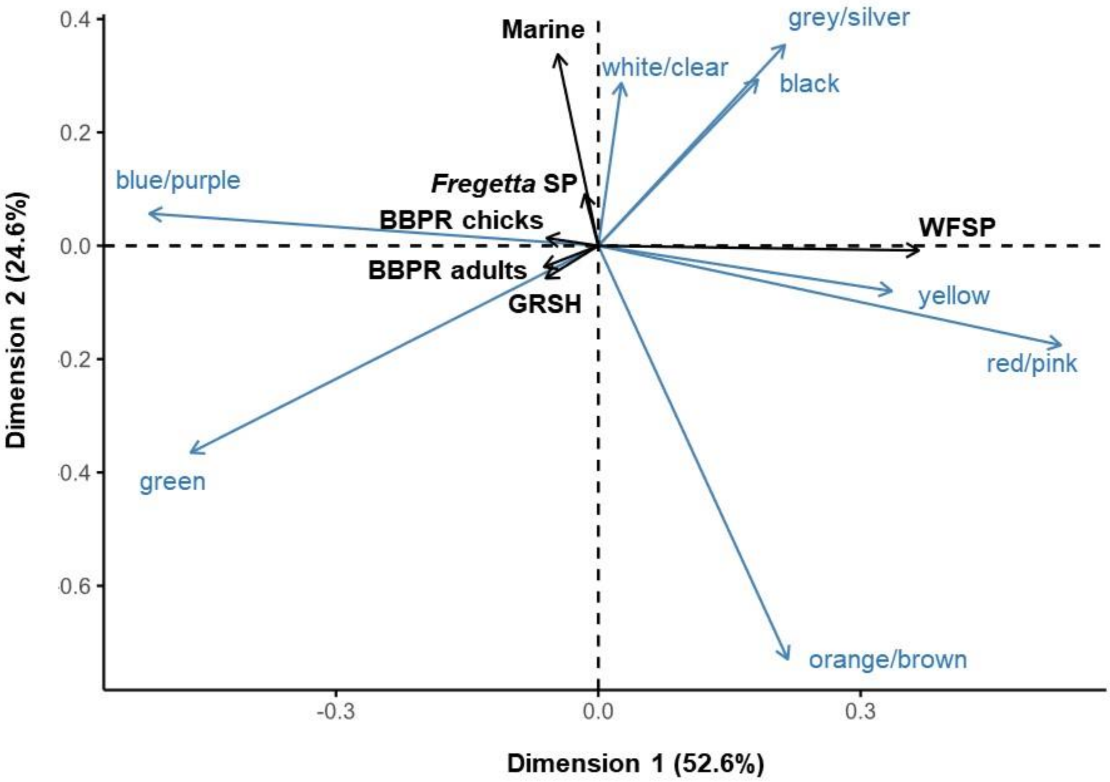


Fig. 3.4 Correspondence analysis indicating the relative relationship between the colour groups of plastics sampled at sea (marine) and ingested by seabirds: SP = storm petrel, WFSP = White-faced Storm Petrel, BBPR = Broad-billed Prion, GRSH = Great Shearwater.

The size of plastic items differed significantly between marine and ingested plastics across all adult seabirds (mass: KW $\chi^2 = 789.6$, df = 4, P < 0.001; Table 3.3; length: KW $\chi^2 = 927.9$, df = 4, P < 0.001; Tables 3.4 and S3.5; Fig. 3.5). Marine plastics generally were substantially lighter (3 [0.1–624] mg vs 14 [0.3–990] mg) and smaller, averaging barely half the length (3.1 [0.5–435] mm vs 5.0 [1.3–103.5] mm) of all ingested plastics (Tables 3.3 and 3.4; Fig. 3.5). However, differences in length were not marked between marine plastics and those ingested by White-faced Storm Petrels (3.3 [2–32] mm; Z = 2.44, P = 0.15; Table 2.4), but showed a trend toward significance when compared to *Fregetta* storm petrels (3.6 [1–104] mm; Z = -2.78, P = 0.05; Table 2.4). Storm petrels ingested similar proportion of microplastics when compared to marine plastics, whereas the larger prions and shearwaters ingested more mesoplastics (Table S3.7).

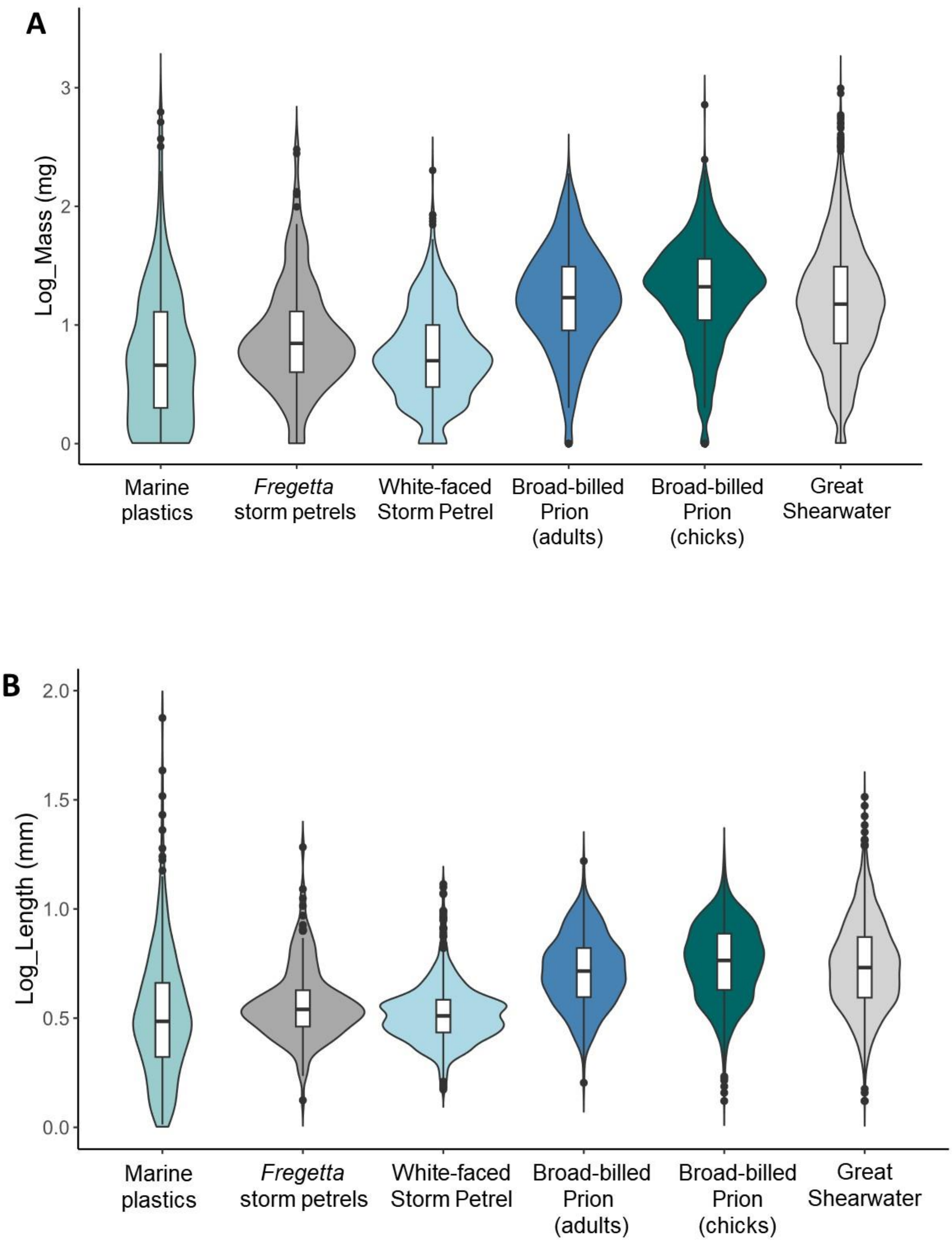


Fig. 3.5 Plots of the log transformed masses (A) and lengths (B) of hard fragments and industrial pellets collected at sea (marine plastics) and ingested by four seabird taxa based on skua pellets. Boxplot conventions as in Fig. 3.2.

PE was the most frequently recorded polymer type in both ingested (78%) and marine (72%) plastics (Fig. 3.6). Polymer types of ingested plastics (all seabirds combined) differed from the marine environment ($\chi^2 = 9.49$, df = 2, $P < 0.05$; PS, PP/EPDM, PVC, PMMA, EPM and PU pooled), mostly because PVC, PMMA and EMP were only recorded in marine samples, and

PU only recorded in ingested plastics (Table S3.5). However, polymer types ingested by Broad-billed Prions and Great Shearwaters did not differ significantly from those collected at sea (Table S3.5). Seven of the eight polymer types recorded were found in marine plastics, whereas only five were found in ingested plastics, despite having nearly double the sample size for ingested plastics. Polymers denser than seawater (PVC and PMMA) were only recorded in marine samples, with the exception of one pellet composed of PS (unexpanded) from a Great Shearwater (Table S3.5). When only comparing hard fragments, polymer type did not differ significantly between ingested and marine plastics ($\chi^2 = 3.27$, df = 1, P = 0.07) and PE remained the most frequently recorded polymer (72% marine; and 78% ingested), followed by PP (25% marine; 21% ingested; Fig. 3.6).

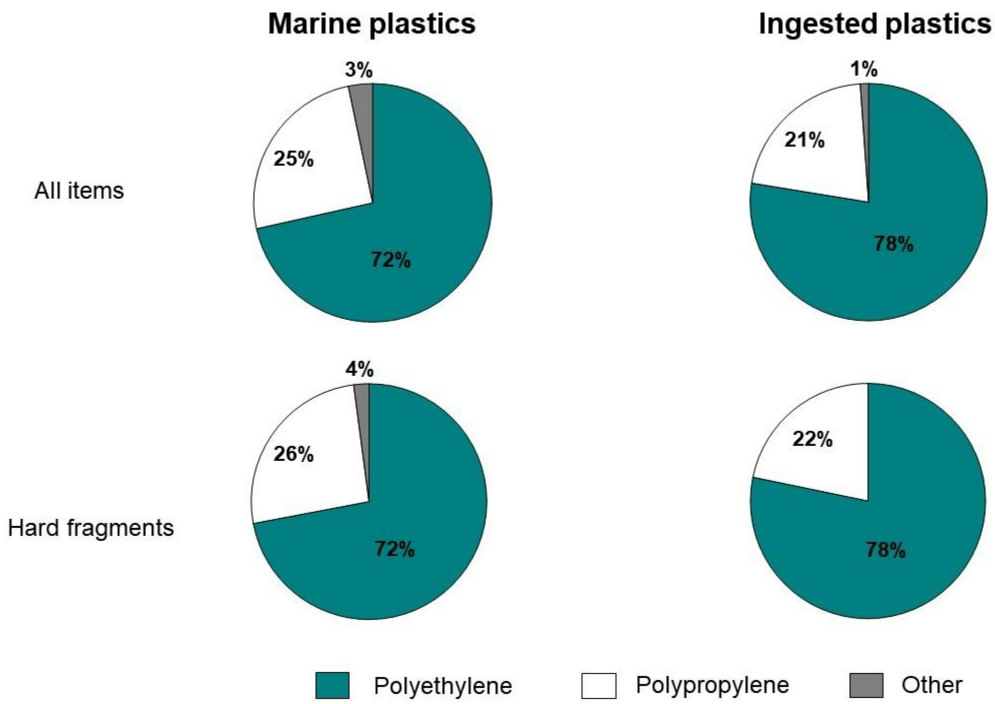


Fig. 3.6 Proportions of polymers recorded for marine and ingested plastics. Other = polystyrene, polypropylene/ethylene-propylene-diene monomer, polyvinyl chloride, polymethyl methacrylate, ethylene propylene rubber and polyurethane.

Discussion

What types of marine plastics are available to South Atlantic seabirds?

Small buoyant plastics are abundant and often found floating on the sea surface and stranded on beaches, and are primarily composed of hard fragments that originate from the fragmentation of larger user items into increasingly smaller pieces (Law et al., 2010; Cózar et al., 2014; Eriksen et al., 2014; Andrady, 2015). Surface net tows are used to estimate their densities at sea (Morét-Ferguson et al., 2010; Law et al., 2014; Courtene-Jones et al., 2021; Hänninen et al., 2021) but mostly sample items < 25 mm in size (e.g. Cózar et al., 2014). While numerous studies have focused on the Northern Hemisphere, only a few have examined plastic densities in the South Atlantic and southwestern Indian Ocean (Morris, 1980; Ryan, 1988a; Eriksen et al., 2014; da Rocha et al., 2021; Zhao et al., 2022; Suaria et al., 2023). For this chapter, I sampled this area with 116 surface net tows over a period of three years. The most commonly sampled plastic type was white/clear hard PE fragments in the microplastics size range, affirming that these are currently the items most likely to be encountered by petrels foraging in this marine environment. The abundance of white/clear items at sea could be related to the process of photo-oxidation which causes discoloration of plastics, potentially explaining the greater numbers of white plastics, which appear to increase with distance from land (Martí et al., 2020). During the late 1970s, industrial pellets dominated floating plastics in the South Atlantic (Morris, 1980; Ryan, 1988a), compared to only 2% in this study. The proportions of hard fragments were considerably lower (0–27%) compared to now (92%). This increase in the proportions of plastic fragments relative to industrial pellets floating at sea has been noted globally in both net samples (Law et al., 2010; Morét-Ferguson et al., 2010; van Franeker and Law, 2015) and in seabirds (Vlietstra and Parga, 2002; Ryan, 2008). The implementation of awareness campaigns in the early 1990s, (e.g. Operation Clean Sweep, www.opcleansweep.org) likely contributed to a decline in pellet loss and, consequently, reduced densities at sea. Flexible items such as pieces of plastic bags were less common, because they are usually found at lower densities and in closer proximity to source areas (e.g. coastlines). This is because the higher surface area to volume ratios of these items leads to increased biofouling rates, impacting their buoyancy, dispersal ability and the probability of being sampled with a surface net at sea, as they are removed at a faster rate from the sea surface compared to items with lower surface area to volume ratios (Ryan, 2015b; Fazey and Ryan, 2016; Naidoo and Glassom, 2019; Ryan, 2020; Maclean et al., 2021; Weideman et al., 2023). Foamed items also were scarce, possibly because their high buoyancy promotes stranding for items deriving from land-based sources shortly after entering the marine environment (Maclean et al., 2021). In my study, flexible and foamed items collected at sea were mostly sampled close to coastal areas, and the few items collected far from land may have originated from ships (Fig. S3.2B), as there has been a marked increase in litter inputs from ships crossing the South Atlantic (Ryan et. al., 2019).

Do skua pellets underestimate plastic loads in seabirds?

Sampling ingested plastics from seabird stomachs ensures that all plastic items are removed, whereas skua pellets may underestimate the number and size of plastics as smaller pieces may fail to be incorporated into the pellet, or be lost before or during collection (Ryan and Fraser, 1988). Ryan and Fraser (1988) found that compared to stomach samples, plastic items < 8 mg were underrepresented in skua pellets, but this was due to observer bias while processing pellets (Ryan, 2008), and careful processing retains items as small as 1 mm and weighing as little as 1 mg (this chapter). Although some smaller plastic items could pass through the skuas' intestines and be excreted in their guano, and thus go undetected (Ryan and Fraser, 1988; Ryan, 2016; Provencher et al., 2018), hard items > 1 mm are seldom found in skua faeces (Ryan, 1987a), suggesting that it is unlikely to influence the results. In my study, skua pellets included plastics ≤ 1 mg and ≤ 2 mm, and generally exhibited overall similar size ranges to those found in the stomachs of Great Shearwaters and White-faced Storm Petrels. Even though the mass of items collected from carcasses were smaller than those from skua pellets, lengths did not differ. This suggests that the risk of overlooking or losing small items is negligible, and that skua pellets are a reliable tool to sample plastics stored in the stomachs of their seabird prey. Storm petrels provide the best measure of plastic loads in the seabirds studied, because they are swallowed whole. Pellets containing the remains of Great Shearwaters and Broad-billed Prions might underestimate plastic loads because the plastics from one bird might be spread over several pellets. However, the stomach and gizzard contents are likely to be ingested together and thus be regurgitated in the same pellet. The risk of counting multiple pellets from the same prey item was reduced by only counting pellets containing cephalopod beaks and other indigestible prey typically found in petrel stomachs.

Interspecific variation in ingested plastics

It is assumed that the likelihood of seabirds ingesting plastics is largely determined by the amount, types and sizes of plastic litter available within their foraging areas, where higher surface plastic densities could result in a higher risk of ingestion (Clark et al., 2023).

Furthermore, seabirds are more likely to ingest plastics that resemble food items, or lighter items that are more conspicuous, and thus are more readily detected at the sea surface (Ryan, 1987a; Shaw and Day, 1994; Lavers and Bond, 2016; Santos et al., 2016; Chapter 2). Most plastics ingested by seabirds and sampled at sea were microplastics (Table S3.7). Only four hard fragments sampled at sea (< 1%) were macroplastics (Table S3.7), while most litter ingested by albatrosses and giant petrels in Chapter 2 fell within this size range. Smaller petrels are therefore better suited to monitor the abundance of small buoyant plastics (2-25 mm in size) at sea compared to the larger albatrosses and giant petrels (Chapter 2). The greater proportions of black items in marine plastics compared to ingested plastics further supports the notion that seabirds select for plastics that resemble prey, or colours that are more conspicuous. In Chapter 2, I found similar results for albatrosses and giant petrels that mostly ingested lighter-coloured items that are more visible on the ocean surface. Although some plastic may be ingested secondarily through prey, this is likely to be appreciably smaller items, and it is widely accepted that surface-feeding seabirds mostly ingest plastic items directly from the ocean surface (Ryan, 1987a; Ryan, 2016; Santos et al., 2016; Roman et al., 2019a). As a community, the types, colours and polymer types of plastics ingested by petrels sampled at Inaccessible Island were overall similar to that collected at sea; white/clear hard fragments, composed of PE that fell within the microplastics size range were the most numerous in both. At a species level however, interspecific variation in the types and colours of plastics ingested were evident. Most noteworthy was the greater proportions of industrial pellets in White-faced Storm Petrels, thread-like plastics in *Fregetta* storm petrels, and flexible and foamed plastics in Great Shearwaters.

There are few data on the at-sea distribution of storm petrels breeding on Inaccessible Island, but it is thought that White-faced Storm Petrels mostly forage north of the island in areas overlapping with the South Atlantic Gyre (Ryan, 2023b; Fig. S3.4). Here, litter densities and plastic exposure risk are higher than the waters south of the Sub-tropical Front (Ryan, 1988a; Eriksen et al., 2014; Ryan et al., 2014; Clark et al., 2023; Fig. 3.1). The greater proportions of industrial pellets ingested by White-faced Storm Petrels could be influenced by foraging area, as many industrial pellets may have been drifting for a long time and thus tend to accumulate within the South Atlantic Gyre (Olivelli, 2019; Zhao et al., 2022). However, it is also likely influenced by species preferences as White-faced Storm Petrels preferred hard fragments and industrial pellets that were white/clear in colour and between 3.2–3.5 mm in length (likely resembling fish eggs), which is similar to the size of fragments and pellets recorded at sea (3.0–

3.2 mm, 88% white/clear). They also ingested more red/pink and orange/brown items than the other seabirds, as these likely resemble crustacean prey items (Ryan, 1987a). In the Northeast Atlantic, White-faced Storm Petrels breeding on the Selvagens Archipelago also ingested mostly white/clear items composed primarily of PE, but only 8% of items were industrial pellets (Furtado et al., 2016).

Fregetta storm petrels likely forage south of the Sub-tropical Front (Ryan, 2023b; Fig. S3.4), where floating plastic loads are generally lower (Ryan, 1988a; Eriksen et al., 2014; Ryan et al., 2014; Clark et al., 2023; Fig. 3.1). Waters just south of the Sub-tropical Front have high industrial fishing intensity (Kroodsma et al., 2018), which may be a source of thread-like plastics (e.g. rope or fishing line), potentially accounting for the greater proportion of thread-like plastics recorded in *Fregetta* storm petrels. However, they seemed to prefer blue/purple and green items (14%), which is generally also the predominant colour of ingested (78%) and marine thread-like plastics (63%). Broad-billed Prions also mostly forage in waters south of the Sub-tropical Front (Jones et al., 2020; Fig. S3.4), but generally did not exhibit a preference for a particular type of plastics like the storm petrels. This is because they are specialist filter-feeders that target small copepods (Ryan, 1987a; Klages and Cooper, 1992; Dell'Ariccia et al., 2017). Similar types of plastics were recorded in both adult and chick Broad-billed Prions, although the items recorded in chicks averaged slightly larger. Juvenile Cassin's Auklets *Ptychoramphus aleuticus* contain larger plastic particles than adults (Nania and Shugart, 2021), and juvenile Barau's Petrels *Pterodroma baraui* and Tropical Shearwaters *Puffinus bailloni* contain plastic items that are heavier than those found in adult birds (Cartraud et al., 2019). It is unclear why plastics in chicks average larger than those in adults. Perhaps possible longer retention periods in adults, where plastics are eroded into smaller sizes over time, may contribute to the size variation of plastic items among different age groups, or perhaps chicks have a less developed GIT system and unable to erode plastics into smaller pieces (Ryan and Jackson, 1987; Ryan, 1988b; Nania and Shugart, 2021). The size discrepancy between chicks and adults warrants further investigation, as it might provide information on the retention times of plastics in seabirds.

Great Shearwaters ingested a wider array of the types of plastics found floating at sea, likely influenced by their foraging areas and their greater tendency to scavenge behind vessels than other species in this study. Great Shearwaters undertake an annual trans-Equatorial migration to winter in the northwest Atlantic (Powers et al., 2022; Robuck et al., 2022; Fig. S3.4). This region borders highly industrialized coastal margins with high human population densities, litter concentrations and plastic exposure risk (Eriksen et al., 2014; Robuck et al., 2022; Clark et al., 2023). Here, they are likely to encounter more items with a higher surface area to volume ratio (e.g. flexible plastics like bags), which is inversely related to the likelihood of dispersal and longevity at sea (Ryan, 2015b; Fazey and Ryan, 2016; Ryan, 2020). These items are also likely to be encountered during their commute back to the South Atlantic via the coastal waters of Brazil and southwest Africa (Powers et al., 2022; Robuck et al., 2022), or ingested while scavenging at fishing vessels.

Conclusions

I found interspecific differences in the types, colours and sizes of plastics ingested by petrels in the South Atlantic Ocean. Storm petrels ingested items more similar in size to plastics sampled with a surface net at sea, but were more selective for thread-like plastics and industrial pellets than the larger prions and shearwaters. Great Shearwaters sampled plastics that were more similar in composition to marine plastics than the other taxa, but generally ingested larger plastics than recorded in net samples. Surface net tows were generally more effective at sampling smaller plastics at sea. This suggests that seabirds may not be ideal indicators of plastic size; however, smaller seabird species closest represented the size range collected by surface nets and thus serve as more reliable bioindicators of environmental plastic sizes collected by these nets compared to larger taxa. However, as a community, seabirds breeding on Inaccessible Island reflected the overall composition of small buoyant plastics in the region, although scarce polymers were found more frequently in marine samples than ingested by seabirds.

Using regurgitated skua pellets to sample ingested plastics in petrels is a simple method that delivers robust results through large sample sizes. It also eliminates the need to sample birds specifically for this purpose, avoiding harm to live birds and reducing sampling costs. Regurgitated skua pellets provide a valuable tool to monitor plastics, which could be used to assess the efficacy of mitigation measures aimed at reducing the prevalence of floating plastic in the marine environment.

Supplementary materials: Chapter 3

Table S3.1 Year, voyage, ocean sampled, latitude (°), longitude (°) and density of plastic litter items by number (items·km^{-2}) and by mass (g·km^{-2}) of all stations sampled in this study. ACE = Antarctic Circumnavigation Expedition; SW = southwest.

Year	Voyage	Ocean	Latitude	Longitude	Density	
					items·km^{-2}	g·km^{-2}
2016	ACE	South Atlantic	-2.70125	-10.23524	0	0
2016	ACE	South Atlantic	-3.85484	-9.30984	0	0
2016	ACE	South Atlantic	-6.07892	-7.52317	1806	1.81
2016	ACE	South Atlantic	-7.25475	-6.57884	7750	62
2016	ACE	South Atlantic	-9.58598	-4.70603	2821	0
2016	ACE	South Atlantic	-10.7709	-3.74105	0	0
2016	ACE	South Atlantic	-13.12091	-1.84834	7226	19.87
2016	ACE	South Atlantic	-14.28853	-0.90514	0	0
2016	ACE	South Atlantic	-16.43465	0.87923	3412	5.12
2016	ACE	South Atlantic	-17.62744	1.8473	0	0
2016	ACE	South Atlantic	-20.23789	3.0499	1171	22.24
2016	ACE	South Atlantic	-31.72894	13.78313	1216	23.1
2017	ACE	South Atlantic	-54.85659	-54.81672	0	0
2017	ACE	South Atlantic	-54.99381	-50.04539	0	0
2017	ACE	South Atlantic	-56.97442	-27.86945	0	0
2017	ACE	South Atlantic	-59.50325	-21.015	0	0
2017	ACE	South Atlantic	-58.65729	-13.94965	0	0
2017	ACE	South Atlantic	-57.50438	-7.01401	0	0
2017	ACE	South Atlantic	-54.65381	0.96423	940	no data
2017	ACE	South Atlantic	-51.04802	6.99373	0	0
2017	ACE	South Atlantic	-48.98665	9.02501	0	0
2017	ACE	South Atlantic	-43.97678	14.11134	0	0
2017	ACE	South Atlantic	-34.67874	17.23094	2996	35.95
2017	SEAmester II	South Atlantic	-34.50888	8.3329	19016	373.98
2017	SEAmester II	South Atlantic	-34.50517	3.56752	9033	33.55
2017	SEAmester II	South Atlantic	-34.50391	0.02498	18176	458.75
2017	SEAmester II	South Atlantic	-34.49666	5.10195	12425	171.46
2017	SEAmester II	South Atlantic	-34.49263	10.18383	47369	658.05
2017	SEAmester II	South Atlantic	-35.01661	12.4497	3549	44.96
2017	SEAmester II	South Atlantic	-35.25027	13.067	2393	7.18
2017	SEAmester II	South Atlantic	-35.41008	13.46849	11729	279.36
2017	SEAmester II	South Atlantic	-35.25582	13.65739	1224	33.04
2017	SEAmester II	South Atlantic	-34.96585	13.93202	0	0
2017	SEAmester II	South Atlantic	-34.74773	14.30475	0	0
2017	SEAmester II	South Atlantic	-34.49927	14.63222	4454	12.25
2017	SEAmester II	South Atlantic	-34.49826	17.13556	2494	12.47

2017	SEAmester II	South Atlantic	-34.40727	17.54742	915	0.92
2019	SCALE winter	South Atlantic	-51.40154	0.00021	0	0
2019	SCALE winter	South Atlantic	-46.99951	4.49608	0	0
2019	SCALE winter	South Atlantic	-44.99836	6.59654	0	0
2019	SCALE winter	South Atlantic	-34.66661	21.9598	6069	6.37
2019	Gough relief	South Atlantic	-34.33236	13.84198	1168	4.79
2019	Gough relief	South Atlantic	-34.78766	9.53771	7272	19.01
2019	Gough relief	South Atlantic	-35.00837	7.34269	30401	95.31
2019	Gough relief	South Atlantic	-35.43629	3.32012	17246	27.72
2019	Gough relief	South Atlantic	-35.67947	0.93021	12092	88.07
2019	Gough relief	South Atlantic	-46.54249	-9.22197	0	0
2019	Gough relief	South Atlantic	-43.74872	-9.59221	1446	0.29
2019	Gough relief	South Atlantic	-40.31752	-9.87953	1110	3.88
2019	Gough relief	South Atlantic	-39.50575	-10.49979	1395	2.93
2019	Gough relief	South Atlantic	-35.81089	-0.27819	6675	115.47
2019	Gough relief	South Atlantic	-35.54664	2.24794	0	0
2019	SCALE spring	South Atlantic	-35.77036	14.88863	0	0
2019	SCALE spring	South Atlantic	-46.99563	5.292	94	0.85
2019	SCALE spring	South Atlantic	-54.01015	-0.01415	0	0
2019	SCALE spring	South Atlantic	-55.0152	0.02534	0	0
2019	SCALE spring	South Atlantic	-49.3016	2.30317	0	0
2019	SCALE spring	South Atlantic	-49.29643	2.304172	0	0
2019	SCALE spring	South Atlantic	-46.996457	4.498129	3023	0.91
2019	SCALE spring	South Atlantic	-46.988096	4.51467	0	0
2019	SCALE spring	South Atlantic	-42.994343	8.503527	0	0
2019	SCALE spring	South Atlantic	-42.984734	8.494199	0	0
2019	SCALE spring	South Atlantic	-44.975091	6.635913	0	0
2019	SCALE spring	South Atlantic	-44.959768	6.647612	0	0
2019	SCALE spring	South Atlantic	-41.501203	9.651539	0	0
2019	SCALE spring	South Atlantic	-41.507953	9.641987	0	0
2019	SCALE spring	South Atlantic	-38.624772	11.856351	0	0
2019	SCALE spring	South Atlantic	-36.302245	13.300463	0	0
2019	SCALE spring	South Atlantic	-36.311609	13.292956	0	0
2016	ACE	SW Indian	-46.91922	38.11349	2197	28.56
2016	ACE	SW Indian	-46.165	51.016	0	0
2017	ACE	SW Indian	-50.9903	72.04491	0	0
2017	ACE	SW Indian	-53.1143	81.77099	0	0
2017	ACE	SW Indian	-54.84316	95.76494	0	0
2017	ACE	SW Indian	-53.79864	112.36885	0	0
2017	ACE	SW Indian	-53.2547	118.21754	0	0
2017	ACE	SW Indian	-49.12376	133.55554	0	0
2017	ACE	SW Indian	-46.38769	150.40645	0	0
2017	ACE	SW Indian	-53.62655	149.31004	0	1.13
2017	ACE	SW Indian	-59.61187	148.65504	0	0

2017	Marion relief	SW Indian	-46.64817	38.01345	2389	0
2017	Marion relief	SW Indian	-41.00037	26.83743	0	0
2017	Marion relief	SW Indian	-37.92712	24.75787	11477	33.39
2017	Marion relief	SW Indian	-37.5001	24.34933	51198	923.88
2017	Marion relief	SW Indian	-36.49902	23.70837	20027	110.15
2017	Marion relief	SW Indian	-36.25413	23.54003	7402	13.75
2017	Marion relief	SW Indian	-36.02685	23.35848	7197	3.6
2018	SEAmester III	SW Indian	-34.26856	26.47443	10063	30.44
2018	SEAmester III	SW Indian	-33.35841	27.46891	19563	60.92
2018	SEAmester III	SW Indian	-33.66716	27.51881	10500	7.87
2018	SEAmester III	SW Indian	-33.7598	27.59531	2076	0.21
2018	SEAmester III	SW Indian	-33.93726	27.74015	6753	340.36
2018	SEAmester III	SW Indian	-34.15901	27.90549	27892	192.92
2018	SEAmester III	SW Indian	-34.4062	28.08919	21296	461.63
2018	SEAmester III	SW Indian	-34.66491	28.26088	7886	44.79
2018	SEAmester III	SW Indian	-34.96581	28.4613	1246	12.09
2018	SEAmester III	SW Indian	-35.3293	28.67354	0	0
2018	SEAmester III	SW Indian	-35.72899	28.91256	12627	721.83
2018	SEAmester III	SW Indian	-33.95322	27.76313	34728	242.94
2018	SEAmester III	SW Indian	-33.75706	27.62773	5477	100.41
2019	Marion relief	SW Indian	-42.00173	27.62021	7607	95.24
2019	Marion relief	SW Indian	-39.99837	26.07868	13196	92.81
2019	Marion relief	SW Indian	-38.59875	25.0986	2116	12.06
2019	Marion relief	SW Indian	-37.67754	24.46256	4883	30.03
2019	Marion relief	SW Indian	-37.3606	24.2161	0	0
2019	Marion relief	SW Indian	-36.87083	23.90104	4739	13.9
2019	Marion relief	SW Indian	-36.04174	23.37893	4059	58.24
2019	SEAmester IV	SW Indian	-34.04901	26.71961	7347	10.78
2019	SEAmester IV	SW Indian	-33.34868	27.48338	13772	17.49
2019	SEAmester IV	SW Indian	-33.46255	27.52694	7615	28.79
2019	SEAmester IV	SW Indian	-33.58266	27.56036	0	0
2019	SEAmester IV	SW Indian	-33.61558	27.60478	0	0
2019	SEAmester IV	SW Indian	-33.69332	27.57467	3839	614.2
2019	SEAmester IV	SW Indian	-33.73051	27.64858	0	0
2019	SEAmester IV	SW Indian	-33.80798	27.59111	729	0.15
2019	SEAmester IV	SW Indian	-33.90346	27.71389	0	0

Table S3.2 The mean (± SD) length (mm) of plastic items (all items and per category) collected during surface net tows (marine plastics) and ingested by seabirds breeding on Inaccessible Island in the central South Atlantic Ocean. Chick samples were only included for Broad-billed Prions.

	All items length (± SD)	Hard fragments	Industrial pellets	Flexible pieces	Thread-like	Foamed plastics
Marine plastics	6 (± 23)	5 (± 23)	3 (± 1)	9 (± 6)	27 (± 27)	4 (± 1)
All seabirds	6 (± 4)	6 (± 3)	4 (± 1)	13 (± 8)	26 (± 24)	6 (± 1)
Fregetta SP*	6 (± 11)	4 (± 2)	4 (± 1)	-	28 (± 28)	-
White-faced SP*	4 (± 2)	4 (± 2)	4 (± 1)	7 (± 1)	13 (± 13)	-
Broad-billed Prion						
adults	6 (± 5)	6 (± 2)	4 (± 1)	16 (± 11)	57 (± 61)	-
chicks	6 (± 3)	6 (± 2)	4 (± 1)	9 (± 3)	13 (± 8)	-
Great Shearwater	6 (± 4)	6 (± 3)	4 (± 1)	14 (± 8)	26 (± 19)	6 (± 1)

*SP = storm petrel

Table S3.3 The mean (± SD) mass (mg) of plastic items (all items and per category) collected during surface net tows (marine plastics) and ingested by seabirds breeding on Inaccessible Island in the central South Atlantic Ocean. Chick samples were only included for Broad-billed Prions.

	All items mass (± SD)	**Hard fragments**	**Industrial pellets**	**Flexible pieces**	**Thread-like**	**Foamed plastics**
Marine plastics	13 (± 50)	14 (± 52)	11 (± 7)	7 (± 8)	3 (± 2)	1 (± 0.4)
All seabirds	25 (± 49)	27 (± 46)	20 (± 10)	10 (± 12)	6 (± 18)	7 (± 6)
Fregetta SP*	15 (± 33)	16 (± 36)	18 (± 8)	-	3 (± 5)	-
White-faced SP*	9 (± 12)	7 (± 12)	14 (± 8)	5 (± 4)	8 (± 12)	-
Broad-billed Prion						
adults	25 (± 26)	25 (± 27)	23 (± 9)	6 (± 9)	6 (± 7)	-
chicks	29 (± 34)	30 (± 36)	24 (± 10)	8 (± 6)	2 (± 1)	-
Great Shearwater	30 (± 55)	31 (± 57)	22 (± 9)	11 (± 13)	8 (± 27)	7 (± 6)

*SP = storm petrel

Table S3.4 Number of items that were assigned a polymer type ($\geq$ 70% match) and the number of items selected for FTIR processing per month for each taxon in parentheses.

Taxa	September	October	November	Total
Fregetta storm petrels	113	14	5	
hard fragments	93 (104)	5 (6)	2 (2)	100 (112)
industrial pellets	13 (15)	4 (4)	1 (1)	18 (20)
flexible pieces	–	–	1 (1)	1 (1)
thread-like plastics	10 (12)	5 (5)	1 (1)	16 (18)
White-faced Storm Petrel	112	54	38	
hard fragments	89 (96)	11 (11)	–	100 (107)
industrial pellets	20 (20)	43 (43)	37 (38)	100 (101)
flexible pieces	1 (2)	–	–	1 (2)
thread-like plastics	2 (3)	–	1 (1)	3 (4)
Broad-billed Prion	44	160	5	
hard fragments	37 (37)	63 (64)	–	100 (101)
industrial pellets	7 (7)	93 (97)	–	100 (104)
flexible pieces	–	3 (4)	3 (8)	6 (12)
thread-like plastics	–	1 (2)	2 (2)	3 (4)
Great Shearwater	103	112	34	
hard fragments	95 (95)	5 (5)	–	100 (100)
industrial pellets	6 (6)	68 (69)	26 (29)	100 (104)
flexible pieces	0 (3)	23 (30)	6 (9)	29 (42)
thread-like plastics	2 (2)	15 (15)	2 (3)	19 (20)
foam	–	1 (3)	–	1 (3)
Total	375 (402)	340 (358)	82 (95)	797 (855)

Table S3.5 Proportions of polymers among marine plastics and plastics ingested by seabirds. Chi-square test results (df = 1) compare the ratios of ingested (for each taxa and all seabirds combined) to marine polymer types. [polypropylene (PP), polystyrene (PS), polypropylene ethylene-propylene-diene monomer (EPDM), polyvinyl chloride (PVC), polymethyl methacrylate (PMMA), ethylene propylene rubber (EPM) and polyurethane (PU) were pooled for comparisons between each taxon and marine plastics due to low sample sizes].

	PE	**PP**	**PS**	**PP/EPDM**	**PVC**	**PMMA**	**EPM**	**PU**
Marine plastics								
Hard fragments	72%	25%	2%	1%	< 1%	< 1%	< 1%	—
Industrial pellets	100%	—	—	—	—	—	—	—
Flexible pieces	70%	30%	—	—	—	—	—	—
Thread-like plastics	57%	29%	—	14%	—	—	—	—
Foam	—	—	75%	25%	—	—	—	—
All seabirds (χ^2 =9.49, df = 2, P < 0.05)								
Hard fragments	78%	22%	—	—	—	—	—	—
Industrial pellets	83%	15%	< 1%	2%	—	—	—	—
Flexible pieces	38%	62%	—		—	—	—	—
Thread-like plastics	71%	24%	—	5%	—	—	—	—
Foam	—	—	—	—	—	—	—	100%
Fregetta storm petrels (χ^2 = 5.12, P < 0.05)								
Hard fragments	88%	12%	—	—	—	—	—	—
Industrial pellets	83%	17%	—	—	—	—	—	—
Flexible pieces	—	100%	—	—	—	—	—	—
Thread-like plastics	44%	50%	—	6%	—	—	—	—
White-faced Storm Petrel (χ^2 = 7.53, P < 0.05)								
Hard fragments	75%	25%	—	—	—	—	—	—
Industrial pellets	89%	11%	—	—	—	—	—	—
Flexible pieces	—	100%	—	—	—	—	—	—
Thread-like plastics	100%	—	—	—	—	—	—	—
Broad-billed Prion* (χ^2 = 0.66, P = 0.42)								
Hard fragments	74%	26%	—	—	`—	—	—	—
Industrial pellets	78%	20%	—	2%	—	—	—	—
Flexible pieces	17%	83%	—	—	—	—	—	—
Thread-like plastics	100%	—	—	—	—	—	—	—
Great Shearwater (χ^2 = 0.76, P = 0.38)								
Hard fragments	74%	26%	—	—	—	—	—	—
Industrial pellets	83%	13%	1%	3%	—	—	—	—
Flexible pieces	45%	55%	—	—	—	—	—	—
Thread-like plastics	84%	11%	—	5%	—	—	—	—
Foam	—	—	—	—	—	—	—	100%

* Industrial pellets include samples collected from both adults (n=12) and chicks (n=88)

Table S3.6 Comparison of the mean mass (mg), and length (mm) of plastic items collected from carcasses to those collected from skua pellets.

Collection method	n birds (items)	Mean mass ± SD (range)	Mean length ± SD (range)	Source
White-faced Storm Petrel				
Carcass	5 (46)	4	–	Ryan (1987a)
Carcass	6 (78)	5 ± 5	3 ± 1	Roman et al. (2019b)
Carcass	2 (54)	5 ± 5 (0.7 – 30)	3 ± 1 (2 – 5)	This study
Skua pellet	147 (767)	9 ± 12 (0.6 – 201)	4 ± 2 (2 – 32)	This study
Broad-billed Prion				
Carcass	28 (85)	15	–	Ryan (1987a)
Carcass	2 (7)	2 ± 1	3 ± 2	Roman et al. (2019b)
Carcass (chick)	1 (50)	18 ± 23 (1 – 99)	5 ± 3 (2 – 13)	This study
Skua pellet	99 (406)	25 ± 26 (0.3 – 190)	6 ± 5 (2 – 100)	This study
Great Shearwater				
Carcass	32 (536)	21	–	Ryan (1987a)
Carcass	15 (229)	23 ± 48 (0.01 – 574)	7 ± 6 (0.9 – 60)	Robuck et al. (2022)
Carcass	2 (204)	22 ± 31 (0.3 – 216)	7 ± 6 (0.4 – 45)	This study
Skua pellet	112 (2580)	30 ± 55 (0.5 – 990)	6 ± 4 (1.3 – 86)	This study

Table S3.7 Proportions and total numbers of hard fragments and pellets within each size category commonly assigned in plastic studies.

	Microplastic (< 5mm)	**Mesoplastic (5-25 mm)**	**Macroplastic (> 25 mm)**	**Total**
Marine plastics	78%	21%	1%	370
All seabirds	56%	44%	0%	3 865
Fregetta storm petrels	82%	18%	0%	191
White-faced Storm Petrel	92%	8%	0%	761
Broad-billed Prion (adults)	48%	52%	0%	398
Great Shearwater	44%	55%	0%	2 515

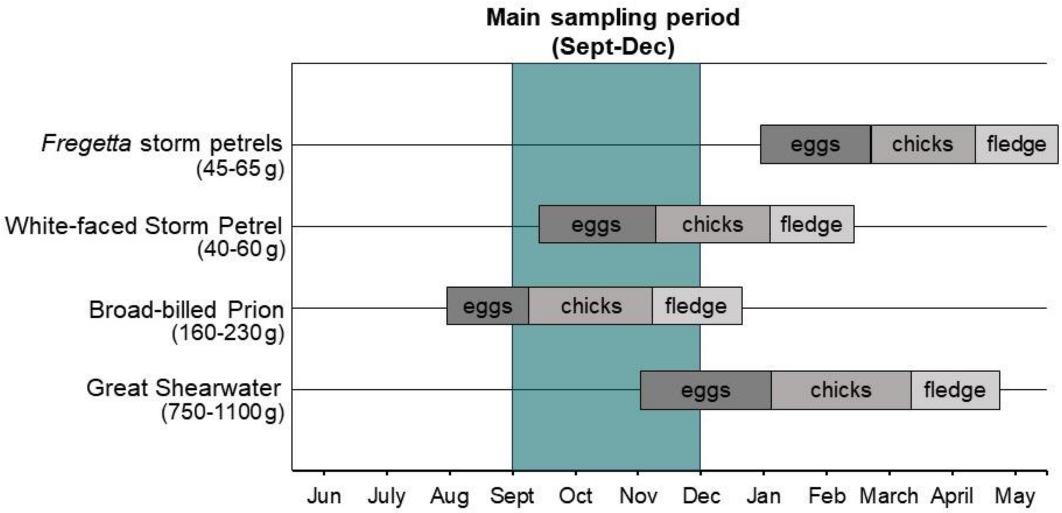
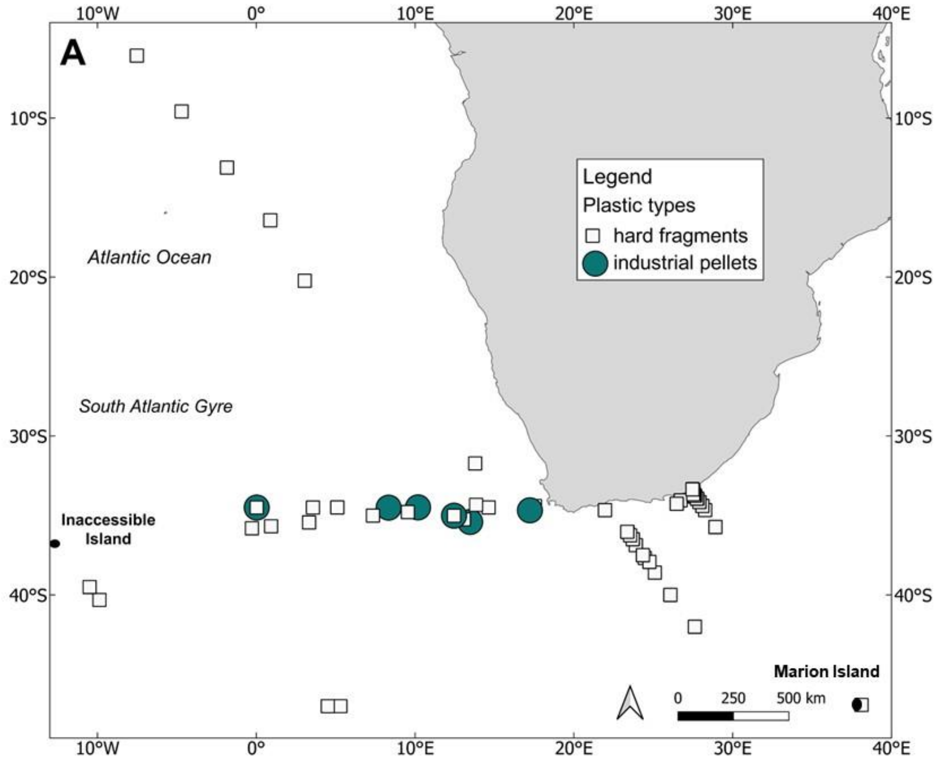


Fig. S3.1 Breeding phenology of the four petrel taxa sampled in this study using data from Ryan (2007), Dilley et al. (2019) and Ryan (2023). Data for *Fregetta* storm petrels are less certain as it was not possible to identify birds to species level (Robertson et al., 2016; Dilley et al., 2019). Body mass (in grams) indicated in parentheses below each taxon name.



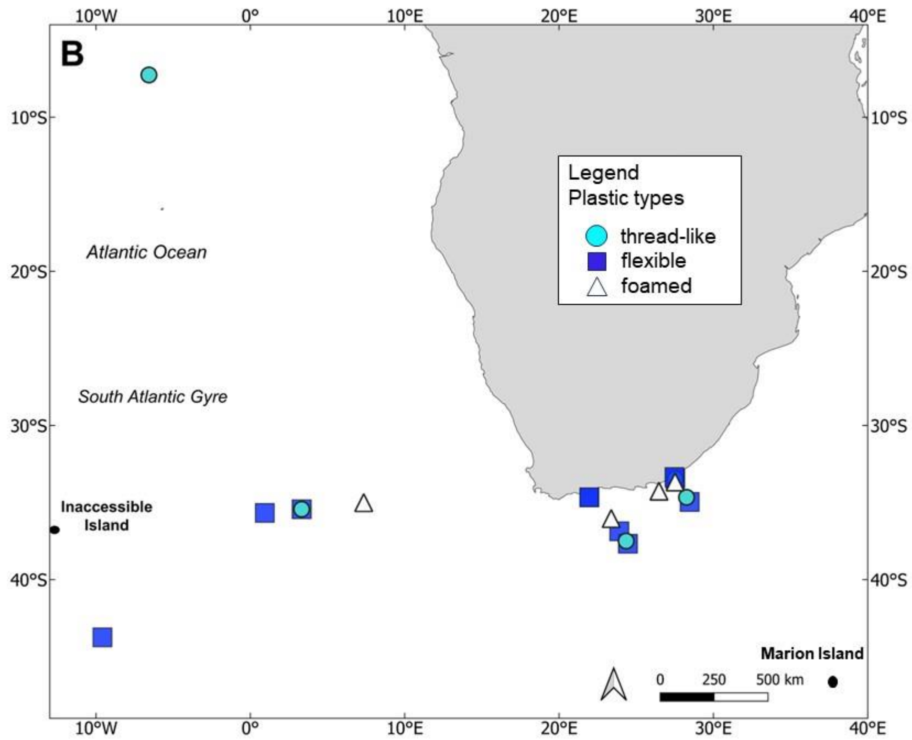


Fig. S3.2 Distribution of hard fragments and industrial pellets (A) and flexible pieces, thread-like plastics and foam (B) in the South Atlantic and southwest Indian Oceans.

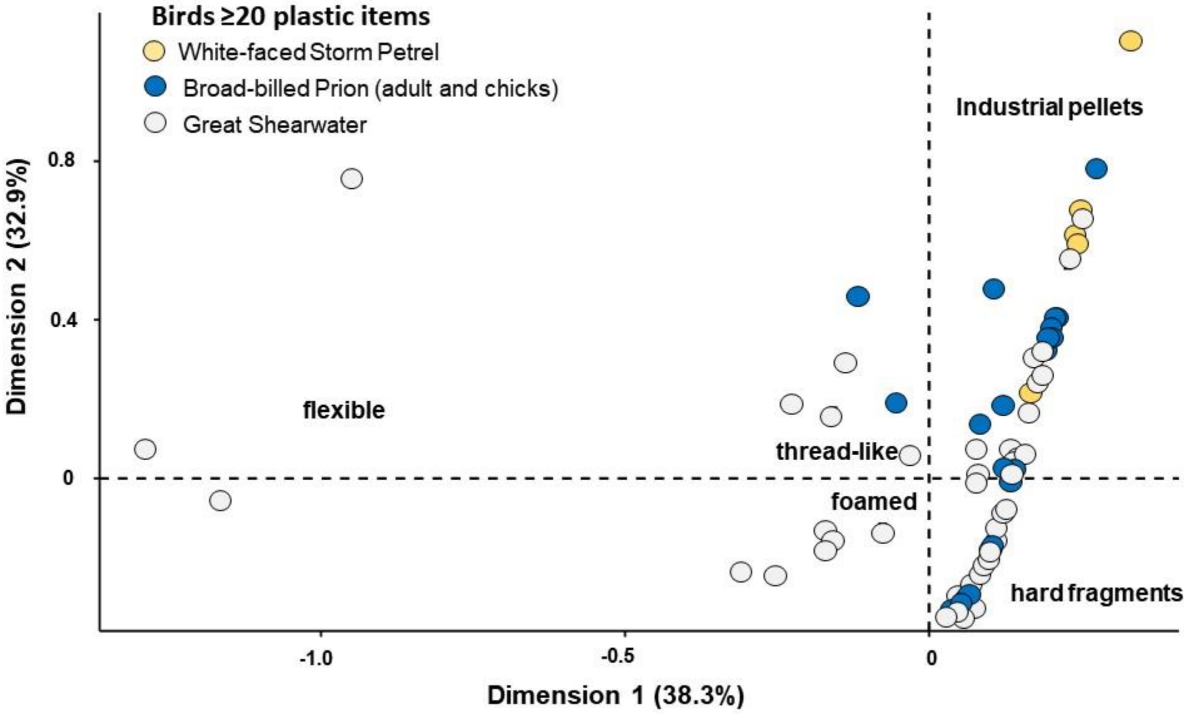


Fig. S3.3 Correspondence analysis indicating the relationship between the types of plastics ingested by White-Faced Storm Petrels (n=5), Broad-billed Prions (n = 1 adult; n = 19 chicks) and Great Shearwaters (n = 48) that ingested more than 20 items.

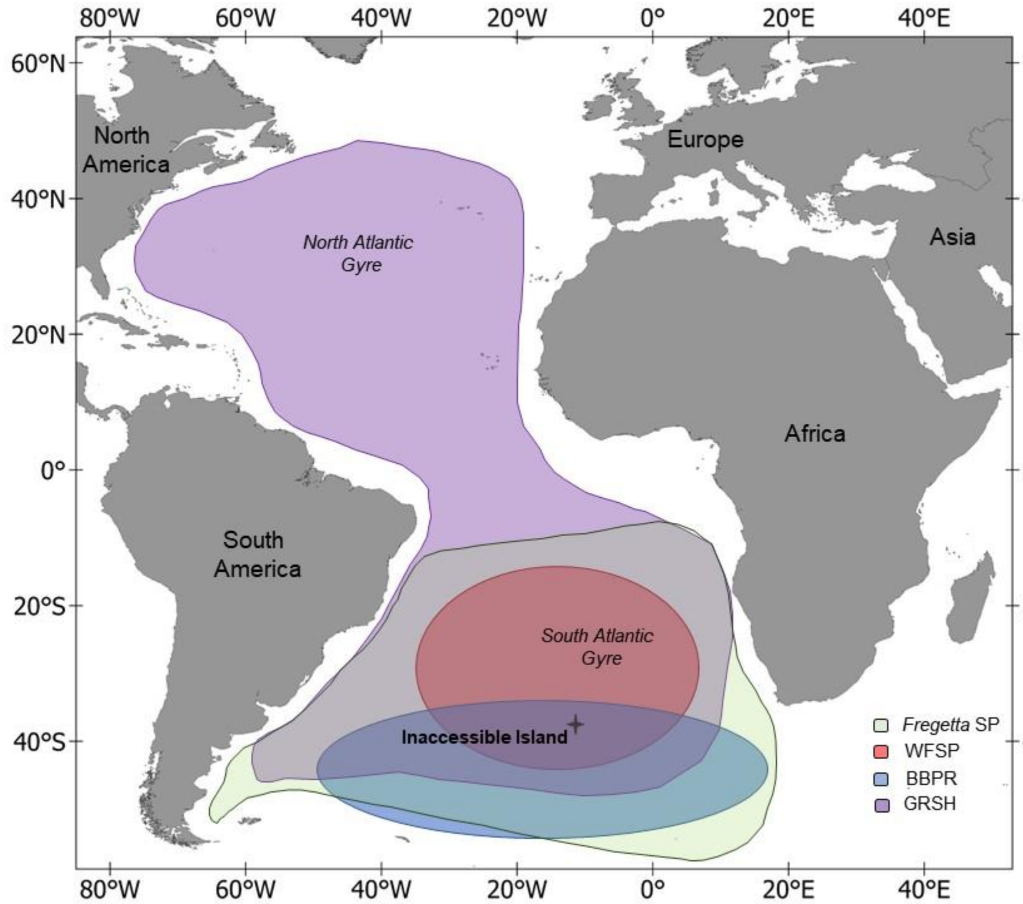


Fig. S3.4 Approximate Atlantic Ocean ranges of *Fregetta* storm petrels (*Fregetta* SP, green), White-faced Storm Petrels (WFSP, red), Broad-billed Prions (BBPR, blue) and Great Shearwaters (GRSH, purple) are based on references listed in the Methods.

Chapter 4

Exploring temporal and spatial variation in plastic loads in South Atlantic petrels since the 1980s

A Great Shearwater found dead on Inaccessible Island in October 2018 that contained 200 plastic items: 4 industrial pellets and 196 other plastic items (186 hard user fragments, 4 flexible packaging fragments, 6 threads) (photo Peter Ryan).

Abstract

Despite the growing concern about the large amounts of mismanaged plastic waste entering marine ecosystems, evidence of an increase in the amount of floating plastic at sea has been mixed. Both at-sea surveys and ingested plastic loads in seabirds and fish show inconsistent evidence of significant increases in the amount of marine plastics since the 1980s, and only recently have studies reported increases at sea and in some seabirds. In this chapter, I use 3 727 Brown Skua regurgitations, each containing the remains of a single seabird, to investigate changes in plastic loads in four petrel taxa breeding at Inaccessible Island, Tristan da Cunha in nine years from 1987–2018. Plastic loads were compared across four near-decadal time periods (1987–89; 1999–2004; 2009–2014 and 2018) and years. Three taxa that remain in the South Atlantic Ocean year-round showed fluctuations, but no clear trends in the total amount of ingested plastic. Plastic loads in Great Shearwaters, which spend the austral winter in the North Atlantic Ocean, increased in 2018, although the proportion of shearwaters containing plastic decreased. Despite global plastic production increasing more than four-fold over the study period, there was no consistent increase in the total amount of ingested plastic in any species. The number and proportions of industrial pellets among ingested plastic decreased consistently over the study period, suggesting that industry initiatives to reduce pellet leakage have reduced the numbers of pellets at sea.

Introduction

Global production of plastics has increased rapidly over the last 75 years and continues to grow at around 8% per year (Geyer et al., 2017; Chapter 1; Fig. 1.2). Of this, about 79% is dumped in landfills or released into the environment, and in 2015 alone 60–90 Mt of mismanaged plastic waste were produced globally, and this figure is likely to triple by 2060 (Geyer et al., 2017; Lebreton and Andrady, 2019). Because plastics are relatively light, with densities similar to that of water, much of this mismanaged waste plastic ends up in water bodies where it can disperse far from source areas (Chapter 1; Figs. 1.3 and 3.1) Unless there is an urgent paradigm shift in the way we treat plastic waste, coupled with extraordinary global efforts to curb plastic emissions, the amounts of plastics entering our oceans are predicted to increase exponentially (Jambeck et al., 2015; Borrelle et al., 2020). As a result, an international treaty to control the use of plastics is currently being negotiated (Landrigan et al., 2023).

Given these alarming estimates, we might expect to detect a marked increase in the amount of plastic floating at the sea surface over the last few decades (Thompson, 2015; Galgani et al., 2021). However, the evidence for such an increase is mixed. Ostle et al. (2019) reported an overall significant increase in open ocean macroplastics between 1957–2016 using records of plastic entanglement in a Continuous Plankton Recorder. This is not surprising, as the plastic production industry was in its infancy during the 1950s, and an increase is to be expected over time. They were however unable to show significant change in macroplastic counts for 2010–2016. Studies failed to detect an increase in the concentration of surface microplastics in the North Atlantic Sub-tropical Gyre between 1986–2008 (Law et al., 2010), or in the Eastern Pacific Ocean between 2001–2012 (Law et al., 2014). However, Wilcox et al. (2020) extended the analysis for the North Atlantic to 2015, and found an increase in plastic concentrations mirroring global plastic production. Lebreton et al. (2018) found an exponential increase in the mass of plastic in surface net tows in the Great Pacific Garbage Patch over the last decade, but little change between the 1970s and 2005. Similarly, Eriksen et al. (2023) analysed data collected between 1979–2019, and found a rapid increase in floating ocean plastics only from 2006 onwards. The difficulty to detect a consistent increasing trend in floating plastics at sea may simply reflect the challenges associated with monitoring floating plastic at sea, given the high levels of spatial and temporal heterogeneity (Ryan et al., 2009a; GESAMP, 2019). This issue is particularly problematic for small plastic fragments which typically are sampled with nets that only sample small areas. For instance, the 116 surface net tows in Chapter 3 (Fig. 3.1) amounted to a sample area of only 0.09 km^2 of sea, showing how very large sample sizes are required to detect changes in plastic abundance (Ryan et al., 2009a). However, Law et al. (2010) failed to detect any change in floating plastic in the North Atlantic despite taking more than 6 100 net samples over 22 years. In addition, buoyant plastics may be lost from the sea surface due to biofouling, either sinking to the sea floor or oscillating vertically between the surface and mid-ocean layer, further complicating the likelihood of detecting marked trends in the density of floating plastics (Ryan et al., 2009a; Reisser et al., 2015).

As discussed in Chapters 1 and 3, a potentially more powerful approach to detect changes in floating plastic is to track changes in plastic loads in biota (both the proportion of individuals containing plastic and the number/mass of items per individual) as they collect litter over large areas and can be sampled at relatively little cost (Ryan et al., 2009a; Provencher et al., 2017; Baes et al., 2024). Because petrels accumulate ingested plastic items in their stomachs, they can integrate plastic loads over their large foraging ranges at a temporal scale of weeks to months (van Franeker et al., 2011; Ryan, 2015a; Ryan, 2016; Clark et al., 2023) and therefore, some have been proposed as indicators of floating plastics (van Franeker and Law, 2015; Avery-Gomm et al., 2018; Baes et al., 2024; Rodríguez et al., 2024; Chapter 3).

Using seabirds as ecological indicators may help to overcome the spatial and temporal limitations posed by *in situ* sampling of the ocean surface for floating micro- and mesoplastics (as seen in Chapter 3), but it can be challenging to obtain sufficiently large numbers of samples without recourse to collecting birds. In Chapter 1-3, I discussed that the best ways to sample ingested plastics to assess plastic loads in birds is to examine the entire gastro-intestinal tract. Emetics may be used for this, but is very invasive and can cause some mortalities (Bond and Lavers, 2013; Lavers et al., 2021), and may not recover all ingested plastic (Provencher et al., 2019). Most studies rely on sampling birds found dead, such as beached birds (e.g. Harper and Fowler, 1987; Ryan, 1987a; van Franeker and Law, 2015; Petry and Benemann, 2017; Lavers et al., 2022; Baes et al., 2024), or birds killed accidentally by fishing activities (Mallory et al., 2006; Ryan, 2008) or by light-induced collisions (Ryan, 1987a; Rodríguez et al., 2012; Cartraud et al., 2019; Rodríguez et al., 2024). However, birds found dead of unknown causes (e.g. washed up on beaches or at colonies), may contain more plastic than individuals sampled randomly (e.g. collision victims or long-line bycatch) either because the ingested plastics contributed to their deaths (Rodríguez et al., 2018; Roman et al., 2019a), or they were in poor condition and became less selective while foraging, leading to inflated plastic loads (Ryan, 1987b; Ryan, 2016; although this is not always the case, e.g. van Franeker and Meijboom, 2002). In Chapter 3, I showed that a simple, non-destructive and repeatable method to sample a large number of birds is to examine the pellets of regurgitated prey remains of predators that feed on seabirds (Ryan and Fraser, 1988; Ryan, 2008; Furtado et al., 2016; Hammer et al., 2016; Provencher et al., 2019). I also showed that the plastics ingested by this community were similar to small buoyant plastics sampled with a surface net within their foraging areas. Using regurgitated skua pellets to sample ingested plastics does have possible drawbacks such as failing to incorporate all ingested plastics and thus underestimating individual plastic loads, but in Chapter 3 I provided support for the reliability of this method, at least for Brown Skuas.

In this chapter, I build on the results from Chapter 3 showing that regurgitated skua pellets containing the remains of small seabirds can be used as biomonitors of plastics in the South Atlantic Ocean. I use regurgitated pellets from Brown Skuas, containing the remains and plastics of the same four petrel taxa assessed in Chapter 3, to investigate if plastic loads have changed from the late 1980s to 2018. I also compare plastic loads among Broad-billed Prion adults and chicks from the late 1990s to 2018. I finally relate ingested plastic loads in the four adult taxa to the growth in global plastic production over this time, to see if ingested loads match production growth estimates. This chapter provides a rare insight into trends in plastic ingestion levels among seabirds breeding at the same site and sampled in the same way over 30 years.

Methods

Plastic items ingested by seabirds were collected following similar methods to those in Chapter 3, by the same experienced observer on Inaccessible Island, Tristan da Cunha, central South Atlantic Ocean (Fig. 1.6) following a standardised protocol (Ryan, 2008; Ryan, 2023c; Chapter 3). Most samples were collected during 2–5 month trips to Inaccessible Island at roughly four decadal intervals in October 1989–March 1990, November 1999–February 2000, October–December 2009, and September-November 2018. Additional skua pellets were collected during 2–4 week visits in October 1987 and 1988, November 2004 and September 2011, and during a brief visit to the island in September 2014; these were individually bagged and returned to South Africa for processing.

All samples were used to record plastic ingestion (Table 4.1). However, in order to limit seasonal influences on plastic loads, and to reduce the risk of confusing chicks or fledglings with adults, only samples collected in spring and early summer (September-December) were used to assess long-term changes in plastic loads among the four study taxa (Table S4.1; Fig. S3.1). The number of plastic items recovered from a skua pellet, containing the remains of a single bird, is reported as the plastic load for that individual. After counting, plastics from individual pellets from the same prey species were pooled in all years except 2018, so mass per bird cannot be compared. For the purposes of this chapter, plastics were classified as industrial pellets or other plastic items derived from the breakdown of user items (almost all fragments of rigid plastics; a few threads were ingested, but they were < 2% of all items; Table S4.2). The presence of non-avian prey-remains, to infer whether plastic was ingested directly or secondarily via other prey species, was also recorded from the observers notes.

Data analyses

Given the difficulty to access this remote study site (Fig. 1.6), data were only collected in 9 years over the 30-year study period. Therefore, samples collected in September-December in different years were combined into four roughly decadal periods: late 1980s (1987–1989), late 1990s (1999–2004), late 2000s (2009–2014), and 2018 to account for large temporal gaps, and small sample sizes in some years (Table S4.1). These time periods were used in the analyses to assess temporal variation in frequency of occurrence, plastic loads and types across the study period. However, to assess if there were subtle variations among years, I also compared plastic loads and types recorded in each sampling year for each taxon.

All data were assessed for normality using Shapiro-Wilk normality test. Chi-squared goodness-of-fit tests were used to test for differences in the proportion of individuals containing ingested plastics, and to compare ratios of plastic types among time periods. I report plastic loads for *all birds* (includes birds containing no ingested plastic), but report data for only *birds with plastic* in the supplementary material (Table S4.3). The former is the standard approach, as reporting the absence of plastics provides vital information on the pervasiveness of marine pollution, especially in species with low ingestion rates (Kühn et al., 2015; Ryan, 2016; Provencher et al., 2019). However, as ingestion data are generally right-skewed, reporting loads in *birds with plastics* might reveal further information on the trends in plastic loads, but should be in addition to the former (Ryan, 2016; Provencher et al., 2017). Numbers of ingested plastic items were not normally distributed (all $P < 0.001$) so Kruskal-Wallis tests were used to test for differences in plastic loads among time periods. A post-hoc Dunn's test with a Bonferroni correction (to decrease the chance of committing Type 1 errors) was used to show which time periods differed. To assess changes in plastic loads per sampling year instead of time periods, I ran Generalized Additive Models (GAMs) for each taxon with year as the predictor and number of items (plastic load per bird) as the response variable using the "*mgvc*" package in R (Wood, 2011). I assigned cubic spline smoothing parameters using restricted maximum likelihood (REML) and tested both a Poisson and a negative binomial (log link) distribution. Residuals were checked for overdispersion, and the negative binomial distribution was found to be the best fit for all models. Mann-Whitney U tests were used to compare plastic loads in prion adults and chicks sampled in the same year. Means are reported ± 1 standard deviation (SD) unless otherwise indicated. I used Pearson's correlation coefficients (r) to assess the relationships between the number of industrial pellets and other plastics ingested per bird in each year for each taxon. Data analyses were performed in R version 4.2.3 (R Core Team, 2023).

In order to compare ingested plastic loads with expected increases in plastic densities at sea based on global plastic production statistics (https://PlasticsEurope.org), I limited the analysis to the primary sampling years: 1989, 1999, 2009 and 2018. Given lags of several years between production and plastics reaching the open ocean (factoring in the need for fragmentation of secondary plastics), I compared ingested plastic loads to global plastic production statistics for 1985, 1995, 2005 and 2014 (an arbitrary lag of 4 years; see Table S4.4 for sensitivity to this lag period). I set production in 1985 (annual and cumulative since 1950) as the baseline index (1), and derived expected increases as a factor of these values in 1995 (1.8–2.0 for annual and cumulative production, respectively), 2005 (3.0–3.8) and 2014 (4.1–6.1); altering the duration of the lag from 2–10 years made little difference to these growth factors (Table S4.4).

Results

Temporal variation in plastic incidence and loads

A total of 4 672 skua pellets containing the remains of a single petrel were recorded across all years and months (Table 4.1). Within the focal sampling months (September-December), 14 039 plastic items were recorded in 3 727 skua pellets (Table 4.1). Great Shearwaters had the overall greatest frequency of ingestion, followed by White-faced Storm Petrels, Broad-billed Prions and *Fregetta* storm petrels (Table 4.1). Of the pellets containing non-avian prey remains (< 5% of all pellets; mainly the shells of goose barnacles *Lepas* spp., fish bones and fur/bones from Subantarctic Fur Seals *Arctocephalus tropicalis*), only one goose barnacle pellet contained plastic (a green thread). The proportions of birds with ingested plastics (September-December) peaked in 2009–2014 for *Fregetta* and White-faced Storm Petrels, in 2018 for adult Broad-billed Prions and in 1987–1989 for Great Shearwaters, but these decadal changes were not statistically significant (Table 4.2). Broad-billed Prion chicks had a higher frequency of occurrence of plastic ingestion (87%) than adults (56%) but there was no significant difference in the proportions of adults or chicks containing ingested plastics among time periods (Table 4.2). There has been little change in the proportions of storm petrels and prions containing ingested plastic over the last decade, whereas Great Shearwaters showed a 16% reduction (Table 4.2).

Table 4.1 Total number of single-prey Brown Skua pellets collected on Inaccessible Island between 1987 and 2018 containing avian prey items (n), proportions containing plastic, and mean plastic load (number of items per bird) for the four petrel taxa (all months), and during September-December separated into all birds, and for birds containing plastic.

Taxa	**n**	**% with plastic**	**Mean load per bird**	
			All birds	**Plastic only**
September-December				
Fregetta storm petrels	1506	35%	0.71	2.06
White-faced Storm Petrel	914	74%	3.80	5.16
Broad-billed Prion				
adults	783	56%	3.15	5.66
chicks*	234	87%	12.68	14.62
Great Shearwater				
adults	290	80%	13.99	17.49
All months				
Fregetta storm petrels	2095	30%	0.59	1.98
White-faced Storm Petrel	1065	69%	3.56	5.13
Broad-billed Prion				
adults	854	55%	3.14	5.71
chicks*	280	85%	12.06	14.13
Great Shearwater				
adults	317	77%	13.10	17.02
chicks*	61	56%	2.30	4.12

*identity less certain than adult birds; may include more than one chick (especially young prions)

Table 4.2 Long-term trends in the proportions of four petrel taxa containing ingested plastic based on their remains in Brown Skua pellets collected from September to December at Inaccessible Island, Tristan da Cunha, from 1987 to 2018. n = number of birds sampled.

Species	**1987–1989 % (n)**	**1999–2004 % (n)**	**2009–2014 % (n)**	**2018 % (n)**	**Significance**
Fregetta SP	32% (509)	29% (339)	40% (384)	38% (274))	$\chi^2 = 7.62$, df = 3, P = 0.05
White-faced SP	67% (307)	76% (170)	80% (242)	75% (195)	$\chi^2 = 3.38$, df = 3, P = 0.34
Broad-billed Prion					
adults	47% (219)	56% (206)	59% (198)	62% (160)	$\chi^2 = 4.06$, df = 3, P = 0.26
chicks	no data	92% (38)	83% (72)	87% (124)	$\chi^2 = 0.22$, df = 2, P = 0.89
Great Shearwater	90% (21)	85% (54)	89% (62)	73% (153)	$\chi^2 = 1.94$, df = 3, P = 0.59

*SP = storm petrel

Fregetta storm petrels had the smallest mean ingested plastic loads, with slightly larger loads in 2009–2014 than in 1999–2004 (Table 4.3; Fig. 4.1A). Significant inter-annual variation in plastic loads was evident from the GAM output results ($Z = -6.93$, $df = 8$, $P = 0.03$), but trends were unapparent from the plot (Fig. 4.2; Table S4.5), and the model only explained 0.4% of the variation observed. On average, White-faced Storm Petrels contained nearly four times more plastic items than *Fregetta* storm petrels, and had significantly larger loads in 2009–2014 than in 1999–2004 or 1987–1989 (Table 4.3; Fig. 4.1B). Loads decreased from 2009–2014 to 2018, but no overall clear trends were apparent across time periods (Fig. 4.1B). Inter-annual variation was evident from the GAM output ($Z = 30.37$, $df = 8$, $P < 0.001$), with the plot illustrating a decreasing trend after 1987, an increasing trend towards 2000, and then more or less stabilized towards 2018 (Fig. 4.2; Table S4.5), but the model only explained 3.8% of the variation observed. Overall, adult Broad-billed Prions contained fewer plastic items than White-faced Storm Petrels, and loads were only marginally significantly different ($P = 0.04$) between periods, with 1999–2004 and 2009–2014 having a few birds with large loads, but the adjusted post-hoc Dunn's test failed to indicate when these differences occurred (Table 4.3; Fig. 4.1C). Inter-annual variation was evident from the model ($Z = 19.59$, $df = 8$, $P < 0.001$), and showed an increasing trend up to 1999, followed by a decreasing and then stabilizing trend towards 2018 (Fig. 4.2; Table S4.5), but the model only explained 5.2% of the variation observed. Prion chick remains contained more plastics than adults sampled in the same year: chicks in 1999, 2009 and 2018 contained 10.3, 17.1 and 10.9 plastic items, respectively, compared to 4.1, 3.5 and 2.5 plastic items in adults (Mann-Whitney *U* test; $P < 0.001$ for all, Table 4.3; Fig. S4.1). Mean plastic loads did not differ between time periods for chicks (Table 4.3). Great Shearwaters contained the largest plastic loads of all four taxa, and were also the only species with a tendency for larger loads in 2018 (17.0 ± 23.2 items per bird) compared to previous time periods (10.8 ± 14.5 items per bird; Fig. 4.1D). However, this difference was not statistically significant. Across years, near-significant variations, and a slight increasing trend, was detected (Fig. 4.2D; $Z = 30.1$, $df = 6$, $P = 0.05$), driven by the large loads in 2018 (Table S4.5), but the model only explained 1.2% of the variation. The increases in average plastic loads across the 30 year study period ranged from 2% in Broad-billed Prions to 58% in Great Shearwaters, substantially less than those expected from the 4-6 fold increase in global plastic production over this period (Fig. S4.2; Table S4.4).

Table 4.3 Mean (± SD) plastic loads (number of items per bird) per species and time period and Kruskal-Wallis test results for all birds (including those with zero ingested plastics). Significant differences are indicated in bold.

Species	1987–1989	1999–2004	2009–2014	2018	Significance (KW)
Fregetta storm petrels					
	0.6 ± 1.3	0.6 ± 1.5	0.9 ± 1.6	0.8 ± 1.4	**χ^2 = 12.50, df = 3, P < 0.05**
White-faced Storm Petrel					
	3.5 ± 5.4	3.2 ± 5.2	4.5 ± 5.2	4.0 ± 5.2	**χ^2 = 16.56, df = 3, P < 0.001**
Broad-billed Prion					
adults	2.4 ± 4.5	4.1 ± 7.2	3.5 ± 6.8	2.5 ± 3.7	**χ^2 = 8.33, df = 3, P < 0.05**
chicks	no data	10.3 ± 10.1	17.1 ± 21.9	10.9 ± 11.0	χ^2 = 2.77, df = 2, P = 0.25
Great Shearwater					
	10.7 ± 13.3	12.6 ± 17.4	9.3 ± 12.1	17.0 ± 23.2	χ^2 = 1.59, df = 3, P = 0.66

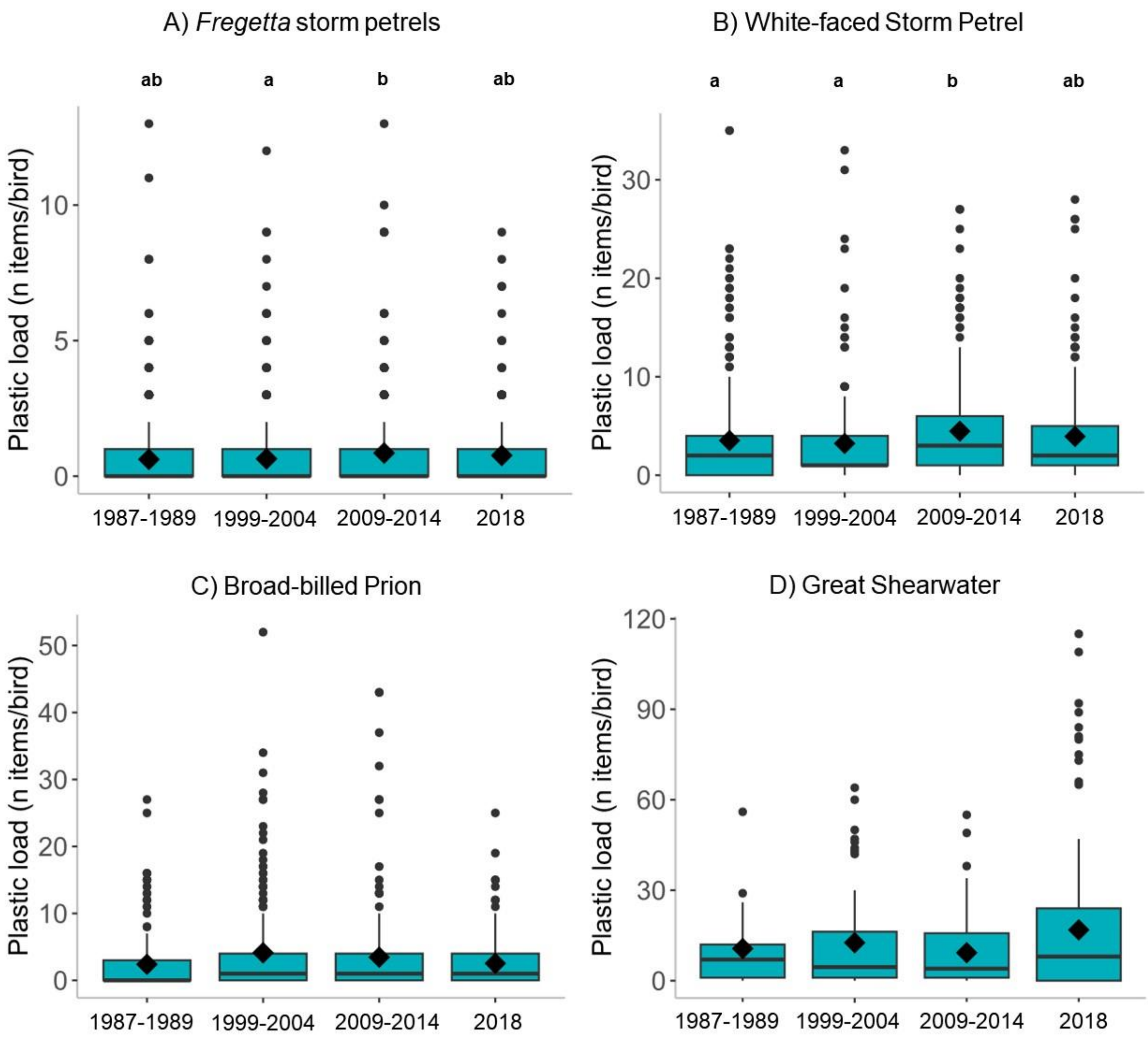


Fig. 4.1 Boxplots indicating the median (black line), mean (diamond), interquartile range (teal boxes), values within 1.5 times inter-quartile range (whiskers) and outliers (values outside 1.5 times the inter-quartile range) of the plastic load for the four adult taxa across time periods. Letters above boxplots indicate which time periods were significantly different (not sharing any letters; post hoc Dunn's test results; $P < 0.05$). Note differences in scaling on y-axes.

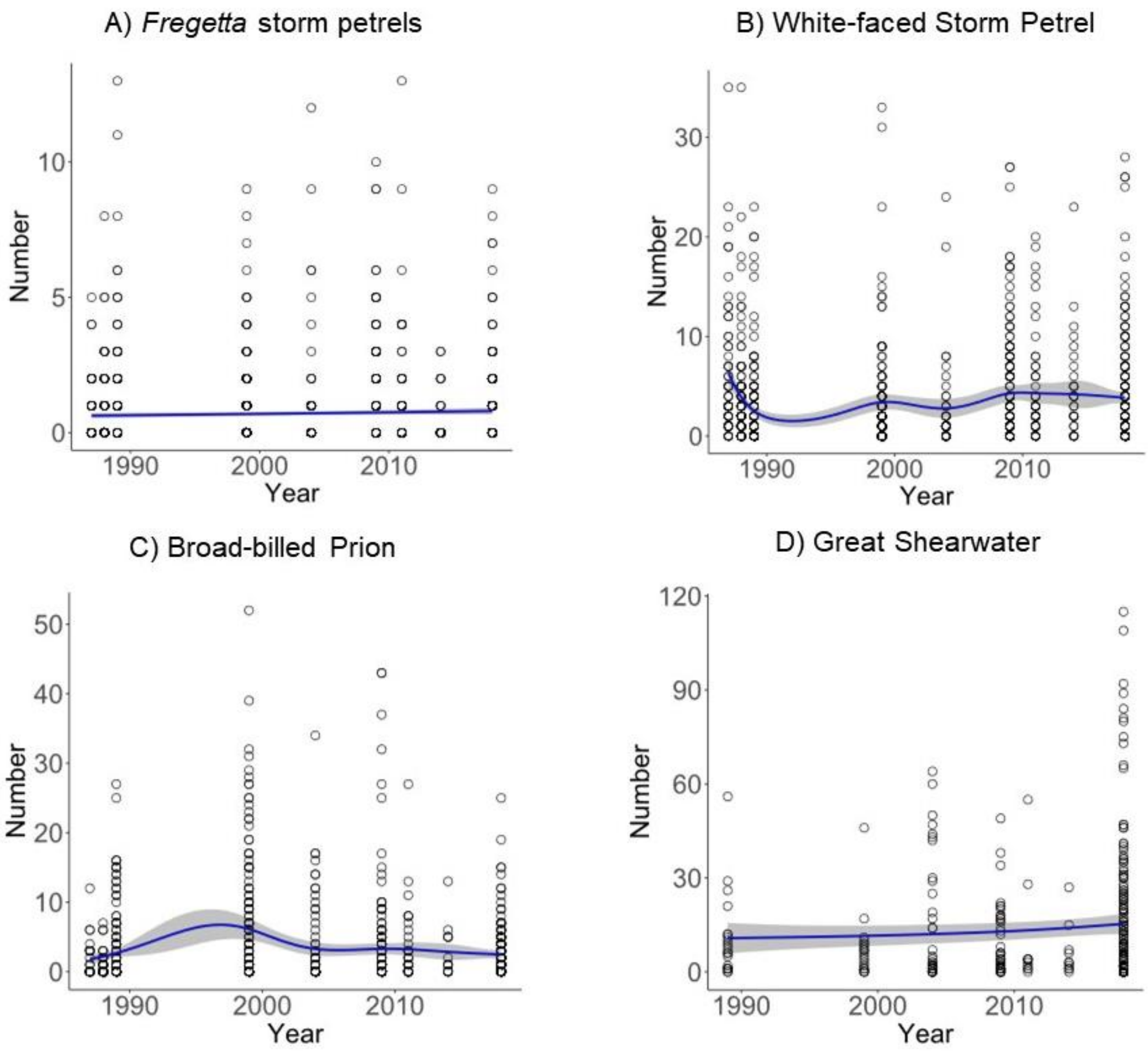


Fig. 4.2 Number of plastic items ingested per bird (plastic load) in each sampling year for the four focal study taxa sampled at Inaccessible Island. The blue line is the GAM fitted curve and the grey is the 95% confidence interval. Note differences in scaling on y-axes.

Temporal changes in birds with large plastic loads

Plastic loads among individuals were strongly right-skewed in all species (Fig. 4.1). The maximum number of items recovered from individual skua pellets were recorded in earlier time periods for *Fregetta* storm petrels (13 items in 1987–1989 and again in 2009–2014), White-faced Storm Petrels (35 items in 1987–1989) and adult Broad-billed Prions (52 items in 1999–2004), but the greatest loads for Great Shearwaters were recorded more recently (115 items in 2018). In order to assess whether the proportions of birds with large plastic loads increased over time, I checked the proportion of birds with loads approaching the maximum load recorded per species. There was no change in the proportion of *Fregetta* storm petrels containing > 5 plastic items ($\chi^2 = 1.86$, df = 3, P = 0.60) or White-faced Storm Petrels with > 10 items ($\chi^2 = 2.81$, df = 3, P = 0.42) over the four time periods (Fig. S4.3). The proportion of adult prions containing ≥ 20 plastic items was 1% in the late 1980s, 4% in 1999–2004 and 2009–2014 and 1% in 2018 ($\chi^2 = 8.41$, df = 3, P < 0.05; Fig. S4.3). Proportions of Great Shearwaters containing ≥ 50 plastic items doubled from 3% in 1987–2014 to 7% in 2018, but it was not significantly higher ($\chi^2 = 2.55$, df = 1, P = 0.11 pooling samples from 1987–2014; Fig S4.3).

Temporal variation in ingested plastic types

As in Chapter 3, interspecific variation in the type of plastics ingested was evident, most notably in White-faced Storm Petrels where, averaged across each period, 35% of ingested plastics were industrial pellets, whereas *Fregetta* storm petrels (16%), adult Broad-billed Prions (21%) and Great Shearwaters (16%) ingested lower proportions of pellets (Table S4.2). *Fregetta* storm petrels ingested more threads (7%) than the other species (all ≤ 1%; Table S4.2). All four taxa showed a highly significant decrease in the ratios of industrial pellets ingested per time period (Fig. 4.3; Table S4.2), but the rate of decrease has slowed in the last decade for White-faced Storm Petrels, while *Fregetta* storm petrels exhibited a 1% increase from 2009–2014 (Fig. 4.3). The number of industrial pellets also showed a highly significant decrease per year over the study period in *Fregetta* storm petrels (r = -0.84, P < 0.05), White-faced Storm Petrels (r = -0.75, P < 0.05), Broad-billed Prions (r = -0.69, P < 0.05) and Great Shearwaters (r = -0.92, P < 0.05; Fig. 4.4). The number of other plastic items per bird have increased significantly over time in *Fregetta* (r = 0.71, P < 0.05) and White-faced Storm Petrels (r = 0.90, P < 0.001), but not in Broad-billed Prions (r = 0.52, P = 0.15) and Great Shearwaters (r = 0.43, P = 0.34; Fig. 4.4).

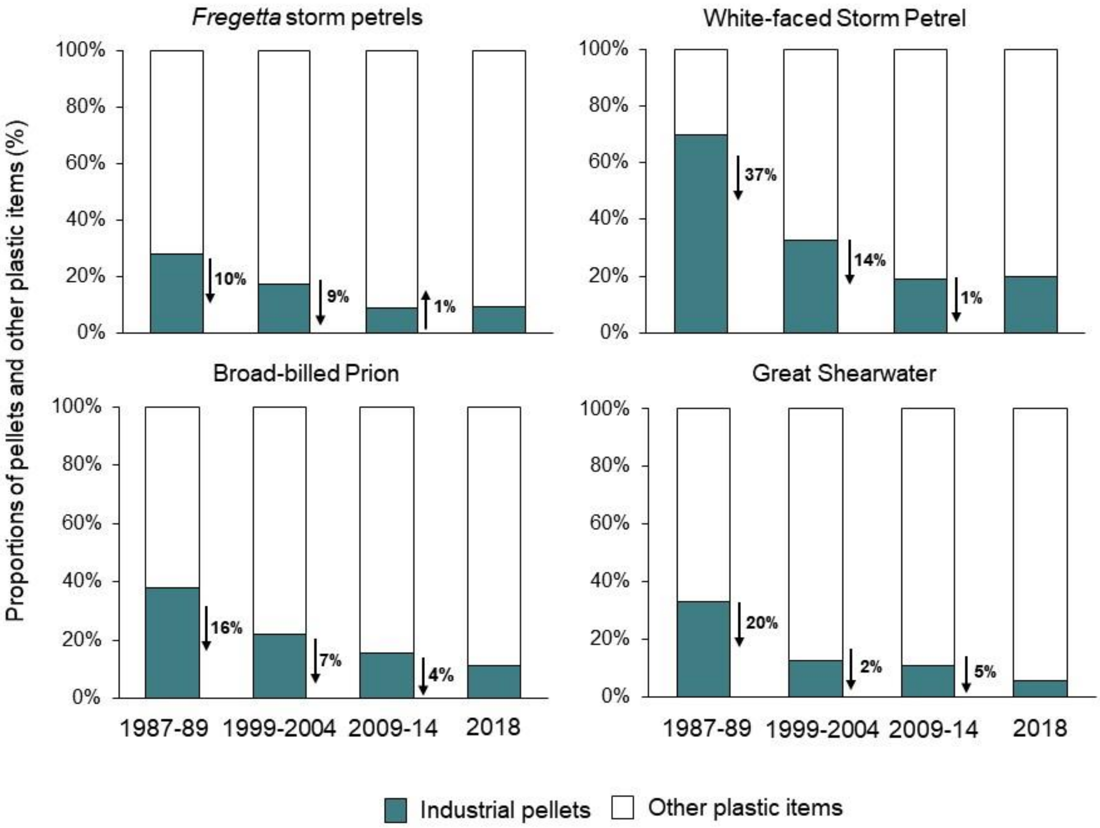


Fig. 4.3 Changes in the proportion of industrial pellets among all plastic items ingested by four seabird taxa based on their remains in skua pellets collected at Inaccessible Island, Tristan da Cunha, from 1987 to 2018. Black arrows indicate the absolute % change in industrial pellets between time periods.

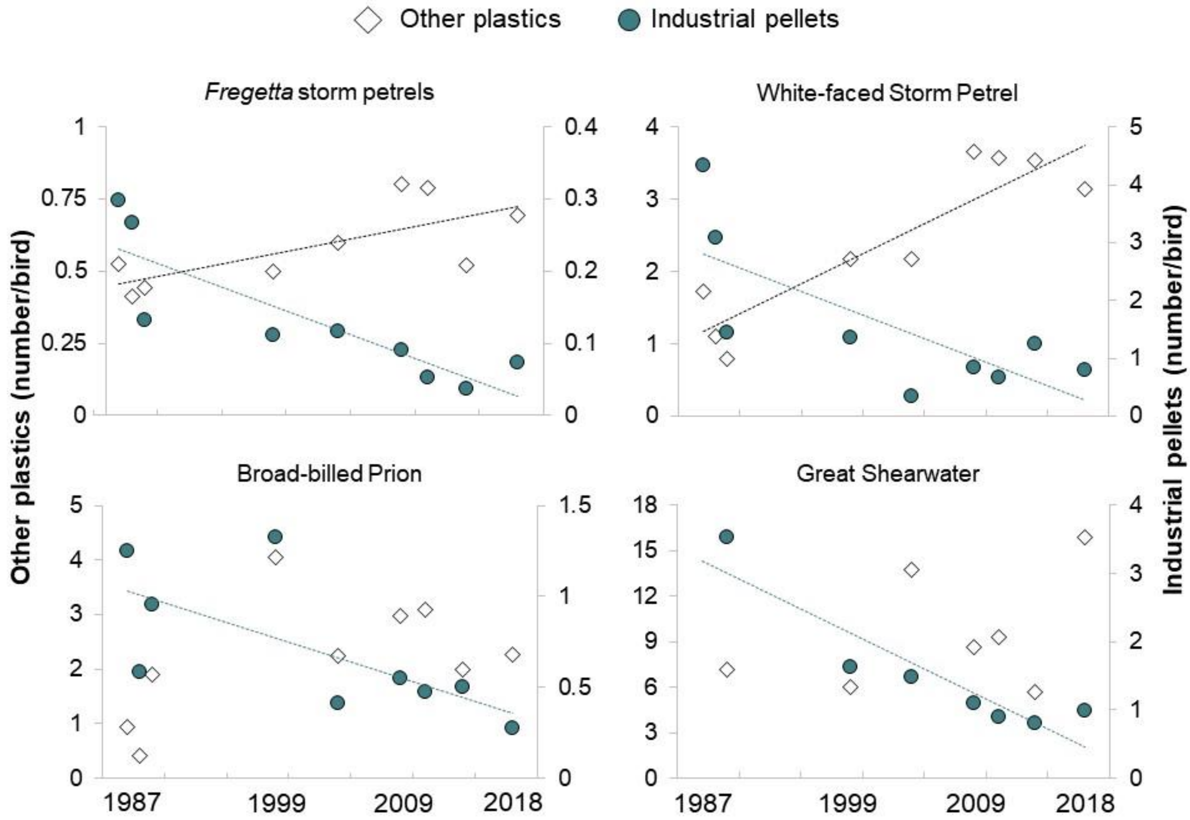


Fig. 4.4 Correlations between the number of other plastic items (primary axis) and industrial pellets (secondary axis) ingested per bird over time for four seabird taxa breeding on Inaccessible Island. Trendlines indicate significant correlations between plastic type and year.

Discussion

How reliable are skua pellets to track changes in plastic loads?

Several studies have used skua pellets to assess plastic loads in prey species and to monitor trends in plastic pollution (Ryan and Fraser, 1988; Ryan, 2008; Hammer et al., 2016; Ibañez et al., 2020). They assume that prey items in pellets can be identified accurately (which requires a good reference collection of prey remains such as bones and feathers), that skuas seldom ingest plastics directly from the marine environment, and that there is little contamination between meals. The paucity of plastics in pellets containing non-avian prey and the fact that almost all pellets containing plastics had only avian prey remains, with consistent differences in the types and amounts of plastic among prey species, gives confidence that the skuas on Inaccessible Island rarely ingest plastics directly and confirm that there is little transfer of plastic between prey items (cf. Votier et al., 2001). The thread found in a pellet containing goose barnacle shells may have been ingested while stripping barnacles from drifting debris.

Among seabirds, factors such as the age, breeding status and sampling months may influence ingested plastic loads (Ryan et al., 2009a; Ryan, 2016; Tulatz et al., 2023). Sampling plastics from skua pellets means that there is no information on the sex or health status of the bird containing the plastic, but collecting pellets at seabird breeding colonies mostly samples breeding or pre-breeding seabirds. The variance in plastic loads can be further reduced by limiting sampling to the early part of the breeding season, before adults transfer plastic to their chicks. In this chapter, the proportions of adult birds containing ingested plastics were lower across all sampling months than when compared to data collected from September to December (Table 4.1), which was one of the reasons why I limited temporal comparisons to samples collected during pre- or early breeding months (Fig. S3.1). Studies assume that skua pellets contain all plastics ingested by the seabird prey, even though the smallest items might not be incorporated into the pellet (excreted by the skua), lost during collection or overlooked during processing (Ryan and Fraser, 1988). However, in Chapter 3 I found that plastics removed directly from seabird carcasses exhibited similar sizes and masses to those found in skua pellets, supporting the use of skua pellets as a tool for monitoring marine plastics.

Prey size can also influence plastic loads as some skua pellets contain the remains of multiple prey species and have to be discarded if the goal is to document plastic loads in the prey species. This is fairly easy to do for adult birds, but can be challenging for young chicks, where it is harder to detect multiple remains from the poorly developed bones (especially for prions, which have smaller chicks than Great Shearwaters), and can potentially include plastics from more than one individual, thus overestimating ingested plastic loads. This might account for the larger plastic loads in Broad-billed Prion chicks compared to adults. However, chicks obtain stored plastics from both parents, and thus often contain larger plastic loads than adults (Ryan, 1988; Rodríguez et al., 2012; Krug et al., 2021; Tulatz et al., 2023). A more challenging issue is when the remains (and ingested plastics) of large prey species are regurgitated in multiple pellets, underestimating plastic loads per bird (Ryan, 2008; Provencher et al., 2019). Taking into consideration these limitations posed by the larger species, I have more confidence in the data obtained from storm petrels than for prions and shearwaters, when using this sampling method.

Interspecific variation in plastic loads

The consistent interspecific differences in plastic loads across all sampling periods presumably reflect differences in foraging ecology and range among species, and further support the accuracy with which skua pellets indicate plastic loads in their prey species. White-faced Storm Petrels contained four times more plastic than *Fregetta* storm petrels, even though they are similar in terms of size, diet and behaviour (Harrison, 1983; Ryan, 2007). White-faced Storm Petrels mostly forage north of the Tristan archipelago where they are exposed to greater litter densities associated with the South Atlantic Gyre (Eriksen et al., 2014; Ryan et al., 2014; Clark et al., 2023; Ryan, 2023b; Chapter 3; Fig. S3.4). *Fregetta* storm petrels most likely forage south of the Sub-tropical Front (Chapter 3; Fig. S3.4), in areas with lower litter densities and plastic exposure risk (Ryan, 1988; Eriksen et al., 2014; Ryan et al., 2014; Clark et al., 2023; Ryan, 2023b). White-faced Storm Petrels sampled from gull pellets at colonies in the northeast Atlantic Ocean between 2013-2015 had a slightly higher incidence of ingested plastic (79%) and larger mean plastic loads (5.3–5.7 items per bird; Furtado et al., 2016) than in this study (69%, 3.8 items per bird), suggesting that the northeastern Atlantic waters may be more polluted than the South Atlantic. Broad-billed Prions tend to have smaller loads than other prions because they are specialist filter-feeders that target small copepods (Klages and Cooper, 1992), and are less likely to ingest plastic than prions that pick individual food items from the sea surface (Ryan, 1987a; Dell'Ariccia et al., 2017). Great Shearwaters had the greatest plastic loads of all species in this study, which is as expected given that *Ardenna* shearwaters are among the most plastic-contaminated seabirds (Ryan, 1987a; Lavers et al., 2021).

Long-term variation in plastic loads

Although plastic loads within species varied over time, there was no consistent change among average or maximum plastic loads across time periods. When comparing time periods, all three taxa confined to the Southern Hemisphere (both storm petrels and Broad-billed Prions) showed a decrease in plastic loads over the last decade, whereas loads in Great Shearwaters nearly doubled over this period. When modeled by sampling year, *Fregetta* storm petrels, White-faced Storm Petrels and Broad-billed Prions showed interannual variation but no overall clear increasing or decreasing trends since 2000 (Fig. 4.2). These findings mirror results obtained for Northern Fulmars from the Netherlands studied between the 1980s to 2012 (van

Franeker and Law, 2015). Great Shearwaters showed a near-significant increasing trend in plastic loads when comparing years, but the model accounted for at most 1% of the variance in plastic loads. Great Shearwaters migrate annually across the Equator to spend the austral winter in the northwest Atlantic Ocean (Powers et al., 2022; Chapter 3; Fig. S3.4). This area borders heavily industrialized coastal regions, where human population densities, litter concentrations, and plastic exposure risks are high (Eriksen et al., 2014; Robuck et al., 2022; Clark et al., 2023). Samples were collected at the start of the breeding season (September-December), when Great Shearwaters return from their migration, having spent ~4 months foraging in the North Atlantic. Data were collected before their eggs hatch in early January, so adults would not have had the opportunity to offload plastic to chicks prior to collection (Ryan, 1988; Fig. S3.1). The increase in plastic loads in Great Shearwaters recorded over the last decade could reflect increases in plastic loads in the North Atlantic (Eriksen et al., 2023). However, Robuck et al. (2022) found no significant difference in the number of plastic items ingested by adult Great Shearwaters between the North and South Atlantic. Among shearwaters collected in Massachusetts Bay there was as a modest increase in the mass of ingested plastic from 2010 to 2019 (22% over 9 years, based on 5 years when >15 birds were sampled per year) but no change in the number of ingested plastic items (Robuck et al., 2022). The large within-species variation in plastic loads in petrels might contribute to masking temporal variation in plastic loads, just as the highly heterogeneous distribution of floating plastic compromises the use of net samples to monitor changes in the density of floating plastics at sea (Ryan et al., 2009a).

The lack of a consistent change among average or maximum plastic loads in all four study species over the last three decades is in stark contrast to the ongoing growth in plastic production over this period (Fig. S4.2). However, other long-term studies also have not detected marked increases in the amount of plastic at sea since the 1980s (Law et al., 2010; Thompson, 2015; Beer et al., 2018). And although several recent studies have reported plastic concentrations at sea rising in line with global plastic production (Lebreton et al., 2018; Wilcox et al., 2020; Eriksen et al., 2023), they mainly focused on the North Pacific and North Atlantic, with little sampling conducted in the Southern Hemisphere. The few long-term studies of plastic in biota since the 1980s also have produced mixed results (Vlietstra and Parga, 2002; Ryan, 2008; van Franeker and Law, 2015; Courtene-Jones et. al., 2019; Lavers et al., 2021; van Franeker et al., 2021). Mrosovsky et al. (2009) reported a rapid increase in the incidence of plastic ingestion by turtles from the 1960s to 1980s, but stable to decreasing trends thereafter (1980s–2000s). Beer et al. (2018) found no change in the concentration of microplastics in planktivorous fish in the Baltic Sea over a 30 year period, mirroring a similar lack of trend in the density of microplastics at sea in this region. van Franeker and Law (2015) found an increase in the number of plastic items in Northern Fulmars from the 1980s to the 1990s, but with loads stabilizing since then. Lavers et al. (2021) found no trends in plastic loads in Flesh-footed Shearwaters *Ardenna carneipes* from 2005 to 2021, and Baak et al. (2020) found a decrease in the number of plastic items ingested by fulmars in the Canadian Arctic between 2008 and 2018, but results for fulmars often vary with sampling region (Mallory et al., 2006; van Franeker et al., 2011; Kühn et al., 2021). Loads in Cory's Shearwaters *Calonectris borealis* sampled in the northeastern Atlantic increased from 2015 to 2022 (Rodríguez et al., 2024), but this is a short time line compared to the other studies cited above. Compared to these petrels, the amount of plastic regurgitated by some populations of albatrosses and giant petrels, which ingest larger items than petrels (e.g. bottle lids, rigid items and plastic bags), has increased over the last few decades at Southern Ocean breeding colonies (e.g. Phillips and Waluda, 2020).

Have initiatives to reduce losses of industrial pellets been successful?

The first records of large numbers of industrial pellets at sea were from the 1970s (Carpenter and Smith, 1972; Colton et al., 1974), when massive densities of stranded pellets were recorded on beaches even in areas with little or no plastic industry nearby (Gregory, 1978; Gregory, 1983). In the early 1990s, industry-led awareness drives were established to reduce the loss of pellets into the environment (e.g. Operation Clean Sweep, www.opcleansweep.org). Since then, a reduction in the proportion of industrial pellets at sea has been reported from sea surface net tows (Law et al., 2010; Morét-Ferguson et al., 2010; van Franeker and Law, 2015; Chapter 3; Fig. S3.2A) and seabird ingestion studies (Vlietstra and Parga, 2002; Ryan, 2008; van Franeker et al., 2011; Petry and Benemann, 2017). In this chapter, I found that this decreasing trend has continued in the proportion (Fig. 4.3) and number of industrial pellets across all four taxa (Fig. 4.4), albeit at a slower rate in the last decade. These decreases could result in part from an increase in the density of other plastic types, but increases in other plastics were only recorded in storm petrels (Fig. 4.4). van Franeker and Law (2015) found that from 2000, user plastic numbers stabilized in Northern Fulmars, while industrial pellet numbers have decreased since the 1980s. van Franeker et al. (2021) also failed to detect significant trends in the mass of other plastics in Northern Fulmars in the Canadian Arctic from 2002–2018, and Petry and Benemann (2017) failed to detect strong changes in the intensity of other plastics ingested by White-chinned Petrels *Procellaria aequinoctialis* in southern Brazil from the 1990s to 2014. In addition, given the lack of clear increases in the total amount of ingested plastic, or in the frequency of large ingested plastic loads, it appears to be unlikely that the decrease in industrial pellets have resulted from an overall increase in other plastics (Ryan, 2008; this study). I infer that the reduction in the proportion and number of industrial pellets in seabirds indicates the probable success of mitigation measures put in place to reduce industrial pellet leakage into the marine environment, and highlights the value of using seabirds as bioindicators to detect and report temporal trends in the types of litter floating at sea (van Franeker and Law, 2015).

Why are we failing to detect consistent increases in floating plastic at sea?

Since the 1950s, large amounts of mismanaged plastic waste have leaked into marine environments, and continue to do so. And once at sea, plastic litter may drift for years (Lebreton et al., 2018). Given the steady increase in global plastic production over the last 75 years, we would therefore expect to find an increase in the amount of floating plastic recorded at sea (Fig 1.2). The magnitude of this increase depends on multiple factors related to the amount entering marine environments and its fate once in the sea (Koelmans et al., 2017; Lebreton et al., 2019). Assuming the amount of plastic at sea is linked directly to the growth in plastic production, we can estimate the expected increase over time. In terms of the small floating plastic items ingested by petrels, we would expect some lag between plastic leakage and ingestion to allow time for the fragmentation of large user items into secondary micro and mesoplastics. If global plastic production was the primary driver of plastic at sea, we would expect the density of floating plastic at sea to have increased 4-6 times from 1989 to 2018 (Fig. S4.2; Table S4.4). This is in sharp contrast to the limited change in plastic loads in petrels breeding on Inaccessible Island over this period, and in the other long-term studies discussed above. Either petrels do not track the density of floating plastic at sea, or the density of plastic in the South Atlantic Ocean has not increased in line with global plastic production.

One challenge to using ingested plastic loads in petrels to monitor floating plastic at sea is that it remains unclear whether seabirds regurgitate plastics outside of their breeding season (Ryan, 1988). It has been assumed that non-breeding petrels only lose plastics through excretion, which results in long retention times until they are small enough to pass into the intestine (e.g. Furness, 1985; Ryan, 2015). However, Terepocki et al. (2017) recorded plastic regurgitation by captive northern fulmars during rehabilitation after stranding ashore. If petrels occasionally regurgitate plastics and other indigestible prey remains while at sea (for which there is no evidence to date), it would potentially invalidate their use as monitors of floating plastics, because the lack of an increase in plastic loads might simply reflect more frequent regurgitation. Two facts suggest that this is not the case. First, ingested plastic loads are strongly right skewed in petrels (Ryan, 2016), which does not indicate that there is a threshold above which birds regurgitate plastic (although this could vary among individuals). And second, the size of plastics passing through the pyloric sphincter into the intestine is larger than previously thought (at least 5 mm long in northern fulmars, Terepocki et al., 2017), so loss through excretion may be appreciably faster than assumed (van Franeker and Law, 2015; Ryan, 2016). More experimental evidence is needed to assess turnover rates of ingested plastics in seabirds.

If petrels are reliable indicators of plastic densities at sea, we have to consider the possibility that changes in the amount of floating plastic at sea do not mirror the growth in plastic production. If we compare global estimates of plastic leakage into the ocean to the amounts recorded with surface nets, we find that they greatly surpass the estimated mass of plastic floating at sea. (Eriksen et al., 2014; Jambeck et al., 2015; Lebreton et al., 2017; Galgani et al., 2021). This might simply be because global models overestimate leakage (Lebreton et al., 2019; Verster and Bouwman, 2020). The amount at sea might also be underestimated due to small sample sizes and high variance among samples (Lebreton et al., 2019). Refining the estimates of leakage of mismanaged waste into the marine environment, and of plastics floating at sea, could close the apparent gap between these estimates (Ryan et al., 2020a). Sedimentation of plastic from the sea surface by biofouling might also account for many items disappearing from the surface of the ocean, especially small and thin plastic sheets that have little inherent buoyancy (Fazey and Ryan, 2016; Lebreton et al., 2019). For example, in Chapter 3 I found that 50% of flexible items that were sampled with nets were record within ~200 km of the coast (Fig. S3.2B). Drifting micro- and mesoplastics may also be removed by seabirds that mistake it for food (Shaw and Day, 1994; Ryan 2008; van Franeker et al., 2011; Avery-Gomm et al., 2018), but this is unlikely to occur at a scale that could significantly contribute to this mismatch. This disparity may also be because most litter reaching the sea from land-based sources strands on beaches (Maclean et al., 2021; Ryan and Perold, 2021), where large amounts are removed by beach cleaning efforts. In addition, growing awareness of the dangers posed by plastics in the environment over the last three decades have resulted in a multitude of interventions to reduce plastic leakage (Willis et al., 2018). These range from the implementation of MARPOL Annex V banning the dumping of plastic at sea in 1989, through installation of interception devices to capture litter in rivers and storm drains, to ever increasing efforts (both formal and informal) to collect litter stranded on beaches (Sidek et al., 2016; Haarr et al., 2020; Gkanasos et al., 2021). Together, these efforts might also explain why plastic loads in seabirds (and by extension, floating at sea) have not increased as fast as the growth in plastic production.

Conclusions

The standardised sampling, processing and reporting protocol followed in this chapter allowed me to non-invasively assess long-term changes in plastic loads in four petrels breeding in the South Atlantic Ocean while controlling for age class, season and location. I found that, when using this method, storm petrels are the most reliable biomonitors (due to the greater probability of being swallowed whole by skuas), and no consistent changes in their plastic loads among the four time periods were apparent. This chapter provides further support for long-term reductions in the numbers of industrial pellets at sea, as also reported in Chapter 3, presumably at least in part due to mitigation measures to reduce the leakage of pellets into the environment. This data can support the maintenance of current policies and the development of new ones to further limit industrial pellet leakage into marine ecosystems. The lack of marked increases in plastic loads in four seabird species over 30 years, despite an estimated 4-6 fold increases in global plastic production over this time, is consistent with other studies tracking the density of floating plastic at sea. Such long-term monitoring studies highlight the value of seabirds as sentinels of ocean health by tracking changes in plastic loads in the marine environment. Continued monitoring of plastic pollution trends is important to assess the efficacy of intervention measures and understanding the dynamics behind the densities recorded at sea.

Supplementary materials: Chapter 4

Table S4.1 Number of skua pellets examined per month (September-December) and per year for the four main taxa in this study.

Year	**Month**	***Fregetta* storm petrels**	**White-faced Storm Petrel**	**Broad-billed Prion** adults	chicks	**Great Shearwater**
1987	Oct	57	59	20	no data	no data
1988	Oct	90	85	46	no data	no data
1989	Oct	68	58	52	no data	10
	Nov	103	51	31	no data	9
	Dec	191	54	70	no data	2
1999	Nov	137	86	63	17	14
	Dec	107	34	48	21	5
2004	Nov	95	50	95	no data	35
2009	Sep	68	44	30	4	7
	Oct	77	49	76	22	5
	Nov	99	63	42	46	30
2011	Sep	113	58	34	0	10
2014	Sep	27	28	16	0	10
2018	Sep	140	79	26	0	44
	Oct	93	75	80	55	72
	Nov	41	41	54	69	37
Total		1 506	914	783	234	290

Table S4.2 Summary of the sample sizes, number and proportions of birds with plastics, total number of plastics and the total number and proportions of industrial pellets, fragments and threads recorded for the four seabird taxa across the four time periods. The chi–square test results comparing changes in the ratios of plastic types (fragments and threads pooled) recorded between time periods are included in parentheses for each taxon.

Species	n birds	n (%) with plastic	Total no. plastics	Industrial pellets	Fragments	Threads
***Fregetta* storm petrels** (χ^2 = 52.92, df = 3, P < 0.001)						
1987–1989	509	165 (32%)	316	89 (28%)	217 (69%)	10 (3%)
1999–2004	339	99 (29%)	217	38 (18%)	157 (72%)	22 (10%)
2009–2014	384	154 (40%)	328	29 (9%)	277 (84%)	22 (7%)
2018	274	103 (38%)	210	20 (10%)	172 (82%)	18 (9%)
Total	**1 506**	**521**	**1 071**	**176 (16%)**	**823 (77%)**	**72 (7%)**
White-faced Storm Petrel (χ^2 = 751.76, df = 3, P < 0.001)						
1987–1989	307	205 (67%)	1080	755 (70%)	325 (30%)	0 (0%)
1999–2004	170	129 (76%)	548	179 (33%)	367 (67%)	2 (< 1%)
2009–2014	242	193 (80%)	1082	205 (19%)	876 (81%)	1 (< 1%)
2018	195	147 (75%)	767	153 (20%)	610 (80%)	4 (1%)
Total	**914**	**674**	**3 477**	**1 292 (37%)**	**2 178 (63%)**	**7 (< 1%)**
Broad-billed Prion (adults) (χ^2 = 123.96, df = 3, P < 0.001)						
1987–1989	219	104 (47%)	525	198 (38%)	312 (59%)	15 (3%)
1999–2004	206	116 (56%)	849	186 (22%)	660 (78%)	3 (< 1%)
2009–2014	198	116 (59%)	684	105 (15%)	577 (84%)	2 (< 1%)
2018	160	99 (61%)	406	44 (11%)	360 (89%)	2 (< 1%)
Total	**783**	**435**	**2 464**	**533 (22%)**	**1 909 (77%)**	**22 (1%)**
Broad-billed Prion (chicks) (χ^2 = 51.89, df = 2, P < 0.001)						
1999–2004	38	35 (92%)	392	100 (26%)	292 (74%)	0 (0%)
2009–2014	72	60 (83%)	1228	146 (12%)	1082 (88%)	0 (0%)
2018	124	108 (87%)	1347	164 (12%)	1180 (88%)	3 (< 1%)
Total	**234**	**203**	**2 967**	**410 (14%)**	**2 554 (86%)**	**3 (< 1%)**
Great Shearwater (χ^2 = 199.71, df = 3, P < 0.001)						
1987–1989	21	19 (90%)	224	74 (33%)	149 (67%)	1 (< 1%)
1999–2004	54	46 (85%)	681	86 (13%)	595 (87%)	0 (0%)
2009–2014	62	55 (89%)	575	63 (11%)	506 (88%)	6 (1%)
2018	153	112 (73%)	2580	150 (6%)	2410 (93%)	20 (1%)
Total	**290**	**232**	**4 060**	**373 (9%)**	**3 660 (90%)**	**27 (1%)**

Table S4.3 Mean (± SD) plastic loads (number of items per bird) per species and time period and Kruskal-Wallis test results only using samples that contained plastics. SP: storm petrel; WFSP = White-faced Storm Petrel; BBPR = Broad-billed prion; GRSH = Great Shearwater. Significant differences are indicated in bold.

Species	**1987–1989**	**1999–2004**	**2009–2014**	**2018**	**Significance (KW)**
Fregetta SP.	1.9 ±1.7	2.2 ± 2.1	2.1 ± 1.9	2.0 ± 1.6	$\chi^2 = 1.92$, df = 3, P = 0.59
WFSP	5.3 ± 5.8	4.3 ± 5.5	5.6 ± 5.3	5.2 ± 5.4	**$\chi^2 = 14.99$, df = 3, P < 0.05**
BBPR adults	5.1 ± 5.3	7.3 ± 8.4	5.9 ± 8.0	4.0 ± 4.0	$\chi^2 = 7.89$, df = 3, P = 0.05
BBPR chicks	no data	11.2 ± 9.9	20.5 ± 22.5	12.0 ± 10.2	**$\chi^2 = 8.47$, df = 2, P < 0.05**
GRSH	11.8 ± 13.5	14.8 ± 17.9	10.5 ± 12.3	23.3 ± 24.3	**$\chi^2 = 21.2$, df = 3, P < 0.001**

Table S4.4 The factor by which at sea plastic densities would be expected to increase relative to 1989 in subsequent sampling years (i.e. 2 = production doubled) based on global plastic production at varying lag intervals of 2-10 years (i.e. a lag of 4 years compares data from 1985 with 1995, 2005 and 2014). The two values for each year are based on annual and cumulative production data since 1950 (from https://PlasticsEurope.org).

Lag (years)	1999		2009		2018	
	annual	cumulative	annual	cumulative	annual	cumulative
2	1.8	2.0	2.9	3.7	3.9	5.8
4	1.8	2.0	3.0	3.8	4.1	6.1
6	1.8	2.0	3.1	3.9	4.2	6.4
8	1.7	2.1	3.1	4.0	4.3	6.8
10	1.6	2.2	2.8	4.2	4.2	7.3

Table S4.5 Mean (± SD) plastic loads (number of items per bird) ingested per species per sampling year in all birds (including those with zero ingested plastics). SP: storm petrel; WFSP = White-faced Storm Petrel; BBPR = Broad-billed Prion; GRSH = Great Shearwater.

Species	**1987**	**1988**	**1989**	**1999**	**2004**	**2009**	**2011**	**2014**	**2018**
Fregetta SP.	0.8 ±1	0.7 ±1	0.6 ±2	0.6 ±1	0.7 ±2	0.9±2	0.8 ±2	0.6 ±1	0.8±1
WFSP	6.1 ±7	4.2 ±6	2.2 ±4	3.5 ±5	2.5 ±4	4.5 ±5	4.2 ±5	4.8 ±5	4.0 ±5
BBPR adults	2.2 ±3	1.0 ±2	2.8 ±5	6.6 ±9	2.7 ±5	3.5 ±7	3.6 ±5	2.5 ±4	2.5 ±4
GRSH	no data	no data	10.7 ±13	7.7 ±10	15.3 ±20	9.7 ±11	10.2 ±18	6.5 ±8	17.0 ±23

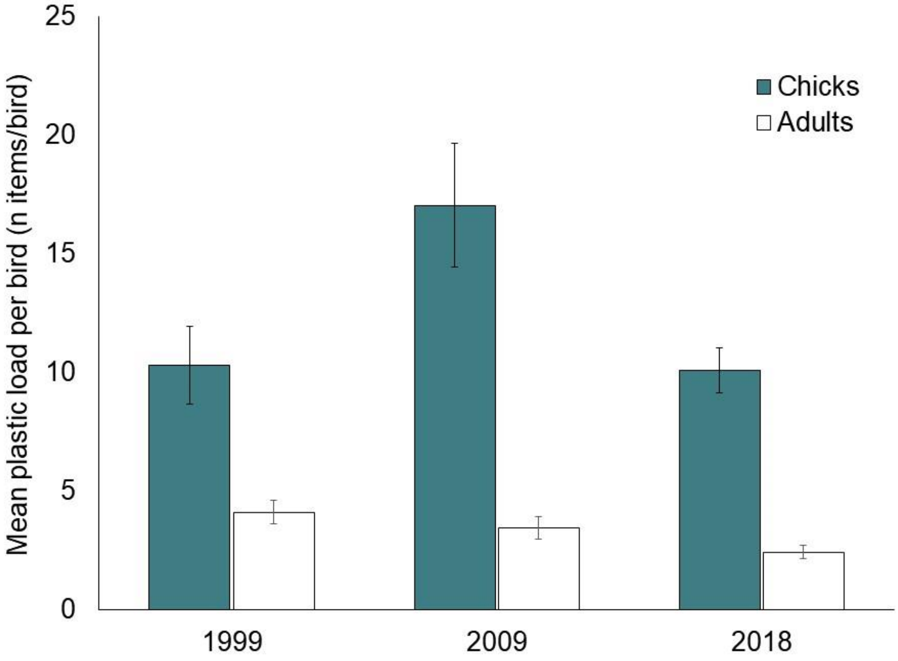


Fig. S4.1 Mean (± SE) plastic loads (number of items per bird) recorded in Broad-billed Prion chicks and adults in 1999, 2009 and 2018.

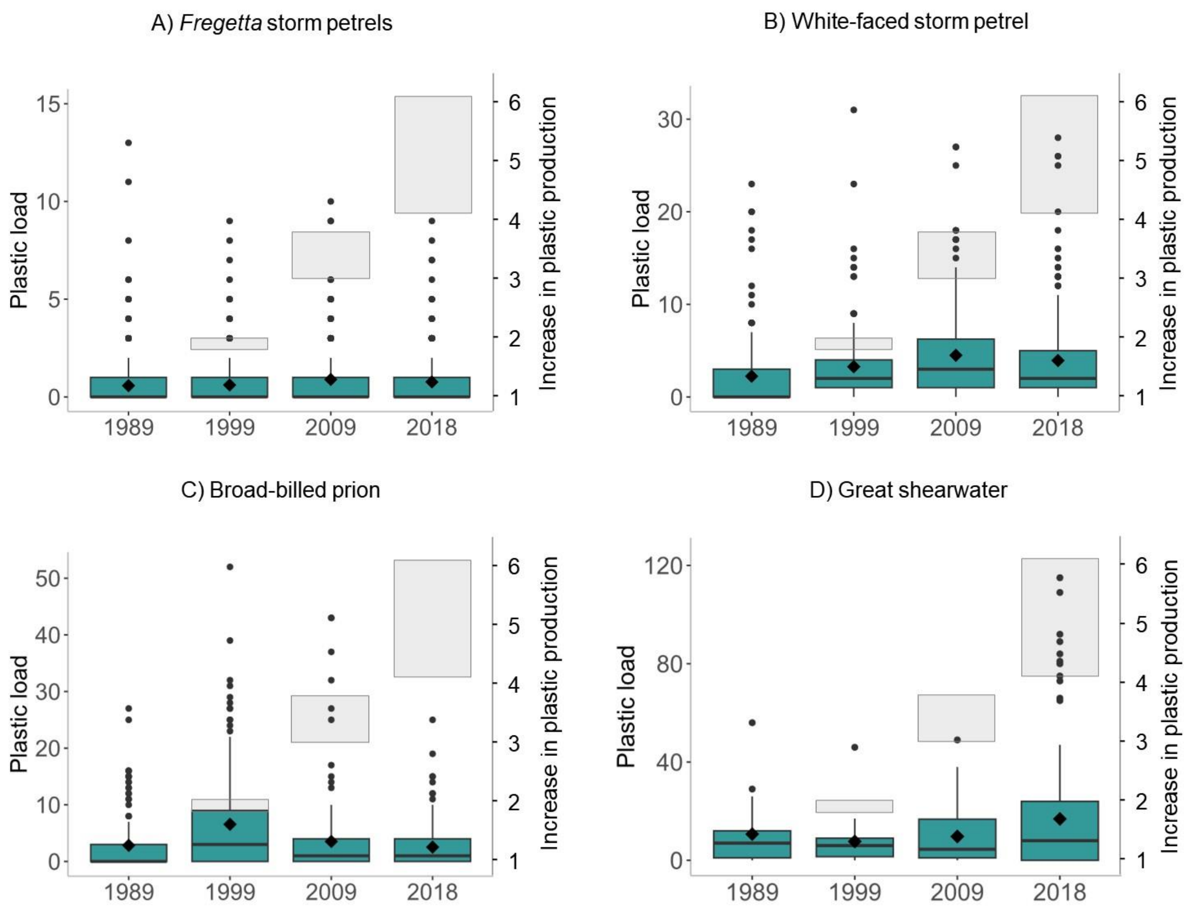


Fig. S4.2 Boxplots indicating the median (black line), mean (diamond), interquartile range (teal boxes), values within 1.5 times inter-quartile range (whiskers) and outliers (values outside 1.5 times the inter-quartile range) of the plastic loads (number of items per bird for all adult birds) for the four taxa at decadal intervals (main sampling years) relative to expected increases in plastic based on global production data (grey boxes, based on annual and cumulative production since 1950; see Methods for details. Note differences in scaling on y-axes.

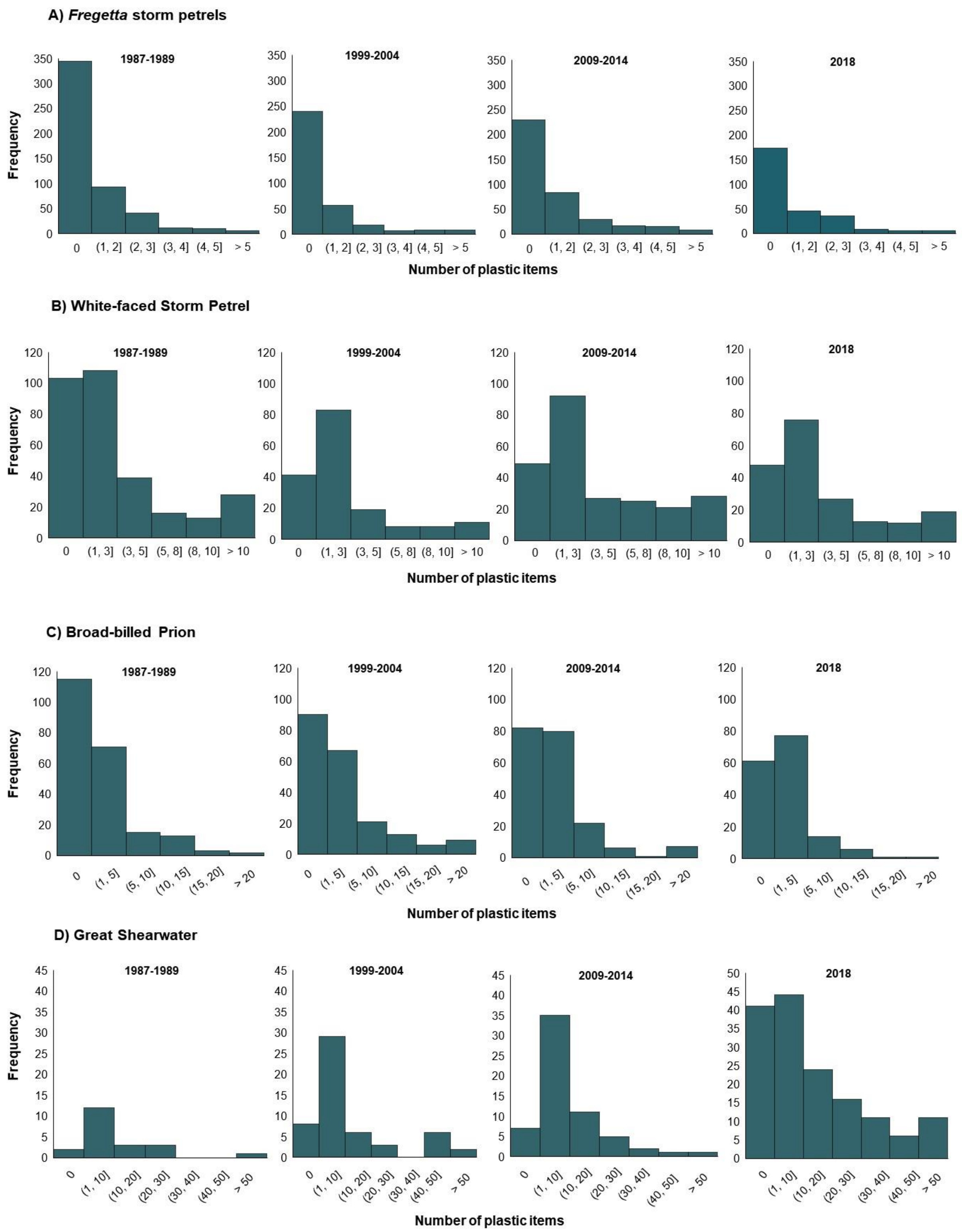


Fig. S4.3 Histograms highlighting the long-term trends in frequency distributions of plastic loads (number of items ingested per bird) in four seabird taxa based on their remains in regurgitated skua pellets collected at Inaccessible Island, Tristan da Cunha, from 1987 to 2018. Note the differences in scaling on the x-axes.

Chapter 5

Have the characteristics of plastics ingested by petrels in the South Atlantic Ocean changed since the 1980s?

Broad-billed Prion family in a cave on Inaccessible Island (photo Peter Ryan).

Abstract

Information on how the size, mass and polymer type of floating plastic items are changing over time may help improve our understanding of the complex dynamics governing plastic fragmentation rates, dispersal and longevity at the sea surface. Procellariiform seabirds directly ingest small floating plastic items (typically 2-25 mm), which they retain in their gizzards. As a result, they can be used as biomonitors to document temporal and spatial changes in the abundance and composition of plastics floating at sea. I compare the characteristics of small plastic items ingested by four seabird taxa breeding at Inaccessible Island, Tristan da Cunha, from 1987 to 2018, sampled at roughly decadal intervals. There was little evidence that the size and mass of plastic items have changed since the 1980s, but the thickness of fragments increased in two taxa that mostly forage south of the Sub-tropical Front. Nearly all plastic items were composed of PE or PP, but the ratio of PP to PE has increased over time. The limited change in the characteristics of ingested plastics over time mirrors trends in plastic densities recorded at sea.

Introduction

In the preceding chapters, I established the utility of both large and small procellariiform seabirds as monitors of marine plastic pollution (Ryan, 2008; Phillips and Waluda, 2020; Chapters 2-4). Albatrosses and giant petrels detected an increase in non-fishery related litter in the southwestern Indian Ocean over the past two decades (Chapter 2; although this may partly be due to population growth over the last decade), while the smaller petrels indicated little overall change in plastic loads between 1987 and 2018, but a decrease in the number and proportion of industrial pellets over time (Chapter 4). In Chapter 4, I discussed how our inability to detect consistent increases in floating plastics ingested by seabirds at sea over time (Law et al., 2010; Law et al., 2014; van Franeker and Law, 2015; Beer et al., 2018) may be attributed to overestimates in the amount of plastic entering our oceans (Lebreton et al., 2019; Ryan, 2020; Verster and Bouwman, 2020) due to beaching of plastics (Eriksen et al., 2014; Ryan, 2020; Onink et al., 2021; Onink et al., 2022), sedimentation of smaller items from the ocean surface due to biofouling (Cózar et al., 2014; Eriksen et al., 2014; Ryan, 2015b; Fazey and Ryan, 2016; Galgani et al., 2021), ingestion by marine animals (Barnes et al., 2009; Eriksen et al., 2014; Katija et al., 2017; Jâms et al., 2020), or a combination of these factors (Fig. 1.3). Over time, fragmentation by UV and mechanical abrasion breaks plastics into smaller pieces (Cózar et al., 2014; Ryan, 2020). Sedimentation rates due to biofouling are directly linked to the size of plastics, more specifically the surface area: volume ratios (Ryan, 2015b; Fazey and Ryan, 2016), and as a result, once the pieces are small enough, they are likely lost from the surface, possibly explaining the paucity of small items (< 1 mm) at the ocean surface (Cózar et al., 2014; Eriksen et al., 2014; Ryan, 2015b; Fazey and Ryan, 2016; Koelmans et al., 2017; Onink et al., 2022). The thickness of plastic items also plays a role in their surface longevity as it is directly linked to their surface area: volume ratio. Thicker items take longer to break down in the marine environment, remain buoyant for longer, and are capable of dispersing farther (Ryan, 2015b; Fazey and Ryan, 2016; Chamas et al., 2020). It is however thought that the average size of plastic item thickness may decrease over time, as manufacturers strive to reduce material costs during production (Worrel et al., 1995; Ho et al., 2005). Temporal variation in the size and mass of items might potentially provide information on plastic trends at sea (Morét-Ferguson et al., 2010; Cózar et al., 2014; Eriksen et al., 2014).

Understanding the rates of fragmentation and subsequent sedimentation of plastic from the sea surface remains poorly understood due to the dynamic interactions involving stranding/settling, resurfacing, and recirculation of buoyant plastics across shoreline, coastal, and offshore waters (Barnes et al., 2009; Gewert et al., 2015; Koelmans et al., 2017; Lebreton et al., 2019; Onink et al., 2022; Chapter 1; Fig. 1.3). However, macroplastics degrade into meso- and microplastics over time, with biofouling aiding in their removal from the ocean surface (Eriksen et al., 2014; Ryan, 2015b; Fazey and Ryan, 2016; Kaandorp et al., 2021). Koelmans et al. (2017) used a whole ocean emission-fragmentation-settling mass balance model to suggest that 1) under current plastic emission scenarios, plastic mass could triple by 2100, 2) stabilize under a constant emission scenario, or 3) rapidly diminish within three years if emissions ceased. Based on these crude estimates, we might infer that the average size of floating plastics items should increase in an increasing plastics scenario, remain stable in a steady-state scenario and decrease in a zero-plastics emission scenario. However, their model was perhaps overly simplistic, assuming rapid turnover at sea and overlooking the dynamic input and loss of plastics due to stranding, release, and recirculation (Lebreton et al., 2019). Onink et al. (2022) predicted that ocean-based fragmentation of marine plastics requires decadal time scales, but their simulations failed to include the influence of biofouling. Consequently, it is hard to infer how the flux of plastics through the marine environment will affect the size of plastics floating at the sea surface. Nonetheless, previous studies have observed the average size of floating plastics sampled in surface trawls to decrease over time (e.g., Morét-Ferguson et al., 2010), suggesting the potential to detect alterations in the size of plastics over longer temporal periods. However, size composition may vary regionally in different water conditions, due to differential biofouling and sedimentation rates which are influenced by factors such as salinity, water temperature, light availability, nutrient levels, water velocity and turbulence (Fazey and Ryan, 2016; Kaiser et al., 2017; Onink et al., 2022). This leads to a complex interplay between time and space on particle size.

Plastics are composed of different polymers, each with specific densities and chemical properties, which influence their buoyancy at sea, fragmentation rates (Song et al., 2017; Masry et al., 2021) and sorption rates of persistent organic pollutants (POPs) such as polychlorinated biphenyls (PCBs) and organochlorine pesticides (Colabuono et al., 2010; Rochman et al., 2013). Buoyant plastics like PE (0.88-0.96 $g \cdot cm^{-3}$) and PP (0.90-0.93 $g \cdot cm^{-3}$) have densities lower than seawater (~1.03 $g \cdot cm^{-3}$) and dominate floating microplastics at sea as well as the plastics ingested by seabirds (Avery-Gomm et al., 2016; Tanaka et al., 2019; Kühn et al., 2021; Robuck et al., 2022; Chapter 3). Some studies have reported faster fragmentation rates for PP than PE under simulated marine weathering conditions (Song et al., 2017; Chamas et al., 2020; Hoseini et al., 2023). Polymer type could therefore also influence the longevity of plastic fragments at the ocean surface (Gewert et al., 2015; Song et al., 2017; Zhang et al., 2017), and also impact seabird health through differential sorption of POPs (Colabuono et al., 2010; Rochman et al., 2013; Tanaka et al., 2013; Bustnes et al., 2015).

Most studies that use seabirds to assess temporal variation in marine litter have focused on the amounts ingested; few have reported changes in the characteristics, such as size, mass and polymer types of ingested particles (Provencher et al., 2017; Kühn et al., 2021; Collard et al., 2022; Shugart et al., 2023). In this chapter, I build on the results from Chapter 4 and assess if the size, mass and polymer types of plastics ingested by petrels have changed over the last three decades. My predictions are that 1) if there has been little change in plastics loads (as found in Chapter 4) there will be little change in the size and mass of ingested plastic pieces over time; 2) the thickness of rigid plastics may decrease over time as producers strive to reduce material costs; 3) changes in polymer ratios might reflect differential fragmentation and retention rates of polymers at sea. I conclude by contextualising my findings with those from Chapter 4, and explore their influence on plastic loads over time.

Methods

Data collection

Regurgitated Brown Skua pellets were collected in a large skua club at West Point on Inaccessible Island, Tristan da Cunha Archipelago, central South Atlantic Ocean from 1987 to 2018 (see Chapters 3 and 4 for more detailed methods). I limited the data in this chapter to adult birds as far as possible, and to items collected between September to December to avoid any seasonal effects linked to breeding seasons (see Chapters 3 and 4). The number of items processed per taxon per year in this chapter differs slightly from Chapter 4. I used the plastics sampled from the 3 493 skua pellets (only adults) in Chapter 4, as well as additional stored plastic items that were distributed over multiple pellets (but for one taxon), in these analyses.

Characteristics of ingested plastics

Each plastic item was assigned into one of five categories: hard fragments (including rigid items), industrial pellets, flexible plastic (i.e. sheet plastics used for food packaging and bags), thread-like plastics (e.g. rope and netting) and foamed plastics such as expanded polystyrene (EPS) as in Chapters 2–4. Following the methods in Chapter 3, I aimed to analyse the polymer types of 100 hard fragments and 100 pellets per taxon per time period. I processed hard fragments and industrial pellets until 100 of each per taxon yielded a > 70% spectral match in each time period. This was done by randomly selecting individual sample bags (samples collected in 2018), containing the plastics ingested by one individual, and processing all items in each bag. In years where plastic items weren't stored separately for each individual, all items were spread onto a small tray, and a ruler was used to spilt the sample in half (large samples were halved again). The right half was returned to the sample bag and the remaining items were processed in a random order, until the desired sample sizes were achieved. For most taxa, all industrial pellets were processed, but in certain instances, too few industrial pellets were collected during September–December, or in adult birds, and had to be augmented with samples collected during January–March, or in the case of Broad-billed Prions in 2018, augmented with samples collected from skua pellets containing the remains of chicks. For Great Shearwaters, samples from 2009–2014 were augmented with 61 industrial pellets collected from stomach samples of chicks harvested for food by Tristan islanders on Nightingale Island (37.42ºS, 12.48ºW), Tristan da Cunha, during March 2006. FTIR spectroscopy followed a similar protocol to that described in Chapters 2 and 3. Due to logistical constraints, 64% of samples were processed with a Bruker Alpha II compact FTIR spectrometer and 36% on an iD7 ATR Nicolet iS5 (ThermoScientific) instrument, using the protocols reported in Chapters 2 and 3. Polymer type was only assigned if there was a ≥ 70% match (98% of all samples examined). The maximum length and thickness of each plastic item was measured with digital callipers (to the nearest 0.01 mm). Since length is a reliable proxy for size, width measurements were not included (Provencher et al., 2017; Terepocki et al., 2017; Jâms et al., 2020). Items were weighed on a precision balance to the nearest 1 mg (0.1 mg for items < 2 mg).

Data analyses

Following the methods in Chapter 4, data were grouped into four roughly decadal sampling periods for analyses: 1987–89; 1999–2004; 2009–2014 and 2018. Data were checked for normality using Shapiro-Wilk tests. The results mostly focus on hard fragments as they were the most numerous items recorded, and most likely to provide the most informative results. In some instances, I also report industrial pellet data, but results for flexible, thread-like and foamed plastics, as well as all items combined, are generally only presented in the supplementary material (Tables S5.2, S5.3 and S5.4). Chi-squared tests of independence were used to compare ratios of polymer types (polymer types where < 5 items were recorded were grouped together as "other") for each taxon between the four time periods. If there were too few items even when grouped (n < 5), Fisher's exact tests were used instead. Global polymer production statistics for 1990–2019 were obtained for PP and PE (HDPE and LDPE combined) from the OECD (2022).

I used violin and boxplots of length and mass (log transformed to improve visualization), and boxplots of thickness to assess temporal variation in the size of hard fragments ingested in each taxon and time period. Kruskal-Wallis tests were used to test for differences in length (mm), mass (mg) and thickness (mm) of hard fragments recorded in each taxon and among time periods. A post-hoc application of Dunn's test with Bonferroni correction (to decrease the chance of committing Type 1 errors) was used to show where significant differences were. I used Pearson's correlation coefficients (r) to assess the relationships between the length and cube root of mass of hard fragments in each taxon and time period. To assess if grouping the data into time periods masked fine-scale temporal variations, I also analysed changes in the length, mass and thickness of hard fragments in each taxon and sampling year using general linear models. Length, mass and thickness were response variables and date (sampling year) was the predictor variable. Data were checked for normality and homoscedasticity using residuals. Boxplots of the length, mass and thickness of hard fragments recorded per year were also used to compare temporal variation between sampling years. Mann-Whitney U tests were used to compare the length, thickness and mass of hard fragments composed of PE and PP, first per taxon, then combined, across all time periods. All statistical analyses were performed in R 4.2.3 (R Core Team, 2023).

Results

I measured and weighed 11 159 plastic items collected from regurgitated skua pellets containing the remains of *Fregetta* storm petrels (n = 947 plastic items), White-faced Storm Petrels (n = 3 249), Broad-billed Prions (n = 2 992) and Great Shearwaters (n = 3 971) collected between September and December from 1987 to 2018 (Table 5.1). Hard fragments were the most numerous items overall (78%), followed by industrial pellets (20%); flexible pieces, thread-like items and foamed plastics collectively constituted only 2% of plastic items (Table 5.1). In accordance with the results from Chapters 3 and 4, *Fregetta* storm petrels ingested more thread-like plastics, while White-faced Storm Petrels ingested more industrial pellets compared to the other taxa (Table 5.1). Flexible pieces and foamed plastics were mostly found in Broad-billed Prions and Great Shearwaters (Table 5.1).

The length and cube root of mass of hard fragments were strongly correlated in all taxa and time periods (Fig. 5.1; Table S5.1). The lengths of hard fragments showed significant differences among time periods in all taxa except White-faced Storm Petrels, but differences were minor and no marked trends were apparent (Fig. 5.2A; Table S5.2). For instance, hard fragments in *Fregetta* storm petrels were shorter in 1987–89 (median 3.2 mm) and 2018 (3.4 mm) than in 1999–2004 (3.8 mm) and 2009–2014 (3.7 mm; Fig. 5.2A; Table S5.2). In Broad-billed Prions, fragments were longest in 1999–2004 (6.3 mm), followed by 1987–89 (6 mm), and were shorter in 2009–2014 (5.7 mm) and 2018 (5.5 mm; Fig. 5.2A; Table S5.2). In Great Shearwaters, fragments were longest in 1987–89 (6.8 mm), followed by 2009–2014 (6.2 mm), and shortest in 1999–2004 and 2018 (both 5.6 mm; Fig. 5.2A; Table S5.2). Analyses based on data collected per sampling year, rather than decadal periods, yielded comparable results (Figs S5.1A and S5.2A). Despite the significant influence of sampling year on plastic fragment length in *Fregetta* storm petrels ($P < 0.05$), Broad-billed Prions ($P < 0.001$), and Great Shearwaters ($P < 0.001$; Fig. S5.1A), the models had very little explanatory power for the variation observed among years ($R^2 = 0.01$). Boxplots further illustrated variability in lengths among years, but consistent trends were lacking (Fig. S5.2A). The length of industrial pellets differed slightly among time periods in White-faced Storm Petrels (longer in 2009–2014 (3.7 mm) than 1987–89, 1999–2004 and 2018 (all 3.5 mm)) and in Broad-billed Prions (longer in 1999–2004 (4 mm) than 1987–89 (3.9 mm) and 2009–2014 and 2018 (both 3.8 mm); Table S5.2).

Table 5.1 Number of plastic items collected from regurgitated skua pellets that were measured and weighed for each taxon within each time period.

	***Fregetta* storm petrels**	**White-faced Storm Petrel**	**Broad-billed Prion**	**Great Shearwater**	**Total**
1987–1989					
Hard fragments	71%	40%	63%	66%	**1 017**
Industrial pellets	25%	60%	36%	33%	879
Thread-like plastics	3%	–	1%	1%	12
Flexible plastics	1%	–	< 1%	–	3
Foam	–	–	< 1%	–	1
Total	**267**	**936**	**510**	**199**	**1 912**
1999–2004					
Hard fragments	67%	68%	74%	87%	1 904
Industrial pellets	26%	31%	25%	13%	582
Thread-like plastics	6%	1%	1%	–	10
Flexible plastics	1%	< 1%	< 1%	–	19
Foam	–	–	–	–	–
Total	**214**	**508**	**1 154**	**639**	**2 515**
2009–2014					
Hard fragments	79%	79%	85%	88%	2 294
Industrial pellet	14%	21%	14%	10%	442
Thread-like plastics	7%	–	< 1%	1%	28
Flexible plastics	–	–	< 1%	1%	5
Foam–	–	–	–	–	
Total	**256**	**1 038**	**922**	**553**	**2 769**
2018					
Hard fragments	80%	79%	87%	92%	3 496
Industrial pellet	11%	20%	11%	6%	369
Thread-like plastics	9%	1%	< 1%	1%	44
Flexible plastics	< 1%	< 1%	1%	2%	51
Foam	–	–	–	< 1%	3
Total	**210**	**767**	**406**	**2 580**	**3 963**
Overall	**947**	**3 249**	**2 992**	**3 971**	**11 159**

The masses of hard fragments were only significantly different in *Fregetta* storm petrels between 1987–89 (median 4 mg) and 1999–2004 and 2018 (both 6 mg) but not compared to 2009–2014 (5 mg; Fig. 5.2B; Table S5.3). The linear model reflected similar results, with sampling year having a significant influence on item mass ($P < 0.001$), but no trends were obvious ($R^2 = 0.02$; Fig. S5.1B). The boxplots highlighted the variability in masses among years (Fig. S5.2B). Industrial pellet mass differed in *Fregetta* storm petrels between 1987–89 (10 mg) and 2018 (19 mg), in White-faced Storm Petrels between 2009–2014 (16 mg) and both 1987–89 (12 mg) and 1999–2004 (13 mg), and in Great Shearwaters between 1987–89 (23 mg) and 1999–2004 (20 mg), but no consistent trends were observed among taxa (Table S5.3).

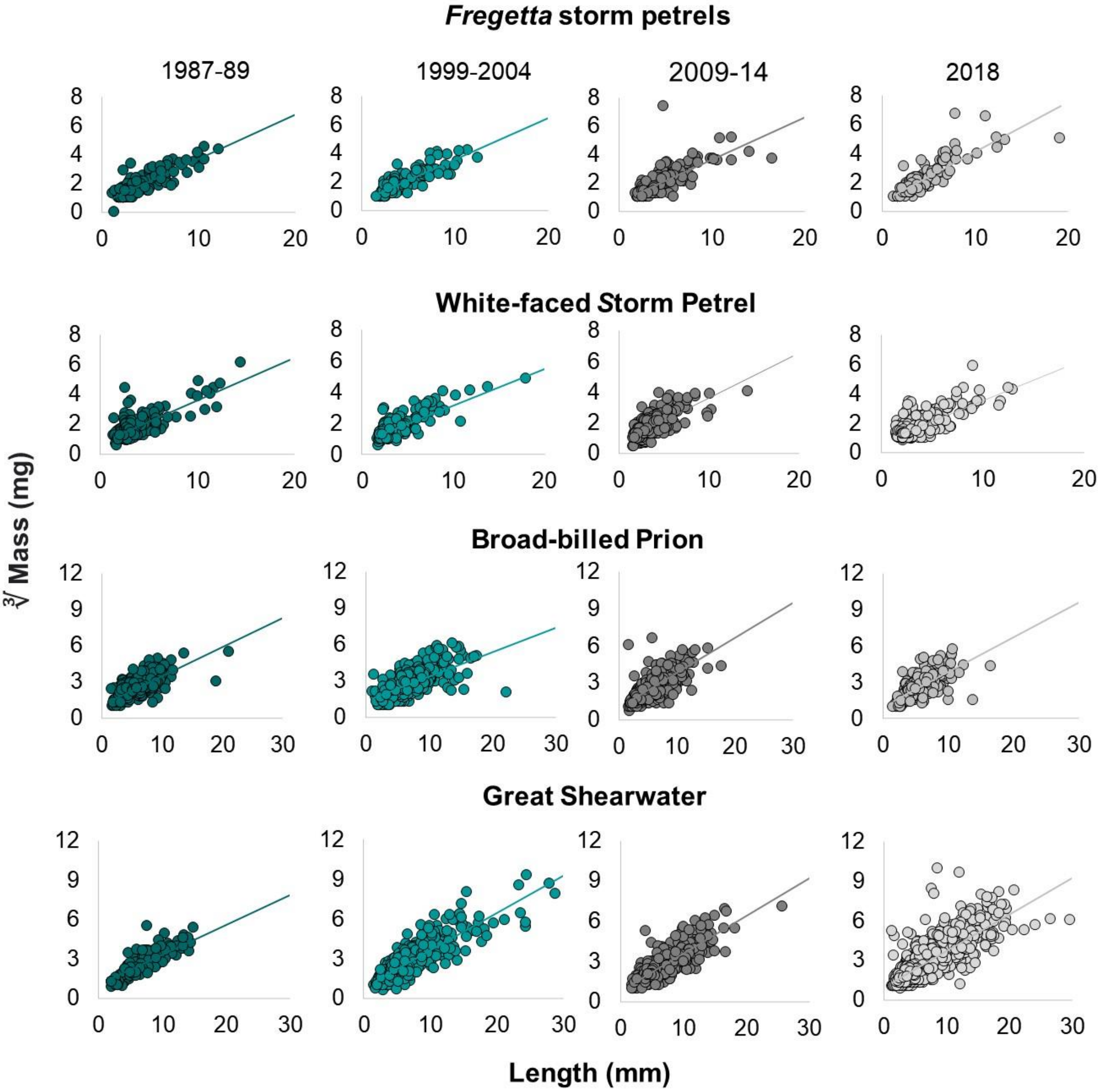


Fig. 5.1 Scatterplots of the length (mm) and mass (cube root, mg) of hard fragments for each taxon over time. Note differences in scaling in axes among taxa

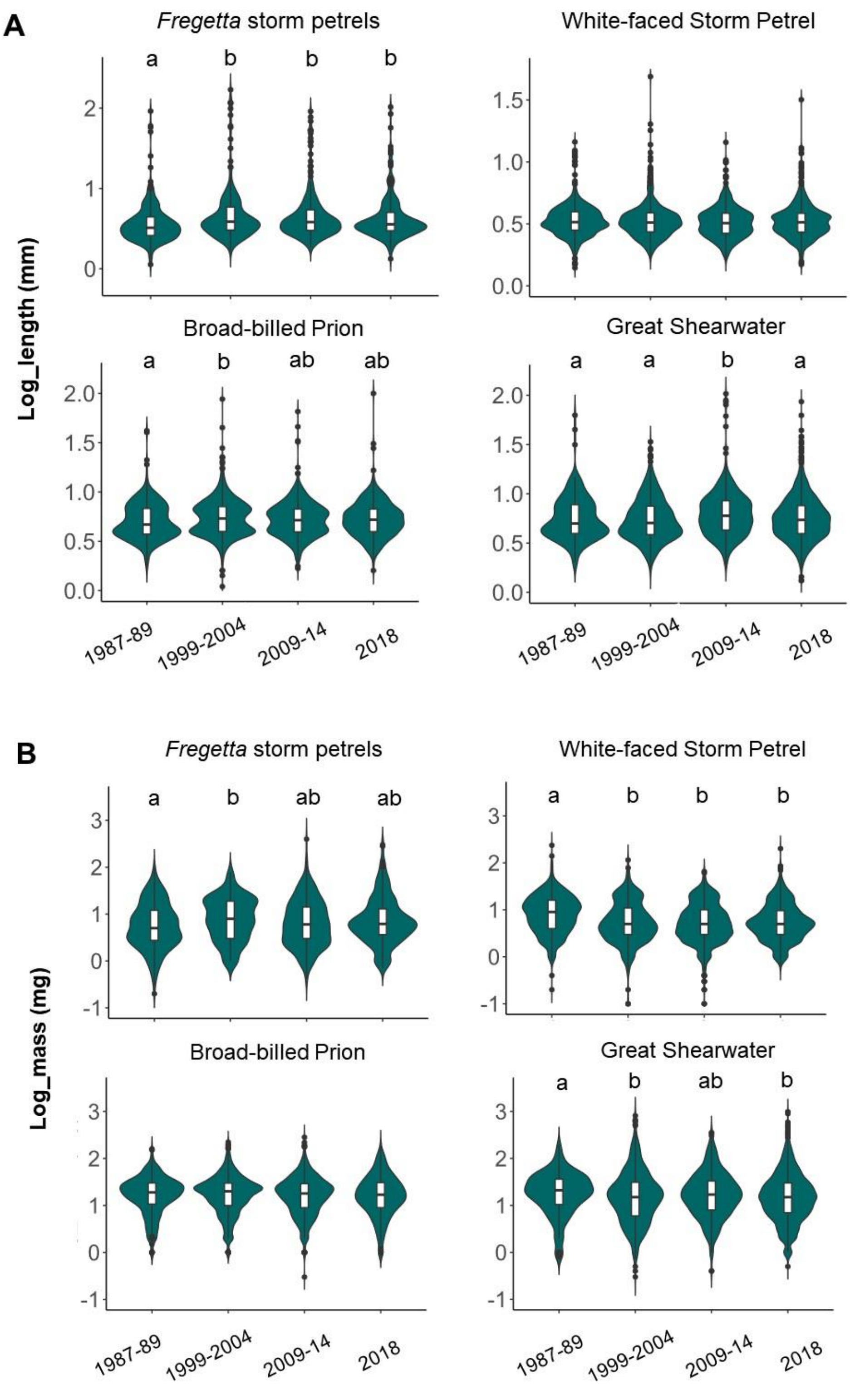


Fig. 5.2 Plots comparing the log transformed lengths (A) and masses (B) of hard fragments recorded in the four taxa for each time period. Boxplots indicate the median (dark line) and interquartile range while violin plots indicate the kernel probability density at different values where the width of each curve corresponds to the frequency of data points in that region. Letters above boxplots signify which time periods differed significantly within each taxon (post hoc Dunn's test results; $P < 0.05$). Note differences in scaling on y-axes.

Significant variations in hard fragment thickness among time periods were observed, with both *Fregetta* storm petrels and Broad-Billed Prions exhibiting a slight consistent increase in the thickness of hard fragments (Fig. 5.3; Table S5.4). *Fregetta* storm petrels ingested thicker items in 2018 (1.3 mm) and 2009–2014 (1.2 mm) than 1999–2004 and 1987–89 (both 1.0 mm) (Fig. 5.3A; Table S5.4), and Broad-billed Prions ingested thicker items in 2018 (1.4 mm) and 2009–2014 (1.3 mm) than 1999–2004 and 1987–89 (both 1.0 mm; Fig. 5.3C; Table S5.4). White-faced Storm Petrels ingested thicker items in 2009–2014 (1.2 mm) than 2018 and 1987–89 (both 1.1 mm) and 1999–2004 (1.0 mm) (Fig. 5.3B). Analyzing data collected for each sampling year revealed a significant correlation between item thickness and sampling year in *Fregetta* storm petrels ($P < 0.001$), Broad-billed Prions ($P < 0.001$), and Great Shearwaters ($P < 0.05$). However, the model could not pinpoint when these differences occurred (R^2 = 0.001-0.04; Figs. S5.1C and S5.2C). Industrial pellet thickness varied only in White-faced Storm Petrels, being greater in 2009–2014 (2.4 mm) than 1987–89 and 1999–2004 (both 2.1 mm; Table S5.4).

There was a significant difference in the ratios of polymer types recorded in hard fragments among years in all taxa. In *Fregetta* storm petrels the ratio of PP:PE increased nearly three-fold from 1987–89 to 2018 (Fig. 5.4A; Fisher's exact test, two-sided, $P < 0.001$). In White-faced Storm Petrels the ratio of PP:PE nearly doubled ($\chi^2 = 12.57$, df = 3, $P < 0.05$; PP and PP-EPDM pooled), in Broad-billed Prions the proportion of PP increased nearly fivefold ($\chi^2 = 14.14$, df = 3, $P < 0.05$), and in Great Shearwaters nearly four-fold from 1987–89 to 2018 ($\chi^2 = 23.88$, df = 3, $P < 0.001$; Fig. 5.4A). Industrial pellet polymer ratios did not differ significantly among time periods in *Fregetta* storm petrels ($\chi^2 = 4.22$, df = 3, $P = 0.24$), White-faced Storm Petrels ($\chi^2 = 1.42$, df = 3, $P = 0.70$), Broad-billed Prions ($\chi^2 = 4.36$, df = 3, $P = 0.23$) or Great Shearwaters ($\chi^2 = 5.56$, df = 3, $P = 0.14$; Fig. 5.4B). Too few thread-like, flexible and foamed plastics were sampled for temporal comparisons (Tables S5.5 and S5.6).

The length of hard fragments composed of PP and PE were not significantly different in any of the taxa, or when combined (Table S5.7). The mass of PE items were significantly greater in White-faced Storm Petrels and Broad-billed Prions, but not when combined for all species (Table S5.7). Hard fragments composed of PE were significantly thicker than PP fragments in *Fregetta* storm petrels, White-faced Storm Petrels, Broad-billed Prions and overall (Table S5.7).

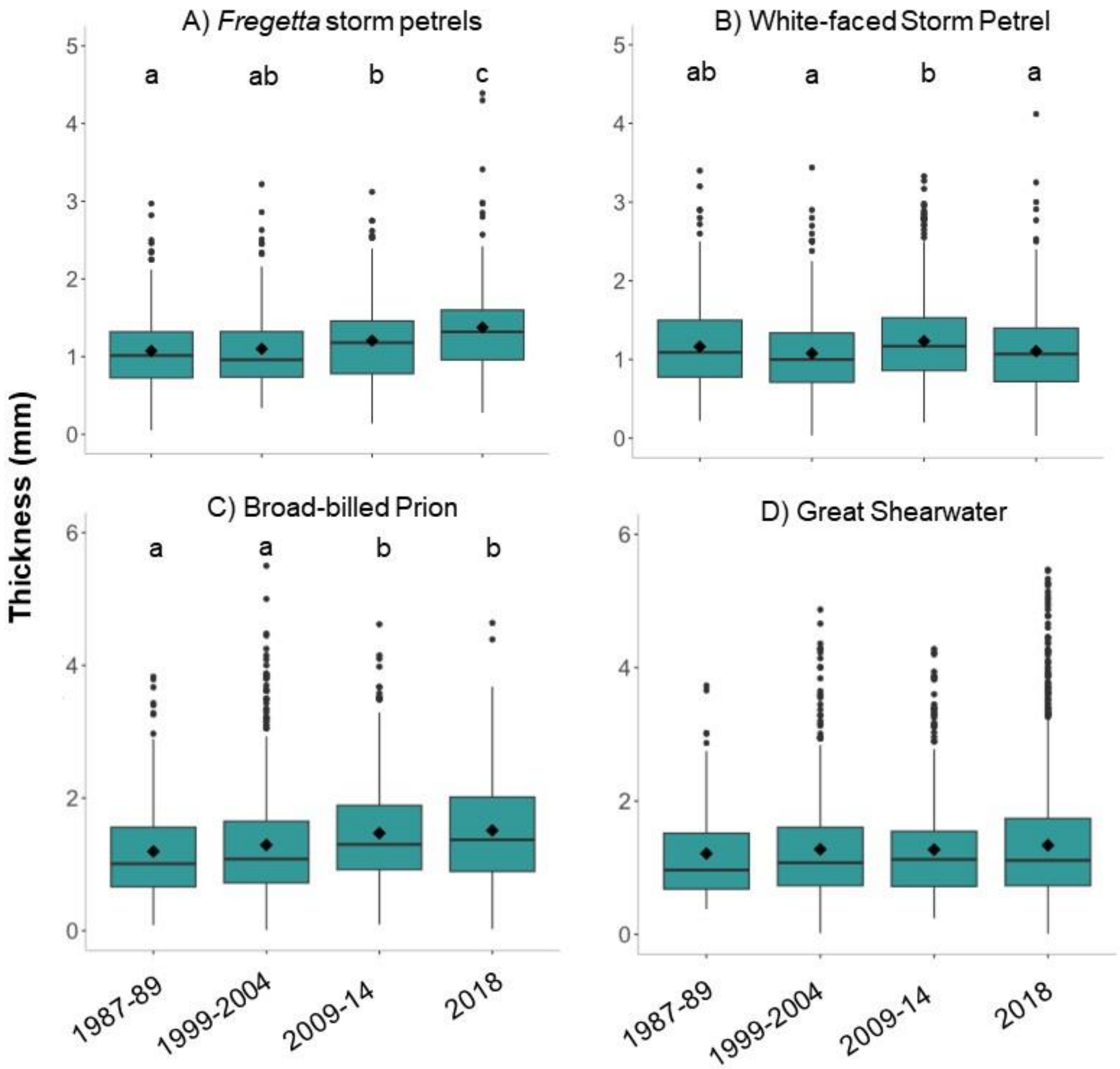


Fig. 5.3 Boxplots indicating the median (black line), mean (diamond), interquartile range (teal box), values within 1.5 times interquartile range (whiskers) and outliers (values outside 1.5 times the inter-quartile range) of the thickness (mm) of hard fragments ingested by the four taxa across the four time periods. Letters above boxplots signify which time periods were significantly different (not sharing any letters) in each taxon (post hoc Dunn's test results; P < 0.05). Note differences in scaling on y-axes among taxa.

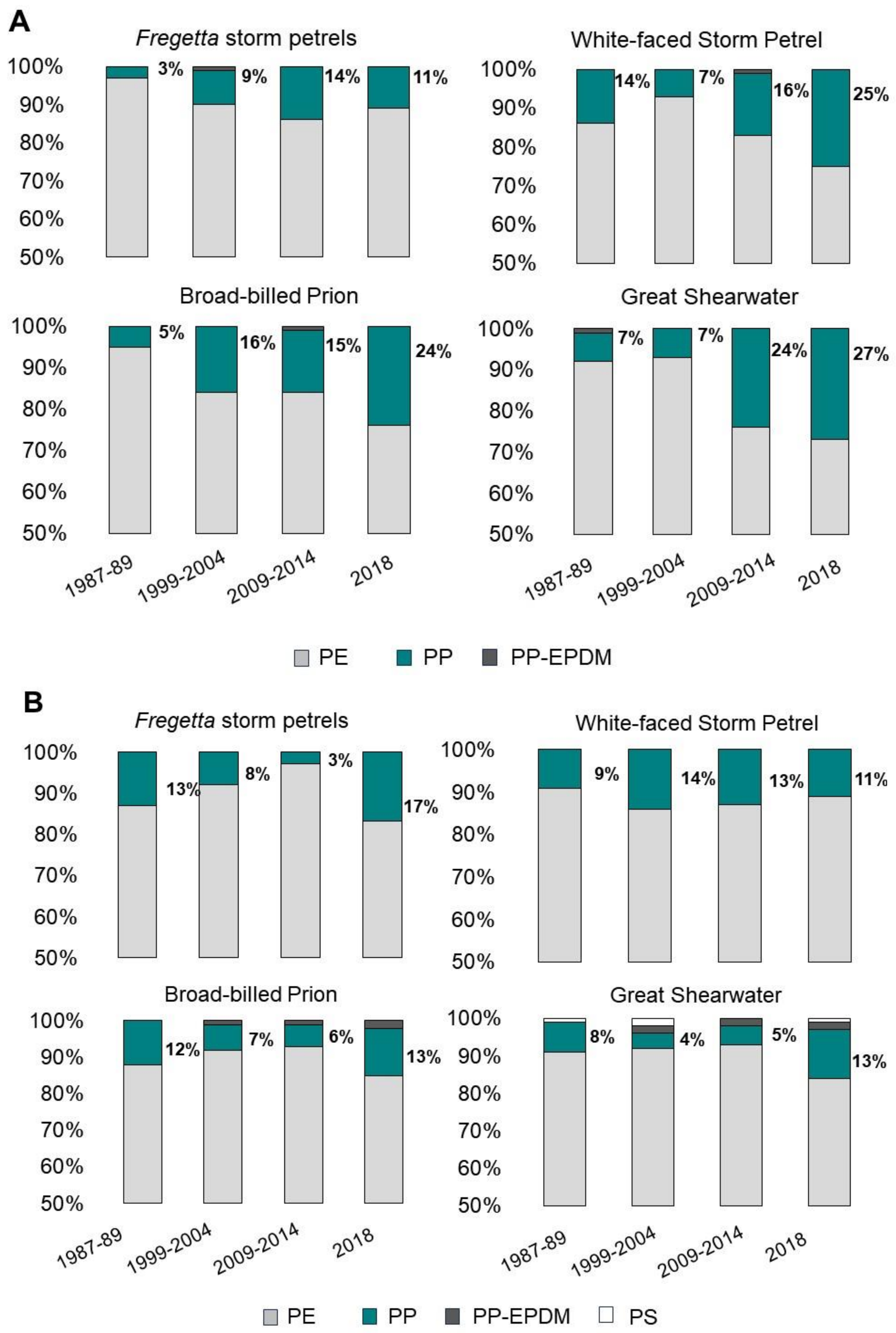


Fig. 5.4 Temporal variation in the proportions of hard fragments (A) and industrial pellets (B) composed of polyethylene (PE), polypropylene (PP), PP-ethylene propylene diene monomer (PP-EPDM) and polystyrene (PS) recorded in four seabird taxa between 1987–2018.

Discussion

This chapter investigated temporal variations in the size and polymer types of plastics ingested by petrels in the South Atlantic over 30 years. Patterns among species for selectivity of plastics types, like *Fregetta* storm petrels ingesting more thread-like items and White-faced Storm Petrels more industrial pellets (Chapter 3), have remained largely constant since the 1980s. Most items were made of PE, followed by PP which is consistent with other studies, suggesting that these are the most common polymer types of small buoyant plastics available to seabirds at sea (Avery-Gomm et al., 2016; Kühn et al., 2021; Robuck et al., 2022; Navarro et al., 2023; Collard et al., 2024; Chapter 3). Although few changes were noted in plastic size, intriguing trends emerged, including greater thickness in hard fragments and an increased ratio of PP:PE. The subtle alterations in plastic dimensions and composition matches the limited change in ingested plastic loads within the same seabirds over the same time period (Chapter 4), highlighting that the characteristics of ingested plastics are useful variables to record in plastic ingestion studies.

Temporal variation in the size and mass of plastics

I found little change and no clear trends in the length and mass of plastics ingested by petrels from 1987 to 2018. Morét-Ferguson et al. (2010) reported that the mean size of plastics collected with surface nets in the western North Atlantic Ocean halved from the 1990s (10.7 ± 31.7 mm) to the 2000s (5.1 ± 6.6 mm), and attributed it to continuous weathering during prolonged periods at sea. Shugart et al. (2023) reported a slight increase in the mean particle size (calculated as mean mg/mean number of items, per time period) ingested by Short-tailed Shearwaters from 1969–1977 to 2019, but attributed it to the transition to fewer industrial pellets and more user fragments over time. If plastic input loads were increasing (in a steady-state system where sedimentation rates remain constant), we may expect there to be more larger items available on the sea surface over time. Seabird species differ in the size of plastics they ingest (Ryan, 1987a; Roman et al., 2019b; Chapters 2 and 3) and if plastic input rates, and the resultant size of plastics available on the sea surface, are increasing over time, we might expect plastic loads to increase in the larger Great Shearwaters and prions (which mainly ingest 5–6 mm plastic items), while decreasing in the smaller storm petrels (which mainly ingest 3–4 mm items). In Chapter 4, I found no evidence of increasing plastic loads in the storm petrels or in Broad-billed Prions, but loads in Great Shearwaters increased over the last decade, possibly suggesting an increase in plastic abundance in the North Atlantic (Chapter 4). In Chapter 2, I reported that hard fragments regurgitated by Wandering Albatrosses on Marion Island showed a marked increase in length, thickness and mass between 1999–2008 and 2009–2018. This may indicate increased litter inputs in the southern Indian Ocean, but this species showed barely any increase in the amount of plastic collected at nests each year among time periods (Table 2.3). In addition, given the steady increase in nest numbers over the last decade, it probably points to an overall decrease in plastic loads in Wandering Albatrosses (Ryan and Connan, unpubl. data). Interpreting temporal variation in plastic densities at sea is challenging because of the dynamic flux of plastics through marine environments, and the large heterogeneity in plastic accumulation and densities at sea (Ryan et al., 2009a; Koelmans et al., 2017; Lebreton et al., 2019; Kaandorp et al., 2021). However, the limited change in the size and mass of ingested plastics in petrels over three decades recorded in this study mirrors the overall limited change in plastic loads in these birds (Chapter 4) and might reflect limited changes in the amounts of small buoyant plastics in the South Atlantic Ocean.

The shape and type of plastic is thought to influence how long plastic items remain at the sea surface. Ryan (2015b) suggested that industrial pellets will take longer to decrease in size over time than hard fragments, because they are compact, cylindrical and probably stronger than thinner hard fragments, especially once weakened by UV degradation. This implies that they will remain buoyant for longer than fragments. However, the number of industrial pellets ingested by seabirds at Inaccessible Island have decreased over time (Ryan, 2008; Chapter 4) presumably reflecting the success of initiatives to reduce the numbers of pellets leaking into marine environments (e.g. Operation Clean Sweep, www.opcleansweep.org) as highlighted in Chapter 4. In spite of these initiatives, large accidental spills do still occur. In October 2017, a storm on the South African east coast resulted in the accidental release of approximately 49 tonnes of PE pellets from two containers lost from a ship into Durban harbour (Schumann et al., 2019). These inputs may have contributed to an apparent increase in the size of industrial pellets documented in 2018, as newly introduced pellets are likely to be larger than those that have been adrift at sea for an extended period. However, this trend was only evident in *Fregetta* storm petrels, which typically ingest fewer pellets than other taxa. However, industrial pellets from the 2017 spill might not have dispersed widely enough in the South Atlantic to be encountered by seabirds breeding on Inaccessible Island in 2018 (based on the surface drifter model from PlasticAdrift - www.plasticadrift.com). For example, Campbell (2024) recorded that White-chinned Petrels caught on fishing lines off the coast of South Africa in 2017–2023 contained more industrial pellets than those sampled prior to 2017 spill. The petrels sampled after 2020 also were impacted by a less well-documented but likely even larger pellet spill off the Southern Cape in 2020 (www.dailymaverick.co.za/article/2020-10-27-ngos-call-on-cape-town-beachgoers-to-help-clean-up-debris-from-massive-nurdle-spill/).

My prediction that average item thickness will decrease over time, to reduce material costs (Worrel et al., 1995; Ho et al., 2005), was not supported. I found little change in item thickness in White-faced Storm Petrels and Great Shearwaters, and only a subtle increase in *Fregetta* storm petrels and Broad-billed Prions. The thickness of plastic items plays a crucial role in their buoyancy, with thicker items being more likely to remain at the sea surface and disperse farther at sea (Ryan, 2015b; Fazey and Ryan, 2016). Differences in foraging areas may be a contributing factor to the slight increase in item thickness observed in these two taxa, as they spend more time foraging south of the Sub-tropical Front in remote oceanic areas, whereas White-faced Storm Petrels predominantly forage within the Sub-tropical Gyre, and Great Shearwaters spend more time foraging over continental shelves, particularly off the coast of South America (Chapter 3; Fig. S3.4). It is however unclear exactly how foraging area will influence the increase in item thickness over time. These results further highlight the complex dynamics governing the longevity and dispersal of plastics on the sea surface. Further research, such as a dynamic model, might be useful to aid in the interpretation of these findings.

Temporal variation in polymer types of plastics

Determining polymer types of ingested plastics have historically been uncommon practice. However, with the increased availability of FTIR spectrometers and related instruments, as well as improved best-practice recommendations (e.g. Provencher et al., 2019), more studies are now including polymer types in their analyses. Consequently, few studies assessing polymer types over long timeframes are available, limiting comparative options. Kühn et al. (2021) reported increased ratios of PP:PE in Northern Fulmars in the Netherlands over three decadal periods since the 1980s, but attributed it to decreased proportions of industrial pellets, that are mostly composed of PE. I analysed temporal variation in polymer types for hard fragments and industrial pellets separately. The ratio of PP to PE in hard fragments ingested by seabirds in my study increased from 1987 to 2018 in all four taxa. Four scenarios may explain this trend. Firstly, it might be driven by an increase in the production of PP relative to PE, but the production ratio by mass of PP to PE has remained remarkably constant over the last 30 years (Geyer et al., 2017; OECD, 2022), and so is unlikely to account for the change in seabirds. Secondly, differential recycling of PE and PP might alter their proportions in the waste stream, and ultimately at sea. For example, only 0.6% of PP, compared to 15.6% of PE (HDPE and LDPE combined) is recycled in the USA, because it is often used in multilayer food packaging containing non-plastic components such as foil, paper labels, and adhesives that complicated recycling (Gandhi et al., 2021). As a result, PP food packaging containers often end up in the waste stream (Bora et al., 2020; Gandhi et al., 2021; Alsabri et al., 2022). However, given the thickness of PP hard fragments ingested by seabirds in this study (mean = 1.1 ± 0.7 mm; range = 0.2-5.1 mm), it is unlikely that fragments originate from food packaging, which are generally thinner (~0.2 mm), and therefore differential recycling rates are an unlikely explanation. Thirdly, differential fragmentation rates of PP and PE might drive the observed change. Experimental degradation studies suggest that PP fragments substantially faster than PE (Song et al., 2017; Chamas et al., 2020; Hoseini et al., 2023), although this is not always the case (Lambert and Wagner, 2016). However, these results were produced during laboratory simulations, and degradation rates under natural conditions are needed to estimate *in situ* fragmentation rates. In addition, most commercially used plastics contain additives and colourants, which further complicate estimates of degradation rates (Gewert et al., 2015; Hoseini et al., 2023; Key et al., 2024). However, if PP fragments break down faster than PE, it could result in a faster increase in the number of small PP items available at the sea surface over time (Kaandorp et al., 2021). Lastly, differential retention rates of plastics on the sea surface could potentially contribute to the observed trend. Factors such as polymer density and shape, influence surface retention times. My results suggest that PP may have longer retention times than PE, but densities of the two polymers are similar, and PP fragments averaged thinner than PE fragments, so this does not appear to explain the observed trend. Additional data, ideally from an independent source, are needed to confirm the increasing importance of PP among meso- and larger microplastics at sea.

Conclusions

In this chapter, I investigated how the characteristics of plastics ingested by four petrel taxa that predominantly forage in the South Atlantic Ocean have changed over a 30-year period. I found little change in the length and mass of plastic items over time, but thickness increased in two taxa foraging at higher southern latitudes. The increase in PP relative to PE in hard fragments was consistent across taxa, and future studies monitoring marine plastics should include polymer identification as part of the protocol. This information may be useful to elucidate patterns and help inform plastic packaging design in the future, potentially reducing the amounts available to seabirds at sea over time, and the risks associated with ingestion. A dynamic model that considers the amount of plastic by polymer type entering and leaving the marine environment may be able to shed some light on the observed trends, but this is complicated by uncertainties regarding polymer-specific fragmentation and sedimentation rates. In conclusion, I found that regurgitated skua pellets offer a valuable approach for monitoring subtle changes in the characteristics of marine plastics. I recommend continuing monitoring initiatives to identify trends that could be used to develop effective strategies to mitigate plastic pollution and protect marine ecosystems.

Supplementary materials: Chapter 5

Table S5.1 Pearson correlation coefficients highlighting the relationship between the length and cube root of mass of hard plastic fragments ingested by four seabird taxa from 1987 to 2018.

	1987–1989	**1999–2004**	**2009–2014**	**2018**
Fregetta storm petrels	$r = 0.85, P < 0.001$	$r = 0.86, P < 0.001$	$r = 0.76, P < 0.001$	$r = 0.84, P < 0.001$
White-faced Storm Petrel	$r = 0.76, P < 0.001$	$r = 0.79, P < 0.001$	$r = 0.71, P < 0.001$	$r = 0.72, P < 0.001$
Broad-billed Prion	$r = 0.71, P < 0.001$	$r = 0.65, P < 0.001$	$r = 0.73, P < 0.001$	$r = 0.70, P < 0.001$
Great Shearwater	$r = 0.81, P < 0.001$	$r = 0.87, P < 0.001$	$r = 0.82, P < 0.001$	$r = 0.78, P < 0.0$

Table S5.2 Median (range) in length (mm) of plastic items (overall and per category) ingested by seabirds breeding on Inaccessible Island collected during roughly decadal time periods. Kruskal–Wallis test results are indicated in parentheses next to each taxon name. Significance indicates the Dunn's test results for which the lengths of all items (1), hard fragments (2) and industrial pellets (3) among time periods not sharing any letters, were significantly different.

Species	**All items median (range)**	**Hard fragments**	**Industrial pellets**	**Flexible pieces**	**Thread-like plastics**	**Foamed plastics**	**Significance**		
							(1)	**(2)**	**(3)**
***Fregetta* storm petrels (**KW all items: χ^2 = 25.64, df = 3, P < 0.001); **(**fragments: χ^2 = 16.68, df = 3, P < 0.001); **(**pellets: χ^2 = 8.01, df = 3, P = 0.05)									
1987–89	3.2 (1.1-92)	3.2 (1.1-2.1)	3.3 (1.4-4.9)	13.2 (8.2-18.2)	50.9 (5.6-92)	–	a	a	-
1999–2004	3.9 (1.7-170)	3.8 (1.7-12.4)	3.7 (2.1-5.1)	16.9 (15.3-18.4)	70.9 (18.5-170)	–	b	b	-
2009–2014	3.8 (1.8-91.3)	3.7 (1.8-16.5)	3.8 (2.3-5)	–	38.3 (9-91.3)	–	b	b	-
2018	3.6 (1.3-103.5)	3.4 (1.3-19.2)	3.7 (2.8-5.1)	6.5	20.7 (2.8-103.5)	–	b	ab	-
White-faced Storm Petrel (KW all items: χ^2 = 7.41, df = 3, P = 0.06); **(**fragments: χ^2 = 5.32, df = 3, P = 0.15); (pellets: χ^2 = 8.9, df = 3, P = < 0.05)									
1987–89	3.3 (1.4-14.5)	3.1 (1.4-14.5)	3.5 (1.5-6)	–	–	–	-	-	a
1999–2004	3.3 (1.7-48.8)	3.1 (1.7-20.2)	3.5 (2.1-6.5)	4.15 (2.9-5.4)	8.2 (4.4-48.8)	–	-		ab
2009–2014	3.2 (1.6-14.4)	3.1 (1.7-20.2)	3.7 (1.9-5.2)	–	–	–	-	-	b
2018	3.3 (1.5-31.8)	3.2 (1.5-13)	3.5 (2.2-5.5)	6.6 (5.7-7.5)	7.3 (4.9-31.8)	–	-	-	ab
Broad-billed Prion (KW all items: χ^2 = 15.97, df = 3, P < 0.001); (fragments: χ^2 = 30.95, df = 3, P < 0.001); (pellets: χ^2 = 11.19, df = 3, P < 0.05)									
1987–89	4.7 (1.7-41.8)	6 (1.7-21.1)	3.9 (2.4-6.7)	16.4	40.3 (11.5-41.8)	13.4	a	ab	ab
1999–2004	5.4 (1.1-87.6)	6.3 (1.1-45)	4 (2.2-9.6)	8.3 (4.5-27.9)	20.0 (16.9-87.6)	–	b	a	a
2009–2014	5.2 (1.7-65.4)	5.7 (1.7-17.7)	3.8 (2.2-5.1)	56 (46-65)	32.6 (32-33)	–	ab	b	b
2018	5.2 (1.6-99.9)	5.5 (1.6-16.6)	3.8 (2.4-5.3)	11 (6.5-30.9)	56.5 (13.1-99.9)	–	ab	b	ab
Great Shearwater (KW all items: χ^2 = 22.96, df = 3, P < 0.001); (fragments: χ^2 = 40.07, df = 3, P < 0.001); (pellets: χ^2 = 1.31, df = 3, P = 0.73)									
1989	5 (2-62.8)	6.8 (2-31.5)	3.9 (2.7-5.5)	–	53.8 (44.8-62.8)	–	a	a	-
1999–2004	5.1 (1.6-33.7)	5.6 (1.6-33.7)	3.9 (2.4-5.1)	–	–	–	a	b	-
2009–2014	6 (1.9-103.6)	6.2 (1.9-25.8)	3.9 (2.7-5.4)	16.4 (14-103.6)	54.9 (12-89)	–	b	a	-
2018	5.4 (1.3-86.2)	5.6 (1.3-32.6)	3.9 (1.9-6.3)	12.2 (2.9-44.1)	21.1 (6.3-86.2)	6.9 (5-7.3)	a	b	-

Table S5.3 Median (range) mass (mg) of plastic items (overall and per category) ingested by seabirds breeding on Inaccessible Island collected during roughly decadal time periods. Kruskal–Wallis test results (for all items) are indicated in parentheses next to each taxon name. Significance indicates the Dunn's test results for which the mass of all items (1), hard fragments (2) and industrial pellets (3) among time periods not sharing any letters, were significantly different.

Species	**All items median (range)**	**Hard fragments**	**Industrial pellets**	**Flexible pieces**	**Thread-like plastics**	**Foamed plastics**	**Significance (1)**	**(2)**	**(3)**
***Fregetta* storm petrels (**KW all items: $\chi^2 = 11.7$, df = 3, P < 0.05); **(**fragments: $\chi^2 = 16.4$, P < 0.001); (pellets: $\chi^2 = 11.1$, df = 3, P < 0.05)									
1987–89	5 (0.2-92)	4 (0.3-92)	10 (1-45)	7 (3-10)	3 (0.2-9)	–	a	a	a
1999–2004	8 (1-78)	6 (1-78)	18 (1-28)	5 (1-9)	6 (1-10)	–	b	b	ab
2009–2014	6 (0.4-398)	5 (0.6-398)	14 (5-38)	–	2 (0.4-12)	–	ab	ab	ab
2018	6 (1-302)	6 (1-302)	19 (6-56)	2	2 (1-24)	–	ab	b	b
White-faced Storm Petrel (KW all items: $\chi^2 = 128.46$, df = 3, P < 0.001); **(**fragments: $\chi^2 = 7.52$, df = 3, P = 0.06)**;** (pellets: $\chi^2 = 13.1$, df = 3, P < 0.001)									
1987–89	9 (0.2-236)	4 (0.2-236)	12 (1-140)	–	–	–	a	-	a
1999–2004	5 (0.1-115)	4 (0.2-115)	13 (1-55)	0.1	1 (0.5-3)	–	b	-	a
2009–2014	5 (0.1-65.5)	4 (0.1-65.5)	16 (2-37)	–	–	–	b	-	b
2018	5 (0.6-201)	4 (0.6-201)	13 (2-52)	4 (2-8)	2 (0.8-25)	–	b	-	ab
Broad-billed Prion (KW all items: $\chi^2 = 3.8$, df = 3, P = 0.29); (fragments: $\chi^2 = 1.18$, df = 3, P = 0.76); (pellets: $\chi^2 = 1.18$; df = 3; P = 0.06)									
1987–89	19 (1-160.5)	17 (1-161)	24 (3-68)	6	2 (1-3)	16	-	-	-
1999–2004	20 (1-223)	17 (1-223)	23 (2-60)	5 (1-8)	3 (2-11)	–	-	-	-
2009–2014	18 (0.3-283)	17 (0.3-283)	21 (4-48)	16 (8-20)	2 (1-2)	–	-	-	-
2018	17 (1-190)	16 (1-190)	23 (5-49)	3 (1-23)	6 (1-11)	–	-	-	-
Great Shearwater (KW all items: $\chi^2 = 15.14$, df = 3; P < 0.05); (fragments: $\chi^2 = 6.96$, df = 3; P = 0.07); (pellets: $\chi^2 = 9.77$, df = 3; P < 0.05)									
1989	21 (0.8-196)	16.4 (0.8-196)	23 (0.9-49.4)	–	4 (1-6)	–	a	-	a
1999–2004	15 (0.3-812)	14 (0.3-812)	20 (3-48)	–	–	–	b	-	b
2009–2014	17 (0.4-357)	16 (1-357)	19 (5-41)	2 (1-104)	6 (0.4-99)	–	ab	-	ab
2018	15 (0.5-990)	14 (1-990)	22 (6-63)	7 (1-59)	2 (1-122)	7 (1-12)	b	-	ab

Table S5.4 Median (range) thickness (mm) of plastic items (overall and per category) ingested by seabirds breeding on Inaccessible Island collected during roughly decadal time periods. Kruskal–Wallis test results are indicated in parentheses next to each taxon name. Significance indicates the Dunn's test results for which the thickness of items, among time periods not sharing any letters, were significantly different.

Species	All items median (range)	Hard fragments	Industrial pellets	Flexible pieces	Thread-like plastics	Foamed plastics	Significance (1)	(2)	(3)
***Fregetta* storm petrels (**KW all items: χ^2 = 4.42, df = 3, P = 0.22); (fragments: χ^2 = 37.03, df = 3, P < 0.01); (pellets: χ^2 = 9.32, df = 3, P < 0.05*)									
1987–89	1.1 (0.1-3.9)	1 (0.1-3)	2 (0.6-3.9)	0.2 (0.2-0.3)	0.2 (0.1-0.5)	–	-	a	-
1999–2004	1.1 (0.01 -3.4)	1 (0.3-3.2)	2.3 (0.9-3.4)	0.1 (0.01-0.2)	0.3 (0.2-0.3)	–	-	ab	-
2009–2014	1.2 (0.1-4)	1.2 (0.1-3.1)	2.5 (0.7-4)	–	0.3 (0.1-0.5)	–	-	b	-
2018	1.3 (0.1-4.4)	1.3 (0.3-4.4)	2.5 (1-3.9)	0.2	0.2 (0.1-0.5)	–	-	c	-
White-faced Storm Petrel (KW all items: χ^2 = 163.76, df = 3, P < 0.001); (fragments: χ^2 = 31.09, df = 3, P < 0.001); (pellets: χ^2 = 13.33, df =3, P < 0.05)									
1987–89	1.7 (0.2-5.5)	1.1 (0.2-3.4)	2.1 (0.2-5.5)	–	–	–	a	ab	a
1999–2004	1.2 (0.03-3.9)	1 (0.03-3.4)	2.1 (0.5-3.9)	0.1	0.3 (0.3-0.5)	–	b	a	a
2009–2014	1.3 (0.2-5.5)	1.2 (0.2-3.3)	2.4 (0.7-4.1)	–	–	–	c	b	b
2018	1.2 (0.03-5)	1.1 (0.03-5)	2 (0.5-4.1)	0.3 (0.2-0.4)	0.4 (0.1-2.1)	–	b	a	ab
Broad-billed Prion (KW all items: χ^2 = 6.35, df = 3, P = 0.10); (fragments: χ^2 = 73.86, df = 3, P < 0.001); (pellets: χ^2 = 1.31, df = 3, P = 0.73)									
1987–89	1.5 (0.02-5.3)	1 (0.1-3.8)	2.8 (1.1-5.3)	0.02	0.3 (0.2-0.4)	1.5	-	a	-
1999–2004	1.3 (0.01-5.5)	1 (0.01-5.5)	2.8 (0.5-4.3)	0.1 (0.02-0.2)	0.2 (0.2-0.5)	–	-	a	-
2009–2014	1.5 (0.02-4.6)	1.3 (0.1-4.6)	2.8 (1.1-3.9)	0.1 (0.02-0.2)	0.2	–	-	b	-
2018	1.4 (0.03-4.6)	1.4 (0.03-4.6)	2.7 (1-3.8)	0.3 (0.03-0.3)	0.3	–	-	b	-
Great Shearwater (KW all items: χ^2 = 19.40, df = 3, P < 0.001); (fragments: χ^2 = 5.38, df = 3, P = 0.15); (pellets: χ^2 = 8.31, df = 3, P = 0.04)									
1989	1.5 (0.3-4.3)	0.97 (0.4-3.7)	2.9 (1.5-4.3)	–	0.3	–	a	-	-
1999–2004	1.2 (0.02-6.7)	1.1 (0.02-6.7)	2.7 (0.9-4.2)	–	–	–	b	-	-
2009–2014	1.2 (0.1-6.1)	1.1 (0.2-6.1)	2.6 (0.7-3.8)	0.1	0.3 (0.2-1.2)	–	b	-	-
2018	1.2 (0.01-11.9)	1.1 (0.01-11.9)	2.7 (1.2-5.1)	0.2 (0.01-0.9)	0.2 (0.1-3.3)	0.9 (0.3-1.3)	b	-	-

*Dunn's post-hoc test failed to show where differences were

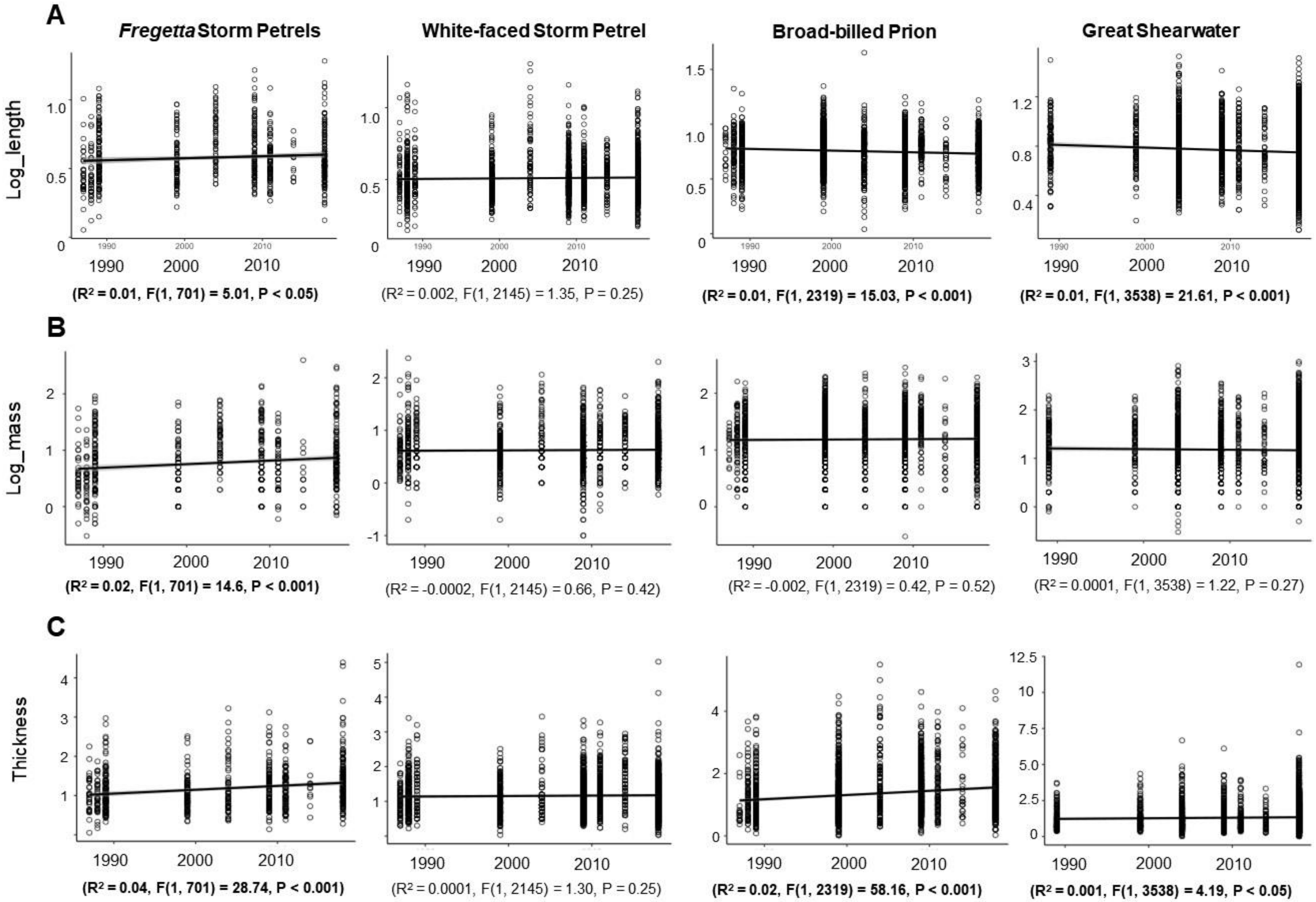


Fig. S5.1 Results of the linear regression models of log transformed length (A), mass (B) and untransformed thickness (C) of hard fragments as the response variables and year as the predictor variable. Results are indicated beneath each figure, and in bold if significant (P < 0.05).

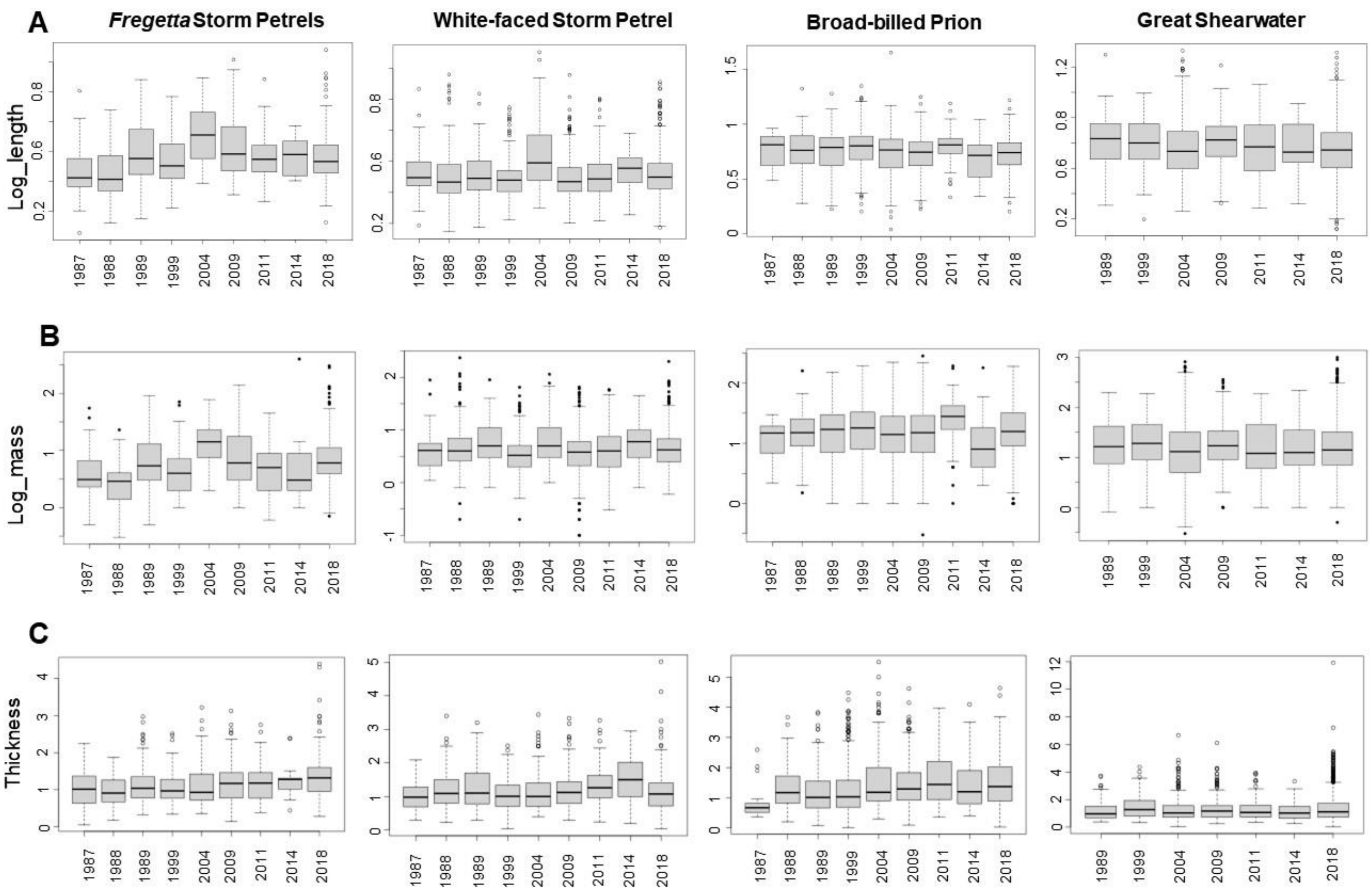


Fig. S5.2 Boxplots indicating the median (black line), interquartile range (grey box), values within 1.5 times interquartile range (whiskers) and outliers (values outside 1.5 times the inter-quartile range) of length (A), mass (B) and thickness (C) of hard fragments recorded in each sampling year in each taxon.

Table S5.5 Number of items in each plastic category that delivered a polymer match > 70% after processing with FTIR spectroscopy, and the total number of items processed with FTIR spectroscopy in parentheses. thread = thread-like plastics; flex = flexible plastics; foam = foamed plastics.

	***Fregetta* storm petrels**				**White-faced Storm Petrel**				**Broad-billed Prion**					**Great Shearwater**				
	fragment	pellet	thread	flex	fragment	pellet	thread	flex	fragment	pellet	thread	flex	foam	fragment	pellet	thread	flex	foam
Period 1																		
1987	32(34)	9 (10)	1 (1)	-	38 (38)	100 (100)	-	-	19 (19)	25 (25)	-	-	-	-	-	-	-	-
1988	37 (39)	18 (18)	3 (3)		62 (62)	-	-	-	43 (43)	61 (61)	1 (1)	-	0 (1)	-	-	-	-	-
1989	31 (32)	39 (40)	3 (3)	2 (2)	-	-	-	-	38 (38)	14 (14)	2(2)	1 (1)	-	100 (100)	86 (86)	0 (1)	-	-
1990	-	34 (35)	-	-	-	-	-	-	-	-	-	-	-	-	14 (14)	-	-	-
Period 2																		
1999	85 (85)	44 (44)	4 (5)	1 (1)	100 (101)	100(100)	-	-	100 (102)	100 (102)	1 (1)	4 (5)	-	89 (89)	26 (26)	-	-	-
2000	15 (15)	19 (19)	4 (5)	-	-	-	-		-	-	-	-	-	11 (11)	16 (16)	-	-	-
2004	0	0	4 (7)	1 (1)	-	-	3 (3)	2 (2)	-	-	3 (3)	1 (1)		0	58 (58)	-	-	-
Period 3																		
2006*	-	-	-	-		-	-	-	-	-	-	-	-	-	61* (61)	-	-	-
2009	100 (100)	24 (24)	-	-	100 (102)	100 (100)	-	-	100 (102)	100 (102)	-	1 (1)	-	100 (100)	39 (39)	2 (3)	1	-
2011	0	12 (12)	9 (13)	-	-	-	-	-	-	-				-	-	1 (1)	-	-
2014	-	-	2 (2)	-	-	-	-	-	-	-	2 (2)	1 (1)	-			4 (4)	0 (2)	-
Period 4																		
2018	100 (105)	18 (20)	16 (18)	1 (1)	100 (100)	100 (100)	3 (4)	1 (2)	100 (103)	100 (102)	3(4)	6 (12)	-	100 (103)	100 (103)	19 (20)	29 (42)	1 (3)
Total	**400 (410)**	**217 (222)**	**46 (57)**	**5 (5)**	**400 (403)**	**400 (400)**	**6 (7)**	**3 (4)**	**400 (407)**	**400 (406)**	**12(13)**	**14 (21)**	**0 (1)**	**400 (403)**	**400 (403)**	**26 (29)**	**30 (45)**	**1 (3)**

* Samples were collected on Nightingale Island during March 2006.

Table S5.6 Proportions of polymers (> 70% match) recorded for each plastic category, taxon and time period.

	Hard fragments			Industrial pellets				Thread-like plastic			Flexible plastic		Foamed plastic
	PE	**PP**	**PP-EPDM**	**PE**	**PP**	**PP-EPDM**	**PS**	**PE**	**PP**	**PP-EPDM**	**PE**	**PP**	**PU**
1987–1989													
Fregetta storm petrels	97%	3%	-	87%	13%	-	-	73%	18%	9%	100%	-	-
White-faced Storm Petrel	86%	14%	-	91%	9%	-	-	-	-	-	-	-	-
Broad-billed Prion	95%	5%	-	88%	12%	-	-	50%	50%	-	20%	80%	-
Great Shearwater	92%	7%	1%	91%	8%	-	1%	-	-	-	-	-	-
Overall	**93%**	**7%**	**< 1%**	**89%**	**11%**	**-**	**< 1%**	**67%**	**26%**	**7%**	**50%**	**50%**	**-**
1999–2004													
Fregetta storm petrels	90%	9%	1%	92%	8%	-	-	38%	63%	-	-	100%	-
White-faced Storm Petrel	93%	7%	-	86%	14%	-	-	67%	33%	-	100%	-	-
Broad-billed Prion	84%	16%	-	92%	7%	1%	-	100%	-	-	100%	-	-
Great Shearwater	93%	7%	-	92%	4%	2%	2%	-	-	-	-	-	-
Overall	**90%**	**10%**	**< 1%**	**90%**	**8%**	**1%**	**1%**	**57%**	**4%**	**-**	**75%**	**25%**	**-**
2009–2014													
Fregetta storm petrels	86%	14%	-	97%	3%	-	-	82%	18%	-	-	-	-
White-faced Storm Petrel	83%	16%	1%	87%	13%	-	-	-	-	-	-	-	-
Broad-billed Prion	84%	15%	1%	93%	6%	1%	-	-	100%	-	100%	-	-
Great Shearwater	76%	24%	-	93%	5%	2%	-	86%	14%	-	100%	-	-
Overall	**82%**	**17%**	**1%**	**92%**	**7%**	**1%**	**-**	**75%**	**25%**	**-**	**100%**	**-**	**-**
2018													
Fregetta storm petrels	89%	11%	-	83%	17%	-	-	44%	50%	6%	100%	-	-
White-faced Storm Petrel	75%	25%	-	89%	11%	-	-	100%	-	-	-	100%	-
Broad-billed Prion	76%	24%	-	85%	13%	2%	-	100%	-	-	-	200%	-
Great Shearwater	73%	27%	-	84%	13%	2%	1%	84%	16%	-	45%	55%	100%
Overall	**86%**	**14%**	**< 1%**	**89%**	**10%**	**< 1%**	**< 1%**	**69%**	**29%**	**2%**	**50%**	**50%**	**100%**

Table S5.7 Results of the Mann-Whitney U tests comparing the length, mass and thickness of hard fragments composed of polyethylene to those composed of polypropylene, per taxon and combined, across all time periods. Significant difference are in bold.

	Length	Mass	Thickness
***Fregetta* storm petrels**	Z = -0.44, P = 0.66	Z = -1.44, P = 0.15	**Z = -2.37, P < 0.05**
White-faced Storm Petrel	Z = -0.40, P = 0.69	**Z = -2.83, P < 0.05**	**Z = -3.16, P < 0.05**
Broad-billed Prion	Z = -1.91, P = 0.06	**Z = -2.88, P < 0.05**	**Z = -2.34, P < 0.05**
Great Shearwater	Z = -0.65, P = 0.52	Z = -1.49, P = 0.14	Z = -0.45, P = 0.65
All taxa	Z = -0.16, P = 0.87	Z = -1.956, P = 0.05	**Z = -3.19, P < 0.05**

Chapter 6

Synthesis – A comparison of meso- and microplastic trends in seabirds and on beaches

Sampling beached meso- and microplastics with a mesh sieve at a remote beach along the southern Cape coast, South Africa in 2015 (photo Peter Ryan).

Introduction

The exponential surge in mismanaged plastic waste and the resultant leakage into marine ecosystems constitutes a pressing environmental crisis. Reducing the influx of plastics into the environment is crucial, and multilevel mitigation strategies have been adopted to curtail this influx (Borrelle et al., 2020; Walker, 2021). The ubiquitous presence and persistence of marine plastics are well documented (Chapter 1) and can serve as baselines to monitor future plastic levels (Derraik, 2002; Law, 2017; Suaria et al., 2023). Assessing the effectiveness of these measures hinges on systematic monitoring of established baselines, with long-term monitoring being essential to identify patterns that might be overlooked in shorter timeframes (Ryan et al., 2009a; Ryan et al., 2020a). Using marine plastics as a monitoring tool however requires a thorough understanding of the dynamics of the system sampled. Changes in plastic densities at sea not only depends on the input rates, but also on the turnover rates (e.g. Chapter 1, Fig. 1.3; GESAMP, 2019; Ryan et al., 2020a), and without a clear understanding of the latter, interpreting changes in input rates from standing stock remains challenging (Ryan et al., 2020a).

Various methods are used to assess the abundance of marine plastics, and for meso- and microplastics at sea, surface net tows typically have been used (Morét-Ferguson et al., 2010; Cózar et al., 2014; Eriksen et al., 2014; Eriksen et al., 2023), but all have their challenges (Chapter 1). Seabirds that store ingested plastic, especially those of the same age and breeding status, offer a promising alternative, as they sample litter over large areas, produce larger sample sizes, and are less resource-intensive compared to other sampling methods (Ryan, 2016). Consequently, seabirds have been proposed as biomonitors of marine plastic pollution (van Franeker et al., 2011; Savoca et al., 2022; Rodríguez et al., 2024; Taurozzi and Scalici, 2024), but their utility in this regard seldom has been tested. This rationale formed the backbone of my thesis, allowing me to assess the use of seabirds as monitors of temporal and spatial variation in marine plastic pollution. Because larger species are capable of ingesting larger items, they may be better suited to assess changes in macrolitter items over time, where smaller petrels are better to assess changes in meso- and micro-sized buoyant plastic items. Accordingly, I assessed variation in the types and amounts of marine litter consumed by large and small procellariiform seabirds foraging in remote oceanic areas over decadal periods.

In Chapter 2, I used albatrosses and giant petrels to monitor decadal changes in macrolitter in the southern Indian and Southern Ocean. In Chapters 3-5, I focused on petrels, as they ingest items similar in size to those sampled with surface nets at sea, which is how floating plastic densities are usually estimated (Cózar et al., 2014; GESAMP, 2019; Eriksen et al., 2023). I first assessed the validity of using regurgitated skua pellets to sample ingested plastics, and then compared the plastics ingested by four petrel taxa to those found in the marine environment to gauge their utility, as a community, to be bioindicators (Chapter 3). I then assessed if plastic loads and types ingested by these four taxa foraging in the South Atlantic Ocean have changed over 30 years (Chapter 4) and assessed if changes in characteristics such as size and polymer type explain any observed patterns (Chapter 5). Overall, my thesis delivered six key findings.

1) Albatrosses and giant petrels are useful monitors of fishing practices and the origins and amounts of macrolitter items in remote oceanic regions

In Chapter 2, I investigated changes in the amount and types of litter ingested by albatrosses and giant petrels breeding at Marion Island in the southern Indian Ocean during the height of the toothfish industry (1996-1998) and two decadal periods thereafter (1999–2008 and 2009–2018). There was a strong correlation between fishing effort and fishery-related litter, with more items recovered during the peak of the toothfish industry, followed by a decline as fishing efforts were reduced (Ryan et al., 1997; Nel et al., 2000; Nel et al., 2002a; Nel et al., 2002b; Tuck et al., 2003; CCAMLR, 2017). Interestingly, the frequency of other litter types increased, especially in Grey-headed Albatrosses, mirroring results found at South Georgia (Phillips and Waluda, 2020), suggesting rising litter loads in the Southern Ocean. When limited to hard fragments, most items ingested by albatrosses and giant petrels were macro-sized (Chapter 2), whereas smaller taxa ingested no macroplastics (Chapter 3), supporting size-mediated litter selection (Roman et al., 2019b). I expected to find an increase in the frequency of other litter items ingested by albatrosses and giant petrels, aligning with findings of increasing macroplastics at sea, such as Ostle et al. (2019) who detected overall increases in macroplastics in the North Atlantic Ocean from 1957 to 2016. Long-term data on the densities of macroplastics in the southwestern Indian and Southern Ocean are sparse, complicating direct comparisons (Suaria et al., 2020; Connan et al., 2021; Eriksen et al., 2023). However, the observed increases in other litter types ingested by the same taxa in two independent studies (Phillips and Waluda, 2020; Chapter 2) suggest a rise in macroplastic density in the Southern Ocean. However, populations of albatrosses and giant petrels breeding on Marion Island have increased over the last decade or so (Ryan and Connan, unpubl. data; Stevens et al., in press), potentially at least partly explaining the increase in litter frequency. This is because the sampling was largely opportunistic, and amounts of litter items were just the totals collected per year, and not expressed as a rate per nest checked. At least some of the macrolitter items ingested by albatrosses and giant petrels could be traced to source, such as drink bottle lids, and highlights the utility of large procellariiform seabirds in this regard. This key finding addresses research aim 1 (Fig. 1.8).

2) Plastic from regurgitated skua pellets containing the remains of seabirds is a reliable method to sample ingested plastics

In Chapter 3, I validated the use of regurgitated skua pellets as an effective tool to sample plastics ingested by seabirds. Some studies have suggested that skua pellets may underestimate plastic loads (Ryan and Fraser, 1988; Provencher et al., 2019), but I found that plastics collected from regurgitated skua pellets were similar in size to those collected from petrel carcasses, even though the smallest of items may not be regurgitated by the skua but rather excreted, or overlooked during processing (Chapter 3).

Several studies have suggested using seabirds as biomonitors of marine plastics (Furtado et al., 2016; Avery-Gomm et al., 2018; Cartraud et al., 2019; Baes et al., 2024; Rodríguez et al., 2024), but few have compared ingested plastic items to those found within the seabirds' foraging areas at sea (Spear et al., 1995; Acampora et al., 2014; Roman et al., 2016; Hidalgo-Ruz et al., 2021), and mostly only compared it to plastics sampled on beaches (Kain et al., 2016; Lavers and Bond, 2016; Verlis et al., 2018). By comparing ingested to floating plastics collected at sea, I found that even though the petrels in my study differed in the types and colours of plastics they ingest, as a community they ingested plastic types similar to those sampled at sea with surface nets (Chapter 3). Therefore, plastics collected from regurgitated skua pellets containing seabird remains provide a simple and non-destructive method to sample floating marine plastics. This key finding addresses research aims 2 and 3 (Fig. 1.8).

3) Plastic loads ingested by seabirds have changed little from the 1980s

In Chapter 4, I found that plastic loads in seabirds foraging in the South Atlantic fluctuated between the 1980s to 2018, but no clear increasing or decreasing trends were observed, which matches the patterns observed among some other long-term studies on petrels (Vlietstra and Parga, 2002; Ryan, 2008; van Franeker and Law, 2015; Baak et al., 2020; Lavers et al., 2021), but not all (e.g. Rodríguez et al., 2024). I discussed how sea surface plastic studies generally produce mixed evidence of increases in floating micro- and mesoplastics over time (Law et al., 2010; Law et al., 2014). Over the last decade, some studies have linked increases in plastic densities at sea to the global growth of mismanaged plastic waste (Lebreton et al., 2018; Wilcox et al., 2020; Eriksen et al., 2023), but these studies were largely confined to the Northern Hemisphere. I found that taxa mostly confined to the South Atlantic year-round showed no evident trend in plastic loads, despite 4-6 fold increases in global plastic production over this time (Chapter 4). Great Shearwaters had greater plastic loads in 2018 compared to earlier years, suggesting that plastic densities in the North Atlantic, where they predominantly forage, might be increasing, as recent studies have indicated (Wilcox et al., 2020). However, another study of plastic ingestion in this species across its range found no evidence of a recent increase in plastic loads by number from 2010–2019 (Robuck et al., 2022). My plastic load data were based on numbers, but using plastic mass to analyse changes might have produced different results. For example, a single fragment might break into several pieces once ingested or be much larger compared to smaller industrial pellets, making mass a potentially more accurate measure for interpreting changes in plastic burdens over time (Provencher et al., 2017; Provencher et al., 2019; Bond and Lavers, 2023). When comparing mass instead of numbers, Robuck et al. (2022) found an increasing trend in plastic loads over time (but only when comparing years with large sample sizes). van Franeker and Law (2015), however reported an increase in both the number and mass of plastic loads in Northern Fulmars from the mid-1980s to mid-1990s, followed by a decrease in mass but not numbers towards 2000, with both stabilising over the last decade (up to 2012). Comparing plastic loads by mass instead of numbers in my study may therefore have elucidated different patterns. It is however unlikely that mass data would have influenced my finding that the numbers of industrial pellets are decreasing over time (Chapter 4), or that it would have matched the estimated 4-6-fold increase in global production of plastics during this time. I recommend that, during future sampling trips, samples be kept separate for each individual to enable analysis of mass data. This key finding addresses research aim 4 (Fig. 1.8).

4) The types of plastic ingested by seabirds have changed from 1987 to 2018

In Chapter 4, I reported how the proportions of industrial pellets ingested by seabirds decreased from 1987 to 2018, continuing the trend from the 1980s to 2000s (Ryan, 2008). This decrease may result in part from an increase in the density of other plastic types, but increases in other plastics were only recorded in storm petrels (Chapter 4). Other studies also failed to detect increases in other plastics ingested by procellariiform seabirds (Petry and Benemann, 2017; van Franeker et al., 2021). Additionally, the modest (if any) increase in the total amount of ingested plastic, or in the frequency of higher loads of ingested plastic in skua pellets, makes this unlikely. This links back to the at-sea data in Chapter 3, showing that only 2% of all items collected with surface net tows between 2016 and 2019 were industrial pellets, compared to the late 1970s, when industrial pellets dominated floating plastics in the South Atlantic (Morris, 1980; Ryan, 1988a). My data therefore indicate that the density of industrial pellets at sea have decreased over time, and is in line with trends reported in other ingestion studies (e.g. Vlietstra and Parga, 2002; van Franeker et al., 2011; Petry and Benemann, 2017), suggesting that industry initiatives aimed at reducing pellet leakage into marine environments are likely making a difference. This key finding highlights the importance of long-term monitoring to assess the efficiency of mitigation measures and addressed research aim 4 (Fig. 1.8).

5) There has been little change in the size of ingested plastics over time

In Chapter 5, I found that although the size and mass of ingested fragments and industrial pellets varied among time periods, there were no consistent trends over time. This key finding addresses research aim 5 (Fig. 1.8). Few studies have tested this over large timescales (Morét-Ferguson et al., 2010; Shugart et al., 2023) so it is difficult to assess the generality of my findings. However, the absence of discernible trends suggests that there may be a rapid turnover of marine plastics from the ocean surface (Chapter 5). I anticipated that the thickness of items might decrease over time as plastic product manufacturers may be reducing the thickness of products to reduce material costs (Worrel et al., 1995; Ho et al., 2005). However, I recorded no decreases, while the thickness of hard fragments ingested by *Fregetta* storm petrels and Broad-billed Prions increased over time. However, the mechanisms behind this remains unclear, and necessitates further investigation, potentially through dynamic modelling incorporating various processes governing the longevity of surface plastics at sea.

6) There has been an increase in the ratio of PP to PE hard fragments in the South Atlantic

In Chapter 5, I reported an increase in the ratio of PP to PE in hard fragments ingested by all four petrels in the Atlantic Ocean. This key finding addresses research aim 5 (Fig. 1.8). Several studies have investigated plastic loads in seabirds, but few determine the polymer types of ingested pieces (e.g. Avery-Gomm et al., 2016; Kühn et al., 2021; Robuck et al., 2022; Navarro et al., 2023), making comparative analyses challenging. However, Kühn et al. (2021) found a similar increase in the ratio of PP:PE among plastics ingested by Northern Fulmars in the Netherlands since the 1980s, but their results combined all ingested plastics, and the decrease in PE may reflect the decrease in the proportions of industrial pellets (mostly composed of PE) over time. In Chapter 5, I proposed several possible theories for this signal, but none is particularly plausible. This trend was not evident in hard fragments ingested by albatrosses and giant petrels over two decadal periods (Chapter 2), but these taxa ingest much larger items than the smaller petrels foraging in the Atlantic Ocean (Chapters 3 and 5). The increase in PP:PE is an intriguing trend, and further data, ideally from an independent source, would be valuable for affirmation. In light of this, I analysed beached plastics (~2–25 mm), collected along the southern Cape coastline of South Africa since the 1980s (Ryan and Moloney, 1990; Ryan et al., 2012; Ryan et al., 2018), to see if these trends in polymer type, as well as size of plastics, follow similar patterns, which may corroborate my findings.

Digging deeper: supporting data from beached plastics collected from 1984 to 2023

To assess if the changes in the characteristics of ingested plastics (Chapter 5) were possibly influenced by seabird selectivity, I investigated temporal variation in the characteristics of beached plastics (~2–25 mm) stranded at remote beaches along the southern Cape coastline from 1984 to 2023. This coastline borders the South Atlantic and southwestern Indian Oceans (Fig. 1.6), and at least macroplastics stranded on remote beaches mostly originate from offshore sources (Ryan et al., 2021; Ryan et al., 2024). Litter washed ashore on these beaches are likely to be most comparable to the suite of plastics available to the petrels in my study (Hidalgo-Ruz et al., 2021), as these beaches are the closest spatial proximity to the seabirds' foraging areas (Chapter 3; Fig. S3.4). For detailed methods see Ryan et al. (2018), but in brief: meso- and microplastics (~2-25 mm) were collected as part of a long-term study monitoring changes in litter on beaches along the South-African coastline (Ryan and Moloney, 1990; Ryan et al., 2012; Ryan et al., 2018). These plastics (~2-25 mm) were sampled from the upper 50 mm of sand by sieving through a 2-mm mesh sieve (square frame, 0.5 x 0.5 m, and 100 mm deep) in a 0.5-m-wide transect running up the beach from the most recent strand line to above the storm strand line, to ensure that all strandlines were sampled. Visible plastic items were picked from the sieved sample using tweezers, and the remains were floated in seawater to detect any cryptic items. Samples were collected in 1984, 1989, 1994, 1999, 2005, 2010 and 2015. For my analyses, I used data from 25 'remote' beaches (Table S6.1) located around the southern tip of Africa (Fig. 6.1). To limit the amount of local litter input, I chose beaches that were at least 100 km from major urban sources and that mostly contained macrolitter debris consistent with offshore sources, largely supported by the results of the preceding surveys (Ryan and Moloney, 1990; Ryan et al., 2018), but also from Ryan et al. (2021) that used plastic bottles on beaches to infer inshore or offshore litter inputs on selected beaches. Over 60% of items were collected at two beaches, Brandfontein and Koppie Alleen, both located inside nature reserves where ~2/3 of bottles were from foreign (offshore) sources (Ryan et al., 2021), and ~2/3 of lids sampled at De Hoop (Koppie Alleen) also originated from offshore sources (Ryan et al., 2024). The remainder of items were obtained from 23 beaches in a preferred order based on remoteness (composition of stranded macrolitter and distance from urban sources) to limit local, land-based inputs (Fig. 6.1; Table S6.1; Ryan et al., 2021). All items were processed for length, thickness, mass and polymer types (as done for marine and ingested plastics in Chapters 3 and 5). Unfortunately, the samples from 2015 were discarded after weighing, so only mass data were available for this year. In May 2023, I collected additional samples at Koppie Alleen and Brandfontein (Fig. 6.1), following the same protocol, to obtain a recent sample.

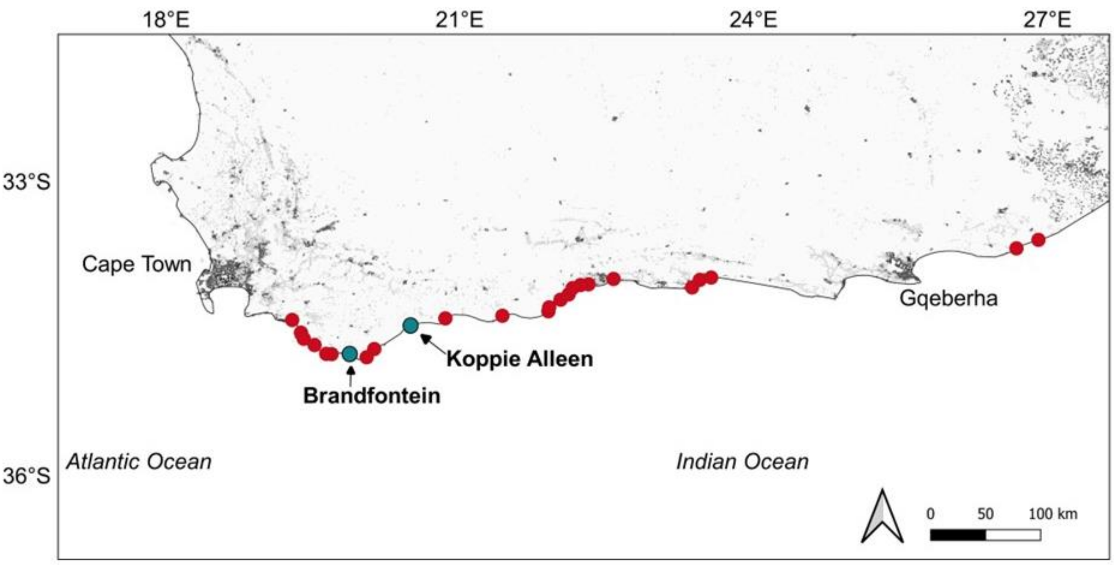


Fig. 6.1 Locations of the 25 beaches used in this study, with the locations of the two prioritised beaches, Koppie Alleen and Brandfontein arrowed. Grey shading indicates human population density.

In order to match the time periods used for the temporal comparison in Chapters 4 and 5, I combined the datasets as follows: 1984–1989 (to match 1987–89), 1994–1999 (to match 1999–2004), 2005–2010 (to match 2009-2014), 2015 (to match 2018) and 2023 (to represent the current situation, and to provide recent measurement and polymer data to compare with 2018 skua pellets). For each period I sampled 400 hard fragments and 400 industrial pellets (200 per year, except for 2015 and 2023, where 400 per year were processed) using the subsampling method used for ingested plastics in Chapter 5 (Table S6.1). Items were measured, weighed, processed for polymer types with FTIR spectroscopy and data were analysed as in Chapters 3 and 5.

The length and thickness of hard fragments over time did not follow any clear patterns (Fig. S6.1; Table S6.2), which is understandable as they are much more variable in size and shape, and therefore any temporal signal will be hard to detect. However, the mass differed, particularly between 2005–10 (median 13 mg) and 2023 (19 mg; $Z = -3.16$, $P < 0.05$; Fig. S6.1; Table S6.2). Onink et al. (2022) estimated that fragmentation of beached plastics can occur in an order of years, whereas buoyant plastics in the open ocean require decades, but did not factor in the dynamic processes governing turnover rates (Lebreton et al., 2019; Ryan et al., 2020a). Interpreting any trends in beached plastics requires an understanding of these processes that influence the arrival, storage and removal of plastics from beaches, but is complicated by legacy pollution in long-term sinks (Ryan et al., 2020a; Ryan et al., 2020b). For example, "new" items, that may have been at sea for prolonged periods of time, can be freshly deposited. Freshly deposited items may become buried over time and only resurface during storm events, or they may get washed away again (Bowman et al., 1998), making any interpretation of temporal variation in litter abundance and types on beaches more challenging than changes in plastics ingested by seabirds (Ryan et al., 2009a; Fok et al., 2017; Fanini and Bozzeda, 2018; Ryan et al., 2018; Ryan et al., 2020b; Andrady, 2022). In light of these challenges, I failed to detect any obvious trends in the size of hard fragments collected on beaches. This mirrors my findings in Chapter 5, where no discernible trends were recorded in the length and mass of hard fragments ingested by seabirds over the same study period.

The length, thickness and mass of industrial pellets differed among time periods (Fig. S6.1; Table S6.2). The decrease in mass in industrial pellets from 1984–89 to 2015 suggests that the size of beached industrial pellets has decreased over time. However, industrial pellets collected in 2023 were heavier (median 22 mg) compared to 2005–10 (18 mg; $Z = -4.68$, $P < 0.001$) and 2015 (17 mg; $Z = -5.92$, $P < 0.001$) (Fig. 6.2; Table S6.2). Pellets that remain in beaches for many years presumably become smaller over time due to physical abrasion. And as fewer new pellets arrive (Law et al., 2010; Morét-Ferguson et al., 2010; van Franeker and Law, 2015; Chapter 4), the average size will slowly decrease. This mirrors the general findings that industrial pellet loss into the environment have decreased over time (Law et al., 2010; Morét-Ferguson et al., 2010; van Franeker and Law, 2015; Chapter 3). The increase in pellet mass observed in 2023 runs counter to this trend, but presumably results from two recent large pellet spills at sea off the South African coast. The first occurred on 10 October 2017, when 49 tonnes of PE pellets were lost from two containers from a ship in Durban harbour (Fig. 6.3). These pellets had a diameter of 5 ± 0.5 mm and weighed 24 ± 1.3 mg (Mofokeng and Glassom, 2022). A large proportion of these pellets dispersed into the southwestern Indian Ocean, washing up on beaches all along the south and east coasts of South Africa, and reaching beaches west of Cape Agulhas by early December (Schumann et al., 2019). A second spill, possibly even larger than the first, occurred off the southern Cape coast in October 2020. Details of this event have been largely suppressed in terms of the legal settlement which saw a large insurance payment to help recover spilled pellets from beaches (https://www.iol.co.za/capeargus/news/watch-transparency-called-for-after-plastic-nurdle-spill-contaminated-cape-coastline-38fcc6d5-3858-45c8-a9fe-42ce8254c261), but it resulted in very high densities of pellets washing up at southern Cape beaches. The major inputs from these two events likely explain the greater average pellet mass in 2023 (Fig. 6.2).

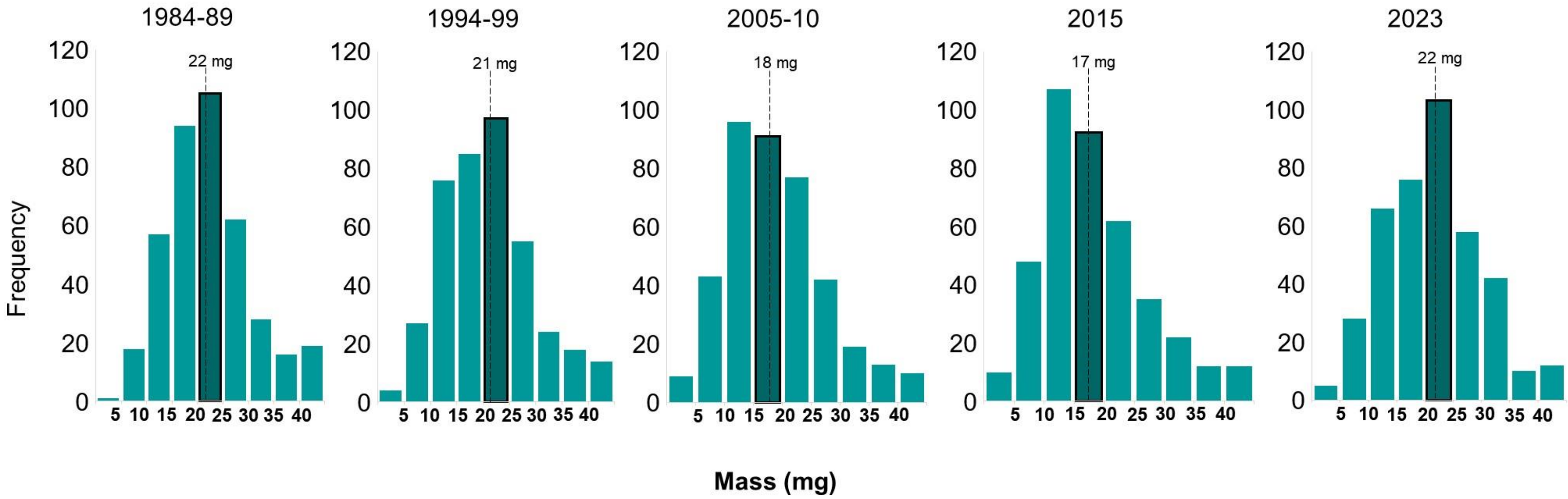


Fig. 6.2 Histograms of industrial pellet mass (mg) recorded across time periods. Median mass for each period indicated with a dashed line.

Fig. 6.3 Thousands of industrial pellets floating among other litter items in Durban harbour, shortly after two containers were lost from a ship during a storm event on 10 October 2017 (photo Roelf Daling).

Among hard fragments, polymer ratios differed among time periods ($\chi^2 = 11.74$, df = 3, $P < 0.05$; PP, PP-EPDM and other polymers pooled), with the ratio of PP:PE nearly doubling from 1984–89 to 2023 (Fig. 6.4A). Among industrial pellets, the ratios also differed ($\chi^2 = 37.64$, df = 3, $P < 0.05$; PP, PP-EPDM and other polymers pooled), mainly due to the sharp increase in the ratio of PE:PP recorded in 2023 (Fig. 6.4B). When only comparing up to 2005-2010 there was no significant change in the polymer composition of pellets ($\chi^2 = 0.69$, df = 2, P = 0.71). Possible reasons for the change in polymer ratios observed in hard fragments are discussed in Chapter 5, but the fact that the same pattern is observed in both abiotic and biotic compartments indicates that it is not a bird-related artefact, and points to a shared variable influencing the observed ratios. The increase in ratios of PE:PP in industrial pellets recorded in 2023 reflects the influx of virgin PE pellets from the 2017 and 2020 pellet spills.

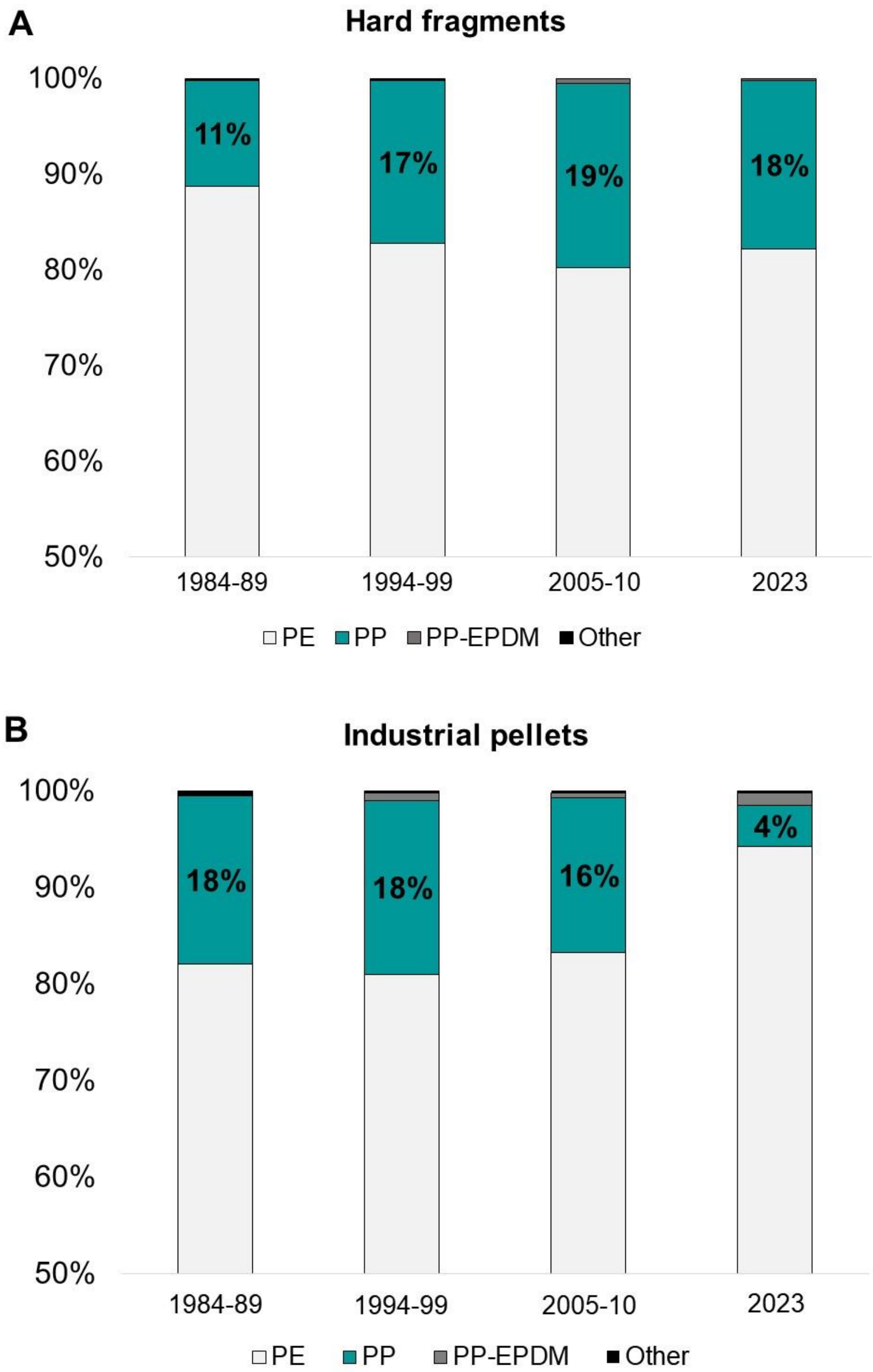


Fig. 6.4 Polymer types recorded in hard fragments (A) and industrial pellets (B) collected on beaches along the southwestern Cape coastline between 1984 and 2023. Other polymer types include structural reaction injection moulding (SRIM), Ethylene-vinyl acetate (EVA) and styrene acrylonitrile (SAN).

Long-term monitoring of beached plastics allowed me to corroborate the change in polymer ratios but not a change in thickness of fragments detected among plastics in seabirds breeding at Inaccessible Island. The observed increase in PP:PE ratios within hard fragments in both seabirds and beached plastics underscores the need for further investigation. Monitoring the characteristics of industrial pellets over time highlighted the importance of major spills at sea on regional pellet loads, and stresses the need for stricter measures to prevent such events in future. This study also highlights the significance of examining both biotic and abiotic compartments to identify parallels when monitoring trends in marine plastic pollution.

Seabirds as monitors of marine plastic pollution: requirements and challenges

To qualify as a biomonitor for marine plastic pollution, a species or community must meet certain criteria. They need to be present in sufficient numbers for robust sampling, have moderate exposure to plastic pollution and be easily identifiable and accessible for repeatable sampling (Bonanno and Orlando-Bonaca, 2018; Provencher et al., 2019; Lavers et al., 2021; Savoca et al., 2022; Rodríguez et al., 2024). The petrels in my study met most of these requirements, as they were sampled in large numbers and had moderate to high plastic loads (Chapters 3, 4 and 5). The albatrosses and giant petrels sampled in Chapter 2 had lower frequencies of ingestion, resulting in smaller sample sizes, making them less suitable for monitoring changes in plastic pollution (Furness, 1985; Ryan, 1987a; Ryan et al., 2016; Roman et al., 2019a), but the larger identifiable items they ingested were better for determining litter sources. Identifying prey remains of small petrels can be challenging (Provencher et al., 2019; Chapters 3 and 4). However, using a comprehensive reference collection and limiting sampling to months when only certain species or age groups are likely to be present (Chapters 3 and 4) can aid in accurate identification. Assigning litter items to specific species was easier for the albatrosses and giant petrels on Marion Island (Chapter 2), as regurgitates were collected in or around nests (Fig. 1.5). The most challenging limitation in my thesis is perhaps the difficulty of accessing the sampling sites. Both Inaccessible Island (~2 500 km from the nearest continent) and Marion Island (~1 800 km) are extremely remote and can only be reached by sea (Fig. 1.6). However, access to these sites can be arranged with the relevant governmental parties to pursue future studies, and adhering to the protocols highlighted in this study will ensure consistency and comparability.

A challenge in all ingestion studies is the uncertainty surrounding the retention times of plastics within seabirds and the potential offloading of plastics while at sea (Ryan, 2016;

Chapter 4). For example, Terepocki et al. (2017) noted that beach-wrecked Northern Fulmars regurgitated plastics during rehabilitation in captivity, which could mean that our inability to detect changes in plastic loads over time may simply be because petrels that accumulate large plastic loads regurgitate them at sea. To date, there is no evidence of birds offloading plastics at sea, and coupled with the strong right-skew in plastic loads among petrels (Ryan, 2016), including all four small petrels sampled (Chapter 4), it is unlikely that there is a specific threshold beyond which they regurgitate plastics. Additionally, the rate of passage of plastics through seabirds' gastro-intestinal tracts, and the maximum excretable size, remains uncertain (Ryan, 2008; Ryan, 2015a; Terepocki et al., 2017). The longer plastics are retained in seabirds, the smaller they become (Ryan and Jackson, 1987; van Franeker and Law, 2015; Ryan, 2015b; Nania and Shugart, 2021), but the rate of wear inside petrels is still debated, and studies have reported ranges from weeks months (Day et al., 1985; Ryan and Jackson, 1987; van Franeker and Law, 2015).

The potential for offloading at sea, and the uncertainty regarding retention times and passage rates of ingested plastics, could influence plastic loads (Chapters 2 and 4) and sizes (Chapters 2, 3 and 5) in seabird stomachs. While understanding the retention times of ingested plastics in seabirds would enhance data interpretation, collecting such data under natural conditions remains challenging. Factors such as the sampling method, sampling area and season can also influence plastic loads in seabirds (Ryan, 1988b; Rodríguez et al., 2018; Provencher et al., 2019; Lavers et al., 2021; Chapters 1 and 4). Sampling the entire gastro-intestinal tract ensures all plastics are recovered (whereas with lavage some items may be retained; Provencher et al., 2019; Lavers et al., 2021), but the cause of death (e.g. sampling birds that have died of unknown causes which may have led to inflated plastic loads) may affect the results. To minimize the potential influence of individuals egesting plastics while at sea, retention times of ingested plastics, the sampling method used, age of birds and season on plastic loads obtained in my study, I 1) employed large sample sizes and included multiple taxa of varying sizes, 2) sampled birds early in the breeding season, before the eggs hatched, to ensure that plastics are collected before they are offloaded to chicks (Ryan, 1988b; Ryan, 2015c; Ibañez et al., 2020), 3) adhered to a consistent sampling protocol (that presumably samples the entire gastro-intestinal tract; Chapter 3) at the same breeding island over 30-years (Chapters 3, 4 and 5), and 4) only sampling breeding or pre-breeding adults (or chicks that are readily identifiable due to poorly developed bones).

Concluding remarks

This thesis used two datasets spanning multiple decades, using the same taxa from the same locations, to assess plastic loads in seabirds. It thus contributes an important perspective and useful data to the expanding field of marine plastic pollution. Albatrosses and giant petrels were useful monitors of fishing practices and litter sources in remote oceanic areas over time. The recent decadal increase in litter amounts, especially in Grey-headed Albatrosses, suggests a rise in litter densities in the southern Indian Ocean, but my results failed to account for growing population size of these seabirds breeding on Marion Island over the last decade. The failure to detect clear trends in ingested plastic loads in the smaller petrels, but matching trends in plastic types, mirrors the general trend in marine plastics sampled with surface nets at sea. Petrels also were valuable indicators of trends in the polymer composition of small buoyant plastics, which parallel variations seen in beached plastics. The increasing ratio of PP to PE observed in ingested and beached secondary meso- and microplastics suggests a common influence, but the cause of this change is not apparent. Future research should include polymer identification to better understand the drivers behind the growing prominence of PP and to determine if this trend continues. These findings, across both biotic and abiotic compartments, underscore the importance of seabirds as bioindicators and the need for ongoing monitoring to accurately assess trends in marine litter densities and types.

Considering the rising global production of plastics (Fig. 1.2), I was expecting to observe an increase in plastic loads across all taxa sampled in my study, but this was only found in albatrosses and giant petrels over the last decade. I did not find an overall clear trend in plastic loads in petrels over nearly four decades (Fig. S4.2). If plastic inputs into the open ocean from land and rivers are overestimated, as suggested in Chapter 4, we may not expect to find a clear trend. The large uncertainty surrounding the intricate dynamics governing turn-over rates of small buoyant plastics at sea however, such as the resurfacing, recirculation, transport, ingestion by biota and sedimentation of plastics (Fig. 1.3), makes the interpretation of these findings challenging (Ryan et al., 2020a). Very few empirical studies have tried to estimate turnover and fragmentation rates of small buoyant plastics at sea. Andrady (2022) used experimental studies to show that little fragmentation of floating plastics occurs in the open ocean, and concluded that most secondary meso- and microplastic fragments found at sea likely occurred due to pre-weathering on land, especially on beaches. Onink et al. (2022) simulated fragmentation rates of open ocean and beached plastics and found that plastics at sea require decadal time periods to undergo fragmentation, whereas on beaches this process can occur within years. Plastic colour can further influence weathering in the environment, as Key et al. (2024) found that degradation rates of PP and HDPE varies depending on the colorants used. Koelmans et al. (2017) used a whole ocean emission-fragmentation-settling mass balance model to theorise how plastic loads at sea would change under different input scenarios and predicted that in a zero-plastics emissions scenario, all plastics would be removed from the ocean surface within three years. However, they did not consider the dynamic input and loss of plastics due to stranding, release, and recirculation (Lebreton et al., 2019).

Studies assessing turnover rates of plastics at sea often exclude essential processes that govern the arrival, fragmentation, resurfacing, biofouling and sedimentation of items. To date, there are few experimental data available to parameterise a model to anticipate what existing plastics at sea will do under varying emission and fragmentation scenarios. A better understanding, ideally from field-based experiments including different polymer types, of the complex physical processes governing size-based fragmentation are needed for better estimates. However, a common thread in these studies is that plastics are more likely to fragment on beaches than at sea, and therefore preventing items from reaching beaches in the first place, or removing them once there, may ultimately reduce the number of small buoyant plastics in the open ocean. Both formal and informal beach cleaning removes large quantities of waste before it can disperse into the open ocean (Haarr et al., 2020), and beach cleanup events have the added advantage of fostering environmental stewardship among citizens (Jorgensen et al., 2021). Growing public awareness drives (Willis et al., 2018), the use of flagship species such as turtles (Fig. 1.1) in behavioural change campaigns (Eagle et al., 2016), stricter enforcement of policies (e.g. MARPOL ANNEX V; Walker, 2021), and the implementation of interception devices (e.g. Sidek et al., 2016; Gkanasos et al., 2021) coupled with initiatives to reduce plastic use at source (e.g. UN treaty negotiations), all will help to reduce the volumes of mismanaged waste produced, the amounts entering the sea and the impacts of plastic pollution in the environment.

The reduction in industrial pellet numbers and proportions recorded at sea and in seabird studies (Chapters 3 and 4), after the implementation of awareness campaigns and stricter regulations, shows that these factors have the potential to reduce litter loads at sea, and may support the lack of overall increasing trends in plastics loads observed in my study (Chapter 4). This suggests that these campaigns are possibly making a difference. It is however only with improved dynamic models, and continued systematic monitoring, that we can confidently assess the efficacy of these measures, and my thesis provides support to use seabirds, as monitors to track temporal variations in plastic densities, types and characteristics at sea.

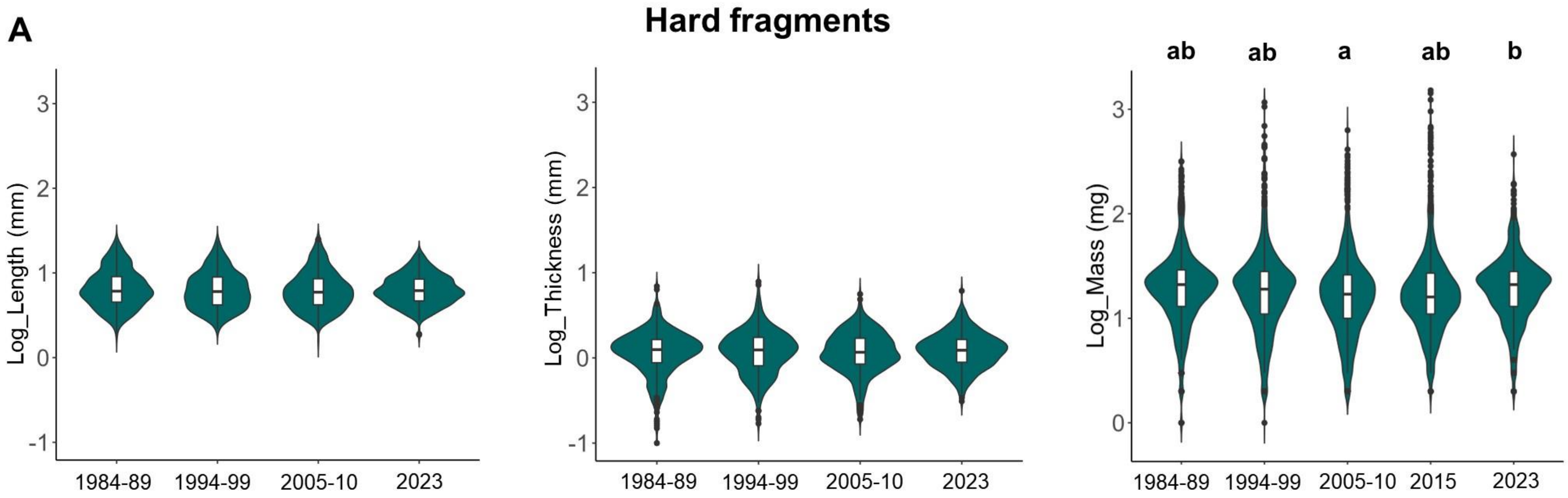
Supplementary materials: Chapter 6

Table S6.1 Number of items (hard fragments and industrial pellets) processed per time period for each beach. Beaches arranged in the order of preference in terms of location and remoteness to limit local, land-based inputs, based on the composition of stranded macrolitter and distance from urban sources.

Beach	**1984-89**	**1994-99**	**2005-10**	**2015**	**2023**	**Total**
Koppie Alleen	301	279	432	109	400	1521
Brandfontein	119	83	255	32	400	889
De Mond	30	67	75	217	0	389
Quoin Point	46	68	0	61	0	175
Struisbaai	54	81	5	20	0	160
Die Dam	6	5	5	1	0	17
Pearly Beach	42	55	7	54	0	158
Franskraal	13	22	1	3	0	39
Die Plaat	0	31	2	1	0	34
Visbaai	40	0	12	37	0	89
Vleesbaai	78	0	3	42	0	123
Stilbaai	0	0	0	1	0	1
Dana Baai	0	2	3	1	0	6
Diaz Beach	0	22	0	1	0	24
Klein Brak	0	8	0	30	0	38
Groot Brak	16	0	0	3	0	19
Glentana	0	0	0	38	0	38
Witsand	0	0	0	65	0	65
Wilderness	10	0	0	5	0	15
Plettenberg Bay	0	0	0	14	0	15
Keurboomstrand	0	0	0	5	0	9
Nature’s Valley	9	39	0	7	0	55
Hermanus	25	0	0	15	0	40
Port Alfred	5	23	0	30	0	58
Kenton-on-Sea	0	15	0	8	0	23
Total	800	800	800	800	800	4000

Table S6.2 Length (mm), thickness (mm) and mass (mg) of hard fragments and industrial pellets collected on 25 beaches along the southern coast, South Africa from 1984 to 2023. Kruskal Wallis test results indicated in parentheses, and in bold if significant.

Period	Hard fragments	Industrial pellets
Length	(KW:χ^2 = 4.12, df = 3, P = 0.24)	(KW: χ^2 = 48.20, df = 3, **P < 0.001**)
1984–89	6.1 (1.8-24.1)	4.0 (2.1-6.8)
1994–99	6.0 (2.2-23.6)	3.9 (1.2-7.5)
2005–10	5.9 (1.6-24.9)	3.8 (2.2-5.6)
2015	–	–
2023	6.2 (1.9-17.1)	4.1 (1.7-5.5)
Thickness	(KW: χ^2 = 0.42, df = 3, P = 0.94)	(KW: χ^2 = 23.6, df = 3, **P < 0.001**)
1984–89	1.3 (0.1-6.9)	2.5 (0.9-4.7)
1994–99	1.2 (0.2-7.9)	2.4 (0.8-4.6)
2005–10	1.2 (0.2-5.6)	2.3 (0.7-4.1)
2015	–	–
2023	1.2 (0.3-6.1)	2.4 (0.9-4.1)
Mass	(KW: χ^2 = 13.01, df = 4, **P < 0.05**)	(KW: χ^2 = 74.1, df = 4, **P < 0.001**)
1984–89	17 (1-318)	22 (3-110)
1994–99	14 (1-1164)	21 (4-110)
2005–10	13 (2-630)	18 (2-210)
2015	15 (2-1516)	17 (3-92)
2023	19 (2-371)	22 (3-61)


A
Hard fragments
Log_Length (mm)
Log_Thickness (mm)
Log_Mass (mg)
ab
ab
a
ab
b
3
2
1
0
-1
1984-89
1994-99
2005-10
2015
2023

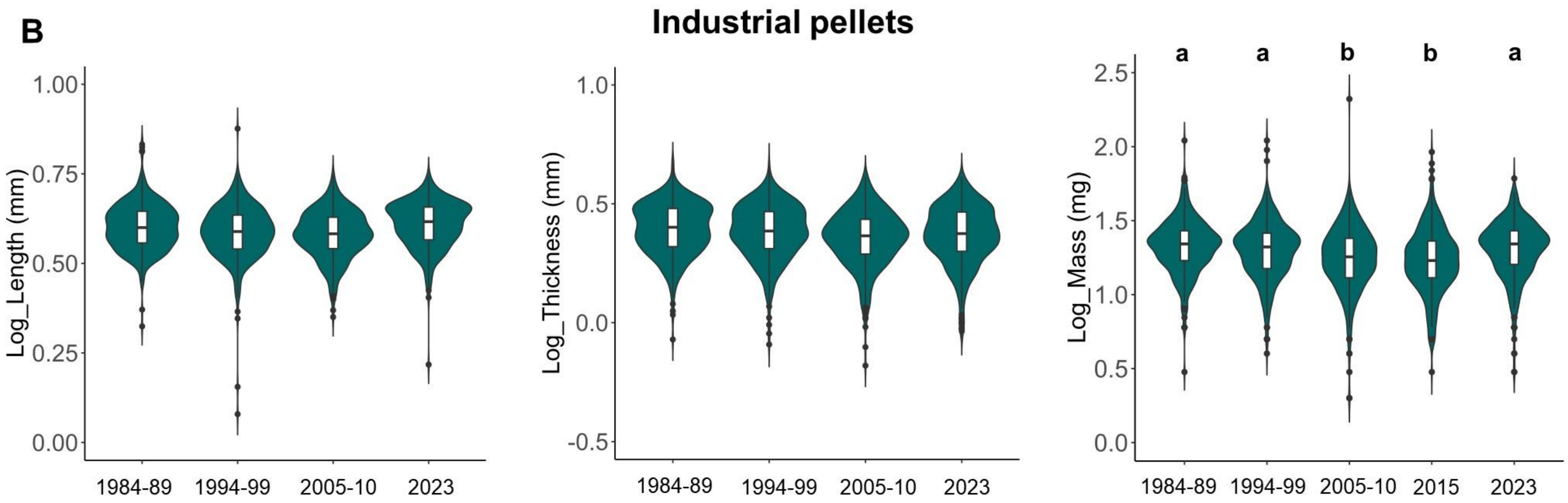


Fig. S6.1 Plots comparing the log transformed length, thickness and mass of hard fragments (A) and industrial pellets (B) among the four/five study periods. Length and thickness not recorded in 2015. Boxplots indicate the median (dark line) and interquartile range while violin plots indicate the kernel probability density at different values where the width of each curve corresponds to the frequency of data points in that region. Letters above boxplots signify which time periods differed significantly (post hoc Dunn's test results; $P < 0.05$). Note differences in scaling on y-axes (B).

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