Influence of fisheries on the foraging ecology and demography of breeding black-browed albatross *Thalassarche melanophris* in the Falkland Islands

Submitted by Amanda Kuepfer to the University of Exeter as a thesis

for the degree of Doctor of Philosophy in Biological Sciences, March 2023

I certify that all material in this thesis which is not my own work has been identified and that any material that has previously been submitted and approved for the award of a degree by this or any other University has been acknowledged.

Signature: …………………………………………………………

Abstract

Maintaining healthy marine ecosystems in the face of worsening and accelerating human impacts urgently requires a globally coordinated and multidisciplinary approach. Seabirds are useful indicators of the status and health of marine ecosystems – they are higher-level marine predators that are relatively visible and accessible to researchers. However, their ability to cross political boundaries and high seas in the absence of a global management framework for biodiversity also makes them vulnerable to anthropogenic threats. For example, seabirds interact directly and indirectly with fisheries across the world's oceans, including through competition for resources, by scavenging on fisheries waste, and through incidental capture in fishing gear. Improving our understanding of seabird-fisheries interactions therefore not only serves the target species, but has the ability to make a vital contribution to global initiatives and multi-disciplinary research aimed at improving the protection and sustainable management of our oceans and climate.

In the Southwest Atlantic, where seabird-fishery overlap is amongst the highest globally, the black-browed albatross *Thalassarche melanophris* (hereafter BBA) represents the main scavenger at fishing vessels, and is also the dominant seabird species of incidental catch (bycatch). The Falkland Islands hold the world's largest population (>70% of breeding pairs), which is currently stable / increasing. However, the extent to which fisheries influence this trend remains poorly understood. Given the global significance of this population, as well as its importance to the Patagonian Shelf Large Marine Ecosystem, gaining an improved understanding of the nature and extent of fishery interaction is integral to biological conservation and ecosystem functioning, and supports fisheries management across the Southwest Atlantic.

Following a general introduction that forms **Chapter 1**, this thesis applied an integrated, multidisciplinary approach to help improve our understanding in relation to (1) BBA bycatch mitigation, (2) BBA diet structure and discard use, and (3) key foraging areas and broad-scale overlap with fishing fleets across the Patagonian Shelf.

Chapter 2 found that, compared to continuous discarding, batch discarding significantly reduced seabird abundance and gear collisions, and zero discarding eliminated gear collisions altogether. The findings validate batch discarding as an effective seabird-bycatch mitigation measure in trawl fisheries where full discard retention is not possible, but highlight the importance of complete waste storage between batches. Further, the positive relationship between bird abundance and collision rates supports the use of abundance as a proxy for collision rates in past and future studies.

Chapter 3 showed that BBA chicks are predominantly fed natural prey. However, discards form an important component in some years, specifically showing a significant increase in years of higher discard availability, higher sea surface temperature anomalies, and lower breeding success. The findings suggest that, although natural prey are the preferred diet, BBA switch to discards when natural foraging conditions are compromised. While fishery discards may act as a buffer, they do not appear to fully offset poor natural foraging conditions for breeding albatrosses in the long term.

Combining stomach content analysis with stable isotope analysis in **Chapter 4** provided complementary diet information. Findings confirmed the importance of natural prey for chicks, but highlighted key diet differences between colonies and years. Although less important, results show that discards are still taken regularly, thus exposing a large proportion of breeding adults to a bycatch risk.

Finally, **Chapter 5** showed that foraging areas of Falkland Islands breeding BBA correspond with productive shallow waters. During egg incubation and chick brooding, these areas overlapped only moderately with fishing activity, and did so predominantly with trawlers within the Falklands Conservation Zones (FCZs) and the Argentine Exclusive Economic Zone (AEEZ).

The General Discussion that forms **Chapter 6** assesses the findings of this thesis in light of the available literature, and discusses the implications for management and future research. Overall, the thesis suggests that BBA-fisheries interactions are relatively low during breeding, and that fisheries have little influence on the ecology and demography of BBA in the Falkland Islands during this period. Efforts to limit fisheries waste and vessel attractiveness would therefore be of overall benefit to this population during breeding. However, important knowledge gaps persist, including in relation to the significance of discards to non-breeders (juvenile and immature birds; adult birds during winter), and the potential indirect impacts that fisheries may have on BBA through food-web alterations. The discussion highlights the importance of interdisciplinary collaborations across national jurisdictions to ensure robust marine science and successful conservation.

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Author’s Declaration

All work in this thesis was completed by A Kuepfer with contributions from collaborators and supervisors as outlined for each chapter.

Chapter 2 – Amanda Kuepfer, Richard B Sherley, Paul Brickle, Alexander Arkhipkin, Paulo Catry, Stephen C Votier

Conceptualisation – AK; SCV; Data collection – AK; Data processing – AK; Data analysis – AK, RBS, SCV; Project administration – AK; Writing original draft – AK; Manuscript review & editing – all; Funding acquisition – AK; Additional resources – PB, AA; Supervision – SCV, PC, RBS, PB, AA.

Chapter 3 – Amanda Kuepfer, Stephen C Votier, Richard B Sherley, Francesco Ventura, Rafael Matias, Orea Anderson, Paul Brickle, Alexander Arkhipkin, Paulo Catry.

Conceptualisation – AK, PC, SCV; Data collection – AK, PC, OA, RM; Data processing – AK, RM; Data analysis – AK, RBS, SCV, PC, FV; Project administration – AK; Writing original draft – AK; Manuscript review & editing – all; Funding acquisition – AK, PC; Additional resources – PB, AA; Supervision – SCV, PC, RBS, PB, AA.

Chapter 4 – Amanda Kuepfer, Paulo Catry, Stuart Bearhop, Richard B Sherley, Olivia Bell, Jason Newton, Paul Brickle, Alexander Arkhipkin, Stephen C Votier

Conceptualisation – AK, PC, SCV, SB; Data collection – AK; Data processing – AK, OB, JN; Data analysis – AK, SCV, SB, RBS, JN, OB; Project administration – AK; Writing original draft – AK; Manuscript review & editing – all; Funding acquisition – AK, PC; Additional resources – JN, PB, AA, SCV; Supervision – SCV, PC, RBS, SB, PB, AA.

Chapter 5 – Amanda Kuepfer, José P. Granadeiro**, Francesco Ventura, Stephen C Votier, Richard B Sherley, Paulo Catry, Paul Brickle**

Conceptualisation – AK, PC, SCV; Data collection – PC, JPG, FV, AK; Data processing – JPG, FV, AK; Data analysis – AK, SCV, FV, RBS, PC; Project administration – AK, PC, JPG; Writing original draft – AK; Manuscript review & editing – AK, SCV, RBS, PC, FV, JPG; Funding acquisition – AK, PC; Supervision: SCV, RBS, PC, PB.

Ethics Statement and Permissions

All colony-based fieldwork conducted as part of this thesis has been approved by the University of Exeter ethics committee (eCORN002230), as well as by the Falkland Islands Environmental Committee. All work was carried out under permission issued by the Falkland Islands Government under Section 9 of the Conservation of Wildlife and Nature Ordinance 1999 (Research Licences number R29/2017 (granted to Dr. Paulo Catry), R34/2019 (granted to Amanda Kuepfer)). All at-sea work was undertaken by the Falkland Islands Government Fisheries Department (FIFD).

Publications Completed During PhD

PhD Chapters

Chapter 2 – **Kuepfer A**, Sherley RB, Brickle P, Arkhipkin A, Votier SC (2022) Strategic discarding reduces seabird numbers and contact rates with trawl fishery gears in the Southwest Atlantic. Biol Conserv 266:109462. doi: 10.1016/j.biocon.2022.109462

Chapter 3 – **Kuepfer A**, Votier SC, Sherley RB, Ventura F, Matias R, Anderson O, Brickle P, Arkhipkin A, Catry P (2022) Prey-switching to fishery discards does not compensate for poor natural foraging conditions in breeding albatross. ICES J Mar Sci. fsac069. doi: 10.1093/icesjms/fsac069

Chapter 4 – **Kuepfer A**, Catry P, Bearhop S, Sherley RB, Bell O, Newton J, Brickle P, Arkhipkin A, Votier SC (2023) Inter-colony and inter-annual variation in discard use by albatross chicks revealed using isotopes and regurgitates. Marine Biology. 170:46. doi 10.1007/s00227-023-04191-7

Other publications

Ventura F, Stanworth A, Crofts S, **Kuepfer A**, Catry P (2023) Local-scale impacts of extreme events drive demographic asynchrony in neighbouring top predator populations. Biol Lett 19:20220408. doi: 10.1098/rsbl.2022.0408

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Acknowledgements

This thesis is dedicated to my mum and dad, Lotti & Marco Küpfer, who nurtured my interest and love for nature, and who introduced me to the most magical of creatures – the albatross.

Thank you and much love, I am forever grateful.

This PhD was funded by Fortuna Ltd and through the Falkland Islands Government Environmental Studies Budget. Specific studies benefited from further financial support from the Fundação para a Ciência e a Tecnologia (FCT), and the National Environmental Research Committee (NERC) Life Science Mass Spectrometry Facility at the Scottish Universities Environmental Research Centre (SUERC). The South Atlantic Environmental Research Institute (SAERI) provided generous in-kind support. Further logistical support was provided by Falklands Conservation, the New Island Conservation Trust and the World Conservation Society.

I owe a great deal of thanks to all my supervisors for their support throughout my PhD. A special thanks goes to my primary supervisor, Steve Votier, for his trust in me as a distant student, and his solid interest, time and attention given to my individual projects. His expert knowledge, guidance and positivity have been instrumental to the success of my PhD and my development as a scientist. I am indebted to Paulo Catry who played a central role in kick-starting this PhD by welcoming me on-board his long-term black-browed albatross demographic study back in 2015. His steady source of wisdom, expertise and encouragement has been invaluable, and the privilege of spending so many months amongst the albatross made my dreams come true. Big thanks also extend to Richard Sherley, who always took the time and patience to discuss my work and provide important statistical guidance. His unique ability to explain and clarify statistics allowed me to go above and beyond what I thought possible. I am also very grateful to Paul Brickle for his every-day support, motivation, and availability, and for allowing me to work amongst a wonderful and energetic team at the Falkland Islands scientific hub that is SAERI.

The studies conducted for this PhD would not have been possible without the large amount of quality data collected by very many scientists and field assistants both before and during my PhD. Special thanks go to Francesco Ventura, Megan Tierney, Tash Gillies, Catlin Frankish, Anne Sophie Bonnet-Lebrun and Georgina Strange for their assistance and wonderful company during my time on New Island. Thank you to Sarah Crofts and Andy Stanworth of Falklands Conservation for facilitating my data collection on Steeple Jason Island. Rebecca Piontek and Olivia Bell kindly assisted with processing of dietary samples. I am also grateful to the captain and crew of the *Santa Mariña FV* and *Petrel FV* for their support with “operacíon tanque”, and to Ludovic Goyot for collecting fish species aboard research cruises funded by the Falkland Islands Fisheries Department. The Marine Predator Ecology and Conservation Group provided an excellent platform for sharing and discussing ideas and findings, and I particularly thank Liam Langley, Alice Trevail, Bethany Clarke, and Stephen Lang for their valuable inputs during the years.

Finally, my biggest thanks go to my husband and closest friend, Brendon Lee, for his unwavering moral support during the many ups and downs, and for his unparalleled assistance and dedication during so many memorable field days. You have made my PhD and life a wonderful adventure.

Chapter 1 – General introduction

1.1. Status of marine biodiversity in the Anthropocene

Marine biodiversity is core to Earth's natural ecosystems, which underpin the healthy functioning of the planet and provide services critical to the health and wellbeing of humans (Talukder et al. 2022). Ocean life is responsible for almost half of global primary production and oxygen generation, provides clean water through nutrient (re)cycling, plays a significant role in controlling our climate through temperature regulation and carbon storage, and provides food and fuel to millions of people (Barbier 2017). Indeed, the ocean is estimated to provide over 3 trillion USD of added value to the global economy by 2030 (OECD 2016). However, losses to ocean biodiversity severely threaten the integrity of marine ecosystems and their services, resulting in their degradation and collapse, and reducing their resilience and adaptability to environmental change (Worm et al. 2006; Talukder et al. 2022).

The Anthropocene – the current geological epoch in which human activity is the dominant influence on climatic and environmental change (Lewis and Maslin 2015) – has put considerable pressure on marine biodiversity, such as in the form of overfishing, pollution, shipping, habitat destruction and fragmentation, the distribution of invasive species, and acceleration of climate change (Talukder et al. 2022). Several studies have shown that a substantial proportion of the ocean area and marine ecosystems face human pressure, with a rapid decline in marine biodiversity richness and abundance (IPBES 2019; Duarte et al. 2020). The Living Planet Index, which synthesises trends in vertebrate populations, shows a 35% reduction in marine species, and a 52% decline in marine vertebrate populations since 1970 (WWF 2015; IPBES 2019). This rapid and alarming biodiversity loss, including in the marine environment, is considered to be one of the most severe environmental issues globally, and is estimated to pose as high a threat to human society as climate change (IPBES 2019).

Maintaining healthy marine ecosystems in the face of accelerated human impacts urgently requires a globally coordinated approach (Satterthwaite et al. 2021). However, important knowledge gaps relating to the status, variability and diversity of marine life, and the nature and extent to which these are impacted, can pose barriers to management priorities and investment choices (Satterthwaite et al. 2021). Scientific research is a key component of the toolkit for evidence-informed management decisions (Ludwig et al. 1993; Dicks et al. 2014). This thesis aims to add to this evidence pool, focusing on the particular issue of seabird-fisheries interactions.

1.2. Seabird ecology and conservation

Pelagic seabirds undergo non-reproductive periods at sea, and return to land to breed at often very large breeding colonies. They are characterised by delayed sexual maturity, low reproductive rates, and long lifespans (Nisbet 1989). Owing to their effective powers of dispersal, seabirds can exploit vast areas of oceans (Ricklefs 1990; Beal et al. 2021a). During the breeding season, seabirds act as central-place foragers, regularly returning to the colony for egg incubation or chick provisioning duties. Many seabirds hold an apex position in their ecosystem, which together with their visibility and accessibility makes them useful indicators of the status and health of marine ecosystems (Cairns 1987; Piatt et al. 2007; Gagne et al. 2018; Church et al. 2019; Velarde et al. 2019). With an overall consumption of biomass of the same order of magnitude as global fisheries landings (Brooke 2004), seabirds play a key role in regulating marine ecosystems (Cury et al. 2011). Seabirds also act as important global drivers of nutrient cycling between terrestrial and marine ecosystems (Otero et al. 2018), thus influencing plant biomass, species composition, abundance and ecosystem productivity (Graham et al. 2018). Seabirds have even been found to play an important role in coupled ecological-chemical processes that result in cooling effects of the climate in the Arctic (Croft et al. 2016). Seabird research and conservation therefore not only serves the target species, but has the ability to make a vital contribution to global initiatives and multi-disciplinary research that aim to improve protection and sustainable management of our oceans and climate (Oppel et al. 2018; Otero et al. 2018).

The ability of migrating predators like seabirds to cross political boundaries and high seas in the absence of a global biodiversity management framework has made them vulnerable to anthropogenic threats (Beal et al. 2021a). Since the late 20th century, many species have undergone dramatic population changes (Lewison et al. 2012; Dias et al. 2019). While some species have increased (e.g. various species of gulls, Croxall et al. 2012), many demonstrated marked declines (Croxall et al. 2012; Lewison et al. 2012). Today, seabirds remain amongst the most globally-threatened of all groups of birds (Phillips et al. 2016; Dias et al. 2019). Based on the International Union for Conservation of Nature (IUCN) Red List criteria, 31% of seabird species are globally threatened (i.e. Critically Endangered, Endangered or Vulnerable), and a further 11% are Near Threatened (IUCN 2018). Although some species are recovering, often as a consequence of long-term targeted conservation efforts (e.g. Amsterdam and short-tailed albatrosses *Diomedea amsterdamensis* and *Phoebastria albatrus,* Croxall et al. 2012; IUCN 2023), almost half of all seabird species (47%) are declining (Dias et al. 2019).

Various factors have contributed to the negative population changes seen in many seabirds. Invasive alien species affect the highest number of seabird species (165 out of 350 species) (Croxall et al. 2012; Dias et al. 2019). Incidental mortality (bycatch) in fisheries affects 100 species but with the greatest average impact, and climate change affects 96 species (Dias et al. 2019). Other important threats include competition for prey through overfishing, direct harvest through hunting and trapping, and disturbance (Dias et al. 2019).

The decline of seabirds continues to be a global conservation concern (Phillips et al. 2016), and has triggered the introduction of national and international policies and conventions that prohibit intentional harm caused to listed species. For example, the multi-lateral Agreement on the Conservation of Albatrosses and Petrels (ACAP) aims to monitor and improve the conservation status of albatross (Diomedeidae) and larger petrel species (Procellaria & Macronectes spp.) – the most threatened of all birds – by tackling key threats such as invasive species and fisheries (Cooper et al. 2006).

1.3. Seabird – fisheries interactions

Global fisheries are amongst the most important drivers affecting seabird populations (Dias et al. 2019). Commercial fisheries have expanded dramatically since the 1960s, both geographically and in their intensity (FAO 2022). In 2020, fisheries and aquaculture production reached an all-time record of 214 million tonnes, worth about USD $424 billion (FAO 2022). Providing up to 50% of animal protein in some countries, and employing almost 60 million people in primary production alone, this industry is of significant importance to global food security and livelihoods (FAO 2022).

While the management of many fisheries globally is improving (Costello and Ovando 2019; FAO 2022), there is compelling evidence that the constant and growing demand for seafood products is negatively affecting marine ecosystems (Jackson et al. 2001; Link and Watson 2019). Fisheries drastically impact marine ecosystems through direct exploitation of target fish, degradation of habitats, and by altering trophic pathways and ecosystem functioning (Pauly et al. 1998; Crowder et al. 2008). To secure ecosystems and their services, fisheries governance is moving from a single-species approach to a more holistic, ecosystem-based management approach, which considers direct and indirect impacts on both target species and the wider ecosystem (Garcia et al. 2003; Pikitch et al. 2004; Crowder et al. 2008; FAO 2022). Top predators such as marine mammals and seabirds are proving to be integral in this management process, because they convey a range of information on the marine environment (Einoder 2009).

As conspicuous and large marine predators and scavengers, seabirds have long been documented to interact with fisheries in all areas of the world's oceans (Tasker et al. 2000; Le Bot et al. 2018). Interactions are both direct and indirect, with implications on their foraging behaviour, distributions and demographics (Votier et al. 2004; Grémillet et al. 2008; Wagner and Boersma 2011).

Fisheries can impact seabirds through competition for the same resource (Skewgar et al. 2007; Cury et al. 2011; Grémillet et al. 2018). Low-trophic level species that often constitute forage prey of many seabirds (e.g. sardines, anchovies, herring, smelt and capelin) account for more than 30% of global fisheries production (Smith et al. 2011), and demand for these species is expected to increase (Cury et al. 2011). This may result in prey scarcity that can affect long-term breeding success and reduce adult survival (Cury et al. 2011; Robinson et al. 2015).

Conversely, fishing operations provide a predictable and abundant food resource for seabirds. Globally, fisheries produce over 10 million tonnes/year of fish waste from unwanted whole fish and processing offal each year (Pérez Roda et al. 2019). Bait and live fish caught in nets present further opportunities for scavenging. At least 52% of seabird species feed on discards to some degree, and fisheries waste is the dominant dietary item in some populations (Bicknell et al. 2013; Oro et al. 2013). Some seabird populations have increased in areas where discards are abundant, e.g. kelp gulls *Larus dominicanus* on the Patagonian Shelf (Yorio and Caille 2004), and great skuas *Stercorarius skua* (Votier et al. 2004; Church et al. 2019) and northern fulmars *Fulmarus glacialis* in the North Sea (Camphuysen and Garthe 1997).

However, in other species, the consumption of discards has been linked to negative population trends, either because discards may be nutritionally inferior to natural prey, or because their consumption implies an underlying problem with natural prey availability (Kitaysky et al. 2006; Grémillet et al. 2008; Navarro et al. 2009; Cianchetti-Benedetti et al. 2018). The demographic and ecological consequences of discard consumption are therefore likely species and system specific, and the long-term consequences are not well understood.

Scavenging at vessels can also lead to gear entanglement and death (Žydelis et al. 2013; Clay et al. 2019). Seabird bycatch is an issue associated with many fisheries around the world, annually accounting for several hundred thousands of seabird mortalities (Clay et al. 2019). Different fishing gear types pose different threats to seabirds (Phillips et al. 2016); for example, in gillnet fisheries, diving birds become entangled in the net while scavenging on the catch. In longline fisheries, the largest threat is from birds being hooked and dragged underwater while attempting to scavenge on bait during line setting. In trawl fisheries, birds are dragged under water and drowned when wings become entangled in trawl cables or net monitoring cables, or when birds become entangled in the net during shooting and hauling operations. Monitoring bycatch and implementing effective mitigation measures has been a focus of seabird conservation research since the 1980s (Weimerskirch and Jouventin 1987; Brothers 1991); however, the issue remains a major cause of adult mortality, thus long-lived and slow reproducing seabirds such as albatross are particularly vulnerable (Croxall et al. 2012; Phillips et al. 2016).

Seabirds are important components of healthy marine ecosystems, and with many vulnerable species interacting with fisheries (Dias et al. 2019), understanding seabird-fisheries interaction is a research priority for seabird conservation and ecosystem-based fisheries management alike (Lewison et al. 2012). However, detailed knowledge of seabird-fisheries interactions remains scarce for many fleets (Calado et al. 2021).

1.4. Study species: The black-browed albatross

This thesis focuses on the effects of fisheries on the ecology and demography of a globally significant population of black-browed albatross *Thalassarche melanophris* (hereafter BBA). The BBA is the world's most numerous and wide-ranging albatross, with an estimated 691,000 breeding pairs distributed across the Southern Ocean (**Figure 1.1**; Phillips et al*.* 2016; BirdLife International 2023). As an abundant top predator that exhibits clear demographic responses to changes in their ecosystems, the BBA is considered a sentinel of the Southern Ocean marine ecosystem (Cherel and Weimerskirch 1995; Ventura et al. 2021a). The BBA therefore forms an important study species of southern marine ecosystem health, threats and changes.

The BBA is long-lived (max. longevity at least 57 years, based on ringing at Bird Island, South Georgia, R. Phillips pers. comm.), reaches sexual-maturity at a median age of 7–8 years (range 6–13), and produces up to one chick a year, which is provisioned by both parents (Prince et al. 1994; Catry et al. 2011; Campioni et al. 2017). Although there is some variation between locations, egg laying generally occurs in austral spring (usually October); chicks mainly hatch in December, and fledge in late April or early May (e.g. Catry et al. 2011). The BBA typically forages over neritic waters of shelf and shelf slope habitats (Cherel et al. 2000; Wakefield et al. 2011; Catry et al. 2013b), where it catches small shoaling fish, crustacean, cephalopods and jellyfish (Reid et al. 1996; Arata and Xavier 2003; McInnes et al.

2017b; Mills et al. 2020). Their limited diving capabilities (max. 19 m, Guilford et al. 2022) restricts them to forage in the upper water column; however, BBA also regularly consume benthic or bentho-pelagic species. These are typically assumed to originate from fisheries (McInnes et al. 2017a), although they may also be obtained through feeding associations with other larger predators such as pinnipeds and whales (Sakamoto et al. 2009). The circumpolar distribution of the BBA means that the species overlaps widely with a large range of fisheries across many political boundaries (BirdLife International 2004; Clay et al. 2019). In the Southwest Atlantic, it is recorded as an abundant scavenger, as well as a common victim of bycatch (Sullivan et al. 2006b; Seco Pon et al. 2015).

In 2003, the BBA was declared *Endangered* due to marked population declines attributed to bycatch in long-line fishing (Favero et al. 2003; BirdLife International 2004). Since then, the BBA has been gradually downgraded to the favourable conservation status of *Least Concern* in 2017 (BirdLife International 2023), linked predominantly to positive population trends at their global stronghold in the Falkland Islands – an archipelago situated in the Southwest Atlantic approximately 500 km from the southern coast of Argentina, and 1200 km from the northern tip of the Antarctic Peninsula (BirdLife International 2018; BirdLife International 2023; **Figure 1.2**; **Table 1.1**). Key drivers behind this positive population trend are assumed to be improved seabird bycatch mitigation, and favourable foraging conditions, including potentially from large amounts of discards made available by trawl fisheries (Wolfaardt 2013; McInnes et al. 2017a). However, the scientific basis for these assumptions is lacking.

Figure 1.1 The black-browed albatross at sea and at its breeding colony on New Island, Falkland Islands, in the Southwest Atlantic. Pictures by A. Kuepfer.

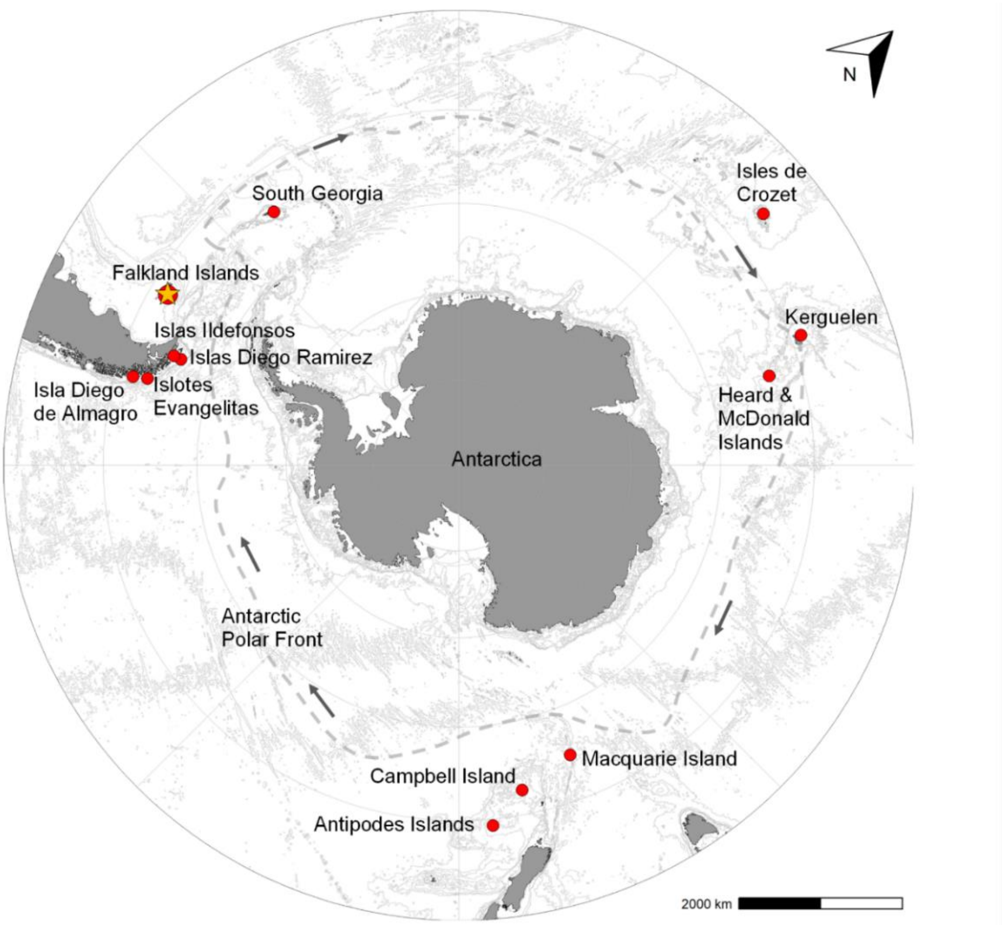


Figure 1.2 Breeding sites of black-browed albatross (red dots). The Falkland Islands, which hold >70% of the breeding population, are denoted with a yellow star. Adapted from ACAP (2010).

Table 1.1 Summary of black-browed albatross population sizes at their breeding colonies. Counts are provided in total bird numbers (N) or breeding pairs (BP). See individual references for more details.

Breeding site	Estimated population	Trend	Reference
Falkland Islands	550,409 (BP)	Increasing	Crofts 2020
Isla Diego Ramírez	17,474 (N)	Increasing	Robertson et al. 2017
Islotes Evangelistas	4,818 (N)	Unknown	Robertson et al. 2017
Isla Diego de Almagro	15,594 (N)	Unknown	Lawton et al. 2003
Islas Ildefonso	57,143 (N) 54,284 (BP)	Decreasing	Robertson et al. 2017
South Georgia	18,298 (BP)	Decreasing	Poncet et al. 2017
Crozet Island Kerguelen Island Heard and McDonald Macquarie Island Antipodes Island Campbell Island	5,800 (BP)	No information	ACAP unpubl. data; BirdLife International 2023

1.5. Study system: Fisheries interactions on the Patagonian Shelf

The BBA breeding in the Falkland Islands forage year-round in the Southwest Atlantic, predominantly on the Patagonian Shelf (Grémillet et al. 2000; Granadeiro et al. 2011; Ponchon et al. 2019), which extends throughout coastal and shelf waters of the Falkland Islands, Argentina, and Uruguay (**Figure 1.3**). The Patagonian Shelf Large Marine Ecosystem is one of the world’s most productive marine ecosystems (Heileman 2009). The region is characterised by highly productive waters, supported by the presence of several oceanic fronts primarily resulting from wind-driven upwelling of cold Antarctic waters. As such, the area is globally important for many large vertebrates including seabirds (Croxall and Wood 2002; Baylis et al. 2019), and sustains regional economies worth ~US$2 billion/year, predominantly from fisheries

(Heileman 2009) (**Table 1.2**). The Patagonian Shelf jigging fleet, targeting Argentine shortfin squid (*Illex argentinus*), represents one of the largest squid fisheries globally (Harte et al. 2019). In addition, diverse trawl fleets target numerous species, predominantly Argentine hake (*Merluccius hubbsi*), hoki (*Macruronus magellanicus*), red-shrimp (*Pleoticus muelleri* – in Argentina only), and Patagonian shortfin squid (*Doryteuthis gahi* – in the Falkland Islands only) (Seco Pon et al. 2015, 2018; Villasante et al. 2015; Falkland Islands Government 2021). A declining demersal long-line fleet catches predominantly Patagonian toothfish *Dissostichus eleginoides* (Seco Pon et al. 2015, 2018; Villasante et al. 2015; Falkland Islands Government 2021).

Seabird-fishery overlap on the Patagonian Shelf, particularly with trawl and demersal longliners, is amongst the highest globally (Clay et al. 2019). The BBA is the most abundant species interacting with these fisheries, where they scavenge on fishery waste but are also at risk of bycatch (González-Zevallos and Yorio 2006; Sullivan et al. 2006b; González-Zevallos et al. 2007; Favero et al. 2011, 2013; Copello et al. 2014; Seco Pon et al. 2015; Tamini et al. 2019; Reid et al. 2021; Jiménez et al. 2022; **Figure 1.4**). However, the nature and extent of interaction varies by fishing gear, season, and BBA life history, and the overall importance of the Patagonian Shelf fisheries in driving the species' ecology and population trends remains poorly understood.

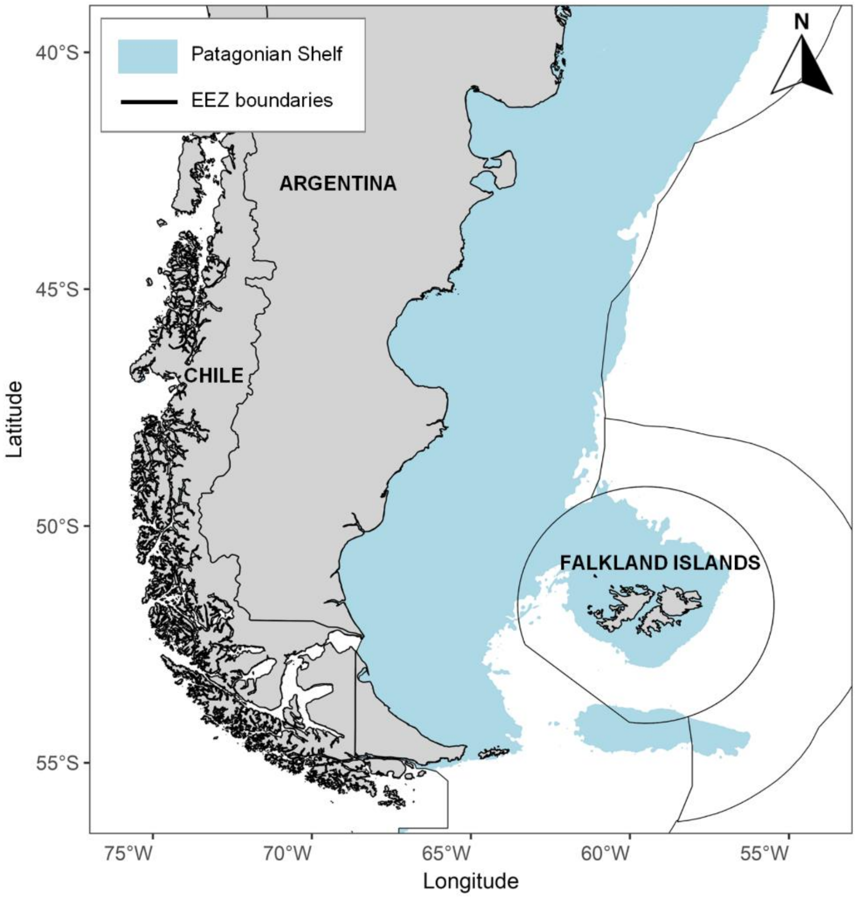


Figure 1.3 The location of the Falkland Islands and the Patagonian Shelf (shown in blue at 200-m isobaths) in the Southwest Atlantic. The Argentine Exclusive Economic Zone and the Falklands Conservation Zones are indicated.

Figure 1.4 Black-browed albatross interacting with trawl fisheries as scavengers, and as victim of bycatch. Pictures by A. Kuepfer.

Table 1.2 Characterisation of commercial fleets operating on the Patagonian Shelf within the Argentine Exclusive Economic Zone and the Falklands Conservation Zones. Fleet sizes are approximate and can vary annually. “Offal” refers to processing waste. “Discards” refers to both offal and waste from non-commercial catch.

Fishery	Main operation location	Dominant target fish	Fleet size	Annual catch (t)	Discard availability & management	Source
Falklands Conservation Zone (2011-2020)				(2011-2020)		
Demersal trawlers	Mainly north, north-west, west and south-west	Demersal finfish, mainly Argentine hake *Merluccius hubbsi*, hoki *Macruronus magellanicus,* Rockcod *Patagonotothen ramsayii*	18	70,000	Offal and bycatch; some vessels use batch discarding since 2018	Falkland Islands Government 2021
Demersal trawlers	North-east, east, south	Patagonian squid *Doryteuthis gahi*	16-17	56,000	Offal and bycatch; Batch discarding since 2018	Falkland Islands Government 2021
Squid jigging	Mainly north and west	*Illex* squid *Illex argentinus*	106	114,000	Minimal offal available. Crushed and discharged without processing	Falkland Islands Government 2021
Demersal longline	All FCZ	Patagonian toothfish *D. eleginoides*	1	1,200	Crushed discards released	Falkland Islands Government 2021
Argentine EEZ						
Ice trawlers	37–48°S	Demersal finfish (mainly Argentine hake) and pelagic fish (Argentine anchovy *Engraulis anchoita* and Chub Mackerel *Scomber japonicus*)	230	1,123,000	No offal discarded, but continuous discharge of all bycatch occurs without processing	Favero et al. 2011; Seco Pon et al. 2015
Bottom freezer trawlers	42°–55°S	Demersal finfish (mainly Argentine hake); Patagonian shrimp *Pleoticus muelleri*	135	250,633 (2000-2012)	Offal and bycatch discards. Equipped with fishmeal plants Discharge occurs without processing	Seco Pon et al. 2015

Demersal longliner	43–55°S	Mainly Patagonian toothfish *D. eleginoides*; Kinclip *G. blacodes*	3 – 4	128,470 (2000-2012)	Fishmeal plants and crushers for non-commercial catch	Favero et al. 2011; Seco Pon et al. 2015
Squid jigging	23–54°S	*Illex* squid	80	104,000	Minimal offal available. Crushed and discharged without processing	Seco Pon et al. 2015; Navarro 2019

1.5.1. Black-browed albatross bycatch

Since the early 2000s, countries responsible for Patagonian Shelf fisheries have been implementing seabird bycatch mitigation approaches (Sullivan et al. 2006a). However, implementation of effective mitigation measures has varied greatly between fleets, and has proven particularly challenging for trawlers (see e.g. Kuepfer et al. 2018; Falkland Islands Government 2021; Tamini et al. 2021).

The positive population trend of BBA in the Falkland Islands suggests that current levels of incidental catches of this species are no longer to the detriment of the local population (statistics summarised by Kuepfer et al. 2018 and Tamini et al. 2021); however, changes in climate or fishing behaviour could alter the additive effects of fishery bycatch (Pardo et al. 2017), and reducing incidental catches of seabirds remains a priority for sustainable fishing practices in all Patagonian Shelf fisheries (Falklands Islands: Kuepfer et al. 2018; Argentina: Consejo Federal Pesquero 2010; Uruguay: Domingo et al. 2015).

Trawl fisheries are the most abundant gear type on the Patagonian Shelf. Discard management is considered the most effective mitigation strategy for trawlers (ACAP 2021), but mitigation effectiveness can vary with differing fleet characteristics and seabird assemblages (Votier et al. 2010; Dias et al. 2019). Therefore, this mitigation measure requires assessment in local fleets.

1.5.2. Black-browed albatross use of discards

It has been argued that a reduction of fishery discards could negatively affect the BBA population (McInnes et al. 2017a). However, there remains much uncertainty about the importance of fishery discards to the BBA, and thus any indirect effects that discard management may have on the population.

Tracking data obtained from BBA foraging trips during the early chick-rearing period from two separate colonies in 2009 and 2011 found little interaction and spatial overlap with fisheries (Granadeiro et al. 2011; Catry et al. 2013b). Birds fitted with geolocation (GLS) loggers during winter and summer showed spatial and temporal overlap of birds and fisheries in the Falkland Islands, but found no definitive hotspot area of overlap (Grémillet et al. 2000). In contrast, satellite-tracked birds captured from fishing vessels off the coast of South America suggested foraging hotspots of non-breeding BBA coincide with fishing activity of fleets in Argentina, Uruguay and Brazil (Copello et al. 2014, 2016).

Dietary studies present similar discrepancies. Studies from the 1990s suggested that the Falkland Islands breeding adult population of BBA obtained approximately 10–15% (Thompson 1992) and 4.4% (Thompson and Riddy 1995) of their annual energy requirement from the Falkland Islands squid and finfish trawl fishery, respectively. Granadeiro et al. (2014) investigated stable isotopes from tissue samples obtained from breeding BBA in 2010 and 2011 and found no evidence of long-term trophic consistency that would indicate birds specialising in discards. Recent dietary data from faeces obtained from breeding birds in the Falkland Islands provided evidence of inter-annual variation in discard use (McInnes et al. 2017a). In contrast, Mariano-Jelicich et al. (2013) found that BBA captured from fishing vessels off the coast of Argentina during the non-breeding period displayed isotopic signatures similar to those of discarded fish species, although results from birds following fishing vessels may be biased should these specialise on fisheries waste.

The importance of discards may vary seasonally and annually (McInnes et al. 2017a), and more research is required to understand more fully how diet variability is influenced by natural foraging conditions and prey availability across years. Furthermore, the potential influence of diet on breeding success has not been assessed for the BBA. In order to reduce the inherent biases associated with individual methods of diet sampling (Barrett et al. 2007), a complementary approach is recommended to allow more robust inference from diet studies (Le Bot et al. 2018). Finally, an improved understanding of regular foraging hotspots, and the extent to which these overlap with fishing efforts, can provide further clarity on the importance of fisheries in driving BBA ecology, including as a proxy for bycatch risk or the importance of discards to BBA.

1.6. Research aims and structure of the thesis

This thesis aims to improve our understanding of the nature and extent to which fisheries influence the ecology and demography of BBA breeding in the Falkland Islands. Specifically, this research focuses on mitigating harmful interactions with fishing gear, and assesses how fisheries affect the diet structure and spatial distribution of birds during breeding.

Chapter 2 presents an experimental study investigating discard management (batch discarding) as a mitigation measure to reduce BBA bycatch in the Falkland Islands trawl fleet.

Chapter 3 uses multi-year BBA chick regurgitates to quantify diet, with a focus on discard consumption. The study further investigates how discard consumption relates to prey availability and breeding success.

Chapter 4 applies complementary dietary assessment techniques of stomach contents and stable isotopes to investigate chick diet from two colonies across two consecutive breeding seasons.

Chapter 5 uses multi-year remote sensing data of seabirds (GPS) and fishing effort (AIS, from Global Fishing Watch) to identify primary BBA foraging areas, quantify broad-scale overlap with fishing activities, and identify areas and fleets where fisheries exposure is highest.

Finally, **Chapter 6** provides a general discussion of the findings, specifically highlighting their implications for conservation management and future research.

Chapter 2 – Strategic discarding reduces seabird numbers and contact rates with trawl fishery gears in the Southwest Atlantic

This chapter has been published as:

Kuepfer A, Sherley RB, Brickle P, Arkhipkin A, Votier SC (2022) Strategic discarding reduces seabird numbers and contact rates with trawl fishery gears in the Southwest Atlantic. Biol Conserv 266:109462. doi: 10.1016/j.biocon.2022.109462

Abstract

Incidental mortality in trawl fisheries is a serious threat to seabird populations. Driven primarily by seabirds attracted to discards, limiting discard discharge through strategic batching is a best practice mitigation measure recommended by the Agreement on the Conservation of Albatrosses and Petrels (ACAP). However, studies supporting the efficacy of batch discarding are rare, limited to the Southwest Pacific, and assess seabird numbers attending vessels only, not gear contact rates. The effectiveness of batch discarding in areas with different seabird communities, fishery assemblages, and natural prey availability is therefore unknown. Here we quantify both seabird numbers and gear contact rates in response to strategic discard discharge in the Falkland Islands trawl fleet for two high-risk species groups: black-browed albatross (*Thalassarche melanophris*) and giant petrels (*Macronectes* spp.). Specifically, we test the effect of three different discharge treatments (zero, batch and continuous discarding) at two vessels. Bird abundance and contact rates were positively related, but zero discarding consistently reduced seabird numbers attending trawlers and eliminated contacts with warp cables and tori-lines. Batching significantly reduced bird abundance and contact rates at the vessel that stored all discards between batches. At the other vessel, however, intermittent release of hashed viscera diminished the mitigation effect. Our findings validate the generality of batch discarding as an effective mitigation measure in trawl fisheries where zero discarding is not possible, while highlighting the importance of complete waste storage.

2.1. Introduction

Death by fisheries bycatch is one of the greatest threats to seabird populations worldwide (Dias et al. 2019), driven primarily by gillnet (~400,000 mortalities year $^{-1}$; Žydelis et al. 2013) and long-line fleets (160–320,000 mortalities year^{-1}; Anderson et al. 2011). However, collision with trawl gears can also lead to significant mortality. For instance, South Atlantic trawl fisheries accidentally kill ~10–34,000 seabirds per annum (Maree et al. 2014; Tamini et al. 2015; Kuepfer et al. 2018; Da Rocha et al. 2021) primarily from collisions with warp cables or via net entanglement (Sullivan et al. 2006b; Kuepfer et al. 2018). Reducing seabird bycatch in trawl fisheries is therefore key for biodiversity and sustainable fisheries.

Seabirds are attracted to trawl fisheries primarily because they generate large quantities of waste (Wienecke and Robertson 2002; Sullivan et al. 2006b; Watkins et al. 2008). These discards can subsidise millions of seabirds (Sherley et al. 2020), benefiting some populations (Church et al. 2019) but can also result in death through entanglement, hooking or colliding with fishing gear (Clay et al. 2019). Managing discards is therefore important for mitigating seabird bycatch.

Strategic discard management may take several forms. A discard ban is the most successful mitigation tool (ACAP 2019a), but this may be unfeasible due to vessel design, processing speed or for political reasons (Bicknell et al. 2013). Storing waste temporarily and releasing it in batches is recommended as the next best mitigation measure for trawlers (ACAP 2019a). Batch discarding reduces seabird numbers attending some New Zealand fisheries (Pierre et al. 2010, 2012a), although direct effect on collision rates has not been assessed. Further, many factors influence seabird-vessel interactions including gear type (Phillips et al. 2016), seabird assemblage composition (Votier et al. 2010; Dias et al. 2019), or environmental factors such as food availability, season, fishing area or weather (Sullivan et al. 2006b; Phillips et al. 2016; Clark et al. 2019). It may therefore be inappropriate to assume that discard management approaches apply across different locations, communities or fleets.

Albatrosses and large petrels (Order Procellariiformes) are particularly vulnerable to trawler mortality. They feed extensively on discards - their size enabling them to swallow large discards and also to dominate scavenging interactions behind vessels. In addition, their long wings have a tendency to wrap around warp cables when struck, resulting in birds being pulled underwater and drowned (Sullivan et al. 2006b; Løkkeborg 2011). Fatal strikes can easily go undetected, however, because corpses are not always retained in the cables, making it difficult to quantify this type of mortality (Parker et al. 2013). Quantifying and mitigating impacts of discard management on large Procellariiformes is a conservation priority.

In the Falkland Islands trawl fleet, annual bycatch averaged over 600 seabirds (range: 174–976) between 2004 and 2018, predominantly of black-browed albatross (*Thalassarche melanophris*) but also various petrel species (Kuepfer et al. 2018). Tori-line entanglements accounted for a maximum of 12.5% of observed seabird mortalities in some years (Kuepfer 2016), highlighting the need for an improved long-term solution to help safeguard or attain a favourable conservation status of internationally protected seabirds. Sullivan et al. (2006b) found that the absence of discards almost eliminated contacts with the warp cables but did not find a significant difference in contact rates of black-browed albatrosses between different levels of discharge. The effectiveness of strategic discard management through batch discarding is therefore unknown in this fleet.

Our aim is to quantify the number and collision rates of two high risk species groups – black-browed albatross and giant petrel species (*Macronectes* spp.) – following trawlers in response to three different fish waste treatments: (1) zero discarding, (2) batch release discarding and (3) continuous discarding. Based on previous studies in New Zealand (Pierre et al. 2010, 2012b), we expect that zero discarding has the ability to eliminate seabird bycatch during trawling, and that batch release, though less effective than zero discarding, will still significantly reduce the risk of bycatch. By focusing on the Falkland Islands trawl fleet we not only test these mitigation methods where a very large fishery co-occurs with ~72% and ~43% of the global populations of black-browed albatross and southern giant petrels (*M. giganteus*, Crofts and Stanworth 2021), respectively, but for the first time also assess the generality of such an approach to fisheries operating outside the south-western Pacific.

2.2. Materials and methods

2.2.1. Study area and experimental set-up

Our study was conducted in the Southwest Atlantic over the Patagonian Shelf and slope, predominantly in the west and north of the Falklands Conservation Zones (FCZs) 48°S–56°S and 52°W–63°W (**Figure 2.1**). These waters are trawled throughout the year by demersal freezer factory vessels targeting a variety of finfish, squid and skate (Falkland Islands Government 2019).

Experimental data were collected aboard two similar-sized trawlers but with differing hold capacities during four commercial finfish fishing trips (**Table 2.1**). For logistical reasons, the two vessels could not be observed simultaneously. Both vessels used obligatory tori-lines during all trawling activities. The factories of the two vessels had been retrofitted with 3 m^3 discard storage tanks designed to receive, store and batch release discards. Once full, the tanks emptied directly into the sea. On Vessel A, a design fault meant that viscera were not collected in the tank. Instead, they passed through two scuppers where they were coarsely cut by a hasher pump, and discharged automatically and intermittently in approximately 2-minute intervals whilst all other waste was stored. This was different from Vessel B where all discards were stored.

Three experimental discarding treatments were implemented during net towing, using a randomised block design (**Table S2.1**):

(1) Continuous – Discards discharged on a continuous/*ad-hoc* basis when available, with tank doors left open and no waste stored.

(2) Batch – Discards temporarily stored before being batch discharged. Once empty, storage resumed. Batch discharges occurred as and when the tanks reached capacity or when factory work was complete. Between batch discharges, filtered factory water continued to be discharged at a continuous rate for practical and safety reasons. At Vessel A, intermittent discharge of hashed viscera continued to occur as well.

(3) Zero – No discards or factory water discharged due to absence of ongoing factory processing during net towing.

Effects of these treatments on seabird-vessel interactions were measured using (a) the abundance of high-risk seabird groups in defined zones at the vessel stern, and (b) contact rates with warp cables and tori-lines. Commercial fishing practices continued throughout the observation periods.

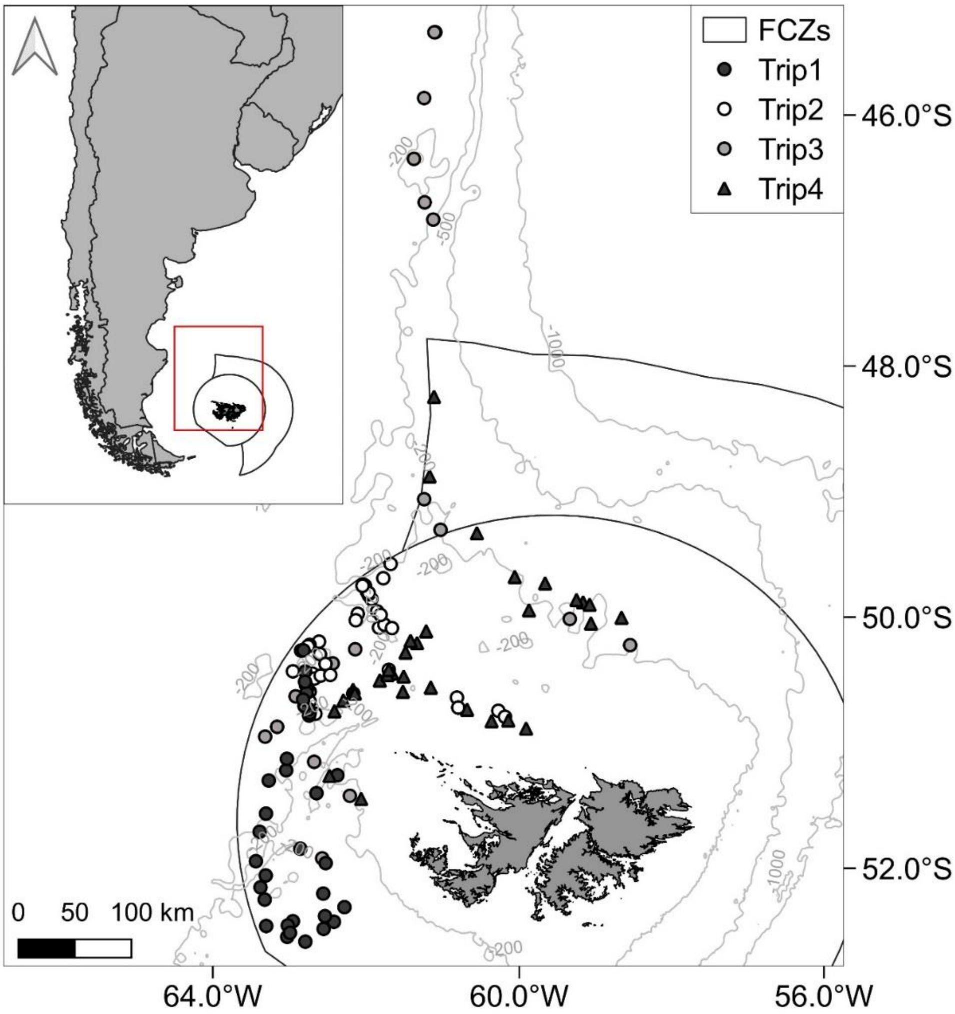


Figure 2.1 Fishing locations east and north of the Falkland Islands over the Patagonian Shelf during the individual experimental trips on Vessel A (Trips 1–3) and Vessel B (Trip 4). FCZs = Falklands Conservation Zones. The 200, 500 and 1000-m depth contours are shown.

Table 2.1 Specifications of study vessels used.

	Vessel A	Vessel B
Length	54 m	53 m
Hold capacity	450 t	720 t
Discard management storage system	3 m^3 tank to temporarily store all discards *except* viscera. Batch release directly out to sea.	3 m^3 tank to temporarily store all discards, *including* viscera. Batch release directly out to sea.
Study dates	Trip 1 (05–22 Apr 2015) Trip 2 (14 May–02 Jun 2015) Trip 3 (29 Oct–10 Nov 2015)	Trip 4 (10–28 Apr 2017)

2.2.2. Data collection

Data were collected by a single observer (AK), who conducted seabird observations from the vessel stern in daylight hours using the naked eye. Use of binoculars was unnecessary given the proximity of the birds to the observer. Although observations were conducted throughout the fishing operation (shoot, trawl, haul), experimental data for the current study were collected during net towing only. For reasons of safety and practicality, observations were not conducted in hours of darkness or at wind speeds exceeding 35 knots.

Environmental and operational parameters

A suite of environmental and operational variables considered relevant to bird interactions (Phillips et al. 2016) was recorded at the start of every observation period: wind speed and sea state (Beaufort scale), wind direction relative to trawling direction, and the number of other vessels operating in the vicinity (as visible by eye). Further, discard level (the volume of discard discharged, based on a subjective assessment of intensity of discharge) and discard rate were recorded to establish compliance with treatment. Observations were combined into sample periods of similar environmental and operational parameters. A new sample period was started whenever one of these parameters changed, or after a maximum of 60 minutes.

Seabird abundance

All seabirds within 500 x 500 m from the vessel stern were recorded at the start of each sample period, and high-risk black-browed albatrosses (hereafter BBA) and giant petrel species (hereafter GP) were allocated a position relative to the stern (modified from Abraham et al. 2009; **Figure S2.1**): (1) 40 m-radius semi-circle of birds on the water; (2) 40 m-radius semi-circle of birds in the air; (3) area between the tori-lines (~ 10 x 30 m) of birds on the water. The latter represents a subset of count (1). Distances were estimated visually, using the 30-m tori-line as a reference point. As data were collected by a single experienced observer, consistency in distance estimation was maximised.

Sweep counts were conducted once at the start of every 10-minute subsample period inside the 40 m areas, and five times during a 10-minute subsample period inside the tori-line area. The latter was implemented only from Trip 2 onwards. The observer spent no more than 30 seconds on each sweep count. Sub-sample periods were always 10 minutes, except when the sample period changed before the completion of a sub-sample period. As such, a sample period contained a maximum of 6 x 10-minute sub-sample periods. Birds take <5 minutes to change their behaviour and abundance when the discard regime changes (A. Kuepfer pers. obs.; Pierre et al. 2010); therefore, samples from the 10-minute counts can be assumed independent. Counts conducted every two minutes were averaged for analytical purpose (see below).

Contact rates

During trawling, visually observed seabird contacts with warp cables and tori-lines were counted and classified as heavy or light based on Sullivan et al. (2006b) (**Table S2.2**), and assigned one of five fates (no apparent harm, minor injury, major injury, death or unknown).

2.2.3. Statistical analysis

All data exploration and statistical analyses were conducted in R (R Core Team 2021). Variables of interest were explored for outliers using Cleveland plots, and the presence of collinearity and correlation of variables assessed using multi-panel scatterplots, Pearson correlation coefficients and variance inflation factors (VIF) (Zuur et al. 2010). To avoid numerical estimation problems and improve interpretation of the parameters, continuous explanatory variables were z-score standardised (Harrison et al. 2018).

A series of models were built using the glmmTMB package (function glmmTMB; Brooks et al. 2017) to determine (a) the effect of discard treatment on seabird abundance and contact rates, and (b) the relationship between contact rate and bird abundance. Count data were modelled using a Poisson error distribution except where over-dispersed, in which case we used a negative binomial error with a log-link function (Harden and Hilbe 2007; Magnusson et al. 2020). For seabird abundance, individual models were built separately for BBA and GP in each of the three count areas behind vessels, using respective abundance counts as the response variables. Contacts models were built for (a) all contacts and (b) heavy contacts individually for BBA and GP, with numbers of contacts used as the response variables, and the natural log of observation duration as the offset (log min) (Zuur et al. 2014).

We generally used sample period (see section 2.2.1) nested in trawl as our random effects with a common slope to account for the fact that bird abundance and contact rates within a sample period and a trawl are not independent. For the tori-line area abundance models, the nested random effects were under-dispersed, thus these were simplified by removing the nested component of the random effect. In all cases, model residuals were checked for autocorrelation (function acf) and there was no evidence of an influence of discard treatment in one trawl on seabird abundance and contact rates at subsequent trawls.

Models assessing discard treatment as the main variable of interest included a range of environmental variables with the potential to influence seabird-vessel interactions (**Table 2.2**). In abundance models, data from the two vessels were combined for analyses, with Vessel_id and the interaction of Vessel_id and Treatment included (**Table 2.2**). In contact models assessing treatment effect, high variance and the absence of certain factor levels at Vessel B meant that the data were analysed separately for the two vessels. However, additional models confirmed that treatment effects on contact rates remained the same at the two vessels when data were analysed jointly, using exclusively Vessel_id, Treatment and their interaction as fixed factors (**Table 2.2**). A stepwise backwards model selection procedure was conducted to determine the final set of covariates for each model, using the lowest Akaike's Information Criterion (AIC) to choose between alternative models.

Where we assessed the relationship between contacts and abundance, data from the two vessels were combined, as exploratory analyses revealed no changes in the overall direction of the relationship between contacts and abundance at the two vessels. The fixed effects in alternative models included exclusively abundance counts in the various sweep count areas (40 m on the water, 40 m in the air, tori-line area) and counts of the 40 m areas combined. The lowest AIC was used to then choose between alternative models.

Model fit was assessed using appropriate diagnostics (Zuur and Ieno 2016) with tools provided by the DHARMa package in R (Hartig 2019), and included assessment of residuals for dispersion, uniformity and zero-inflation (functions testDispersion, simulateResiduals, testZeroInflation). Influential outliers as assessed through residual plots were removed, and whilst this generally improved model fit, it never changed the significance levels of individual parameters. Significance level for all tests was $\alpha = 0.05$.

Table 2.2 Variables used for modelling seabird interactions. AT = Abundance models where Treatment is the main variable of interest; CT = Contacts models where Treatment is the main variable of interest; CA = Contacts models where Abundance is the only variable of interest. Interaction terms included in models are indicated by identical superscript numbers. BBA = Black-browed albatross. GP = Giant petrel species.

Effects	Explanatory variable	Definition	Type	Models
Fixed	Treatment^{1,2}	Experimental discard treatment	Factor	AT, CT
	Vessel id^{1}	Vessel A or Vessel B	Factor	AT
	Wind speed	Wind speed in knots	Continuous	AT
	Sea state	Sea state in Beaufort scale	Continuous	CT
	Season	Chick-rearing, winter and egg-laying	Factor	AT, CT (Vessel A only)
	Relative wind direction	Wind direction relative to trawling direction: 45˚, 90˚, 135˚, astern, into	Factor	AT,CT
	Vessels visible	The number of vessels counted around the experimental vessel	Continuous	AT,CT
	Abundance (BBA / GP)	Bird abundance inside sweep count areas, or combined 40 m areas	Continuous	CA
	Cumulative trawl duration^{2}	Based on sample numbers 1, 2, 3, 4, etc.	Continuous	AT, CT
Random	Trawl	Trawl number	Factor	AT, CT, CA
	Sample (nested in Trawl)	A unique number representing the sample period given a particular trawl	Factor	AT (except tori-line area models), CT, CA
Offset	Log(min)	Duration in minutes of subsample period	Continuous	CT, CA

2.3. Results

2.3.1. Experimental summary

Experimental observations were conducted for a total of 216 hrs, comprising 58 fishing days and 106 experimental trawls (Vessel A: 68; Vessel B: 38) (**Table 2.3**).

The numbers of trawls per discard treatment were 23 trawls during zero discarding, 51 trawls during batch discarding, and 32 trawls during continuous discarding. Mean discard storage time was 33 min (9–120 min) on Vessel A, and 18 minutes (3–42 min) on Vessel B, with batch discharge events taking a mean of 1.1 and 1.6 min, respectively. During batching events, bird abundance inside the count areas changed too quickly to conduct abundance counts, as numbers first increased when discards appeared, and then decreased as the vessel moved forward and the birds remained with the discard patch (**Figure 2.2**).

Table 2.3 Summary of data collected. (d) = days, (t) = trawls, (s) = samples, (c) = counts. Trip 1, 2, and 3 were made on Vessel A; Trip 4 was made on Vessel B.

Treatm.	Trip	All observations				40m abundance data (water & air)			Tori-line area abundance data			Contacts data	
		N(d)	N(t)	N(s)	obs. (hrs)	N(s)	N(c)	obs. (hrs)	N(s)	N(c)	obs. (s)	N(s)	N(c)
Cont.	1	6	10	72	20.83	66	119	19.55	0	0	00.00	55	102
	2	6	9	73	23.12	72	142	22.95	71	141	22.93	67	142
	3	4	4	22	5.88	13	16	2.52	13	16	2.52	13	16
	4	9	9	29	18.97	28	113	13.80	24	97	12.33	28	114
Batch	1	10	17	228	41.65	191	280	38.30	0	0	0.00	206	307
	2	12	16	207	39.12	202	311	37.95	200	312	38.05	195	312
	3	5	5	48	10.95	37	53	6.70	37	53	6.7	36	52
	4	11	13	163	28.77	90	189	21.90	63	126	21.9	160	244
Zero	1	1	1	1	0.93	1	6	0.93	0	0	0	1	6
	2	0	0	0	0.00	0	0	0.00	0	0	0	0	0
	3	6	6	17	9.08	17	55	9.08	16	49	9.08	12	55
	4	14	16	22	17.08	21	105	16.82	21	105	6.82	20	104
Total				882	216.4	738	1389	190.5	445	899	130.3	793	1454

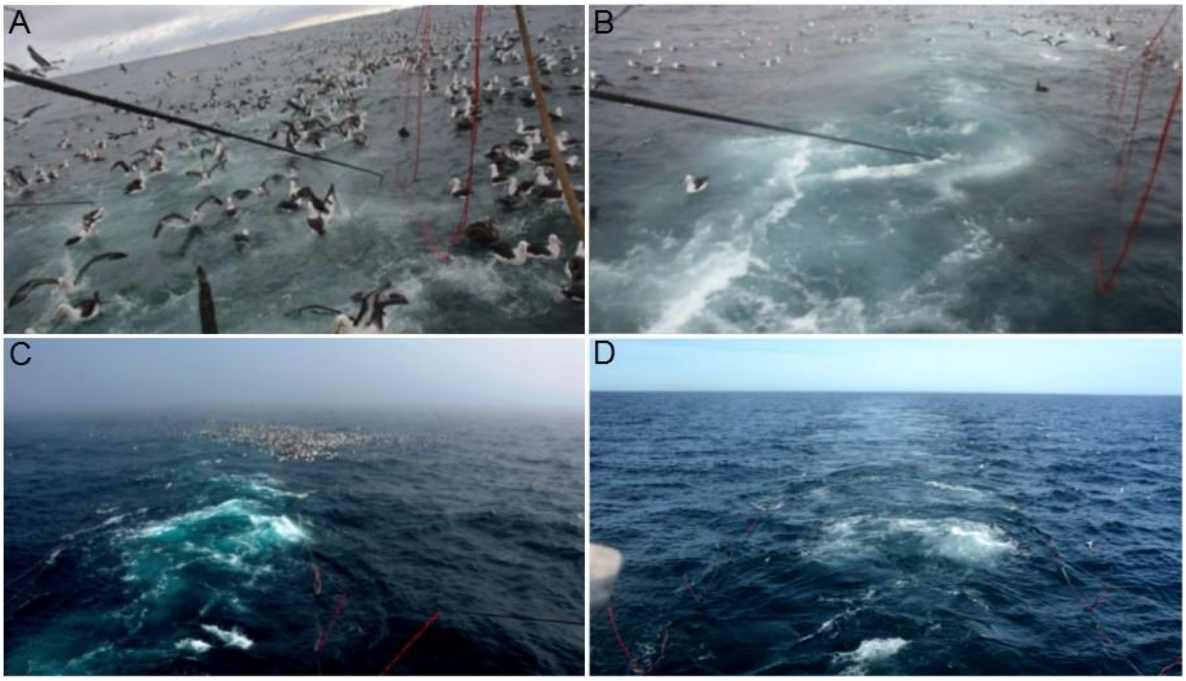


Figure 2.2 Typical view from the vessel stern during continuous discarding (A), discard storage (B), shortly after a batching event (C), and during zero discarding (D). The portside warp cable and tori-line are visible on A–C; both tori-lines are visible in D.

2.3.2. Seabird abundance

During the experiments, 14 seabird species were observed within 500 m of the vessel with BBA most abundant and frequent, followed by Cape petrels (*Daption capense*) and GP (**Table S2.3**). BBA and GP were present on 100% of occasions (BBA: 10 to >500 birds; dominant abundance class = 201–500 birds; GP: 10 to 200 birds; dominant abundance class = 51–200 birds). Other Procellariiformes recorded were royal albatross species (*Diomedea epomophora / sanfordi*), white-chinned petrel (*Procellaria aequinoctialis*), grey-headed albatross (*Thalassarche chrysostoma*), wandering albatross (*Diomedea exulans*), great shearwater (*Ardenna gravis*), sooty shearwater (*Ardenna grisea*), Wilson's storm-petrel (*Oceanites oceanicus*), and southern fulmar (*Fulmarus glacialoides*) (**Table S2.3**).

Impact of discarding on seabird abundance

Relative to continuous discarding, zero discarding significantly reduced abundance of BBA and GP within 40 m of the vessel ($p < 0.05$; **Figure 2.3**; **Table S2.4**), with none inside the tori-line area. Batch discarding reduced the number of BBA in all count areas, and significantly so for birds in the air ($z = -2.64$; $p < 0.008$) and within the tori-line area ($z = -10.02$; $p < 0.001$; **Figure 2.3**). The preferred model did not indicate an effect of vessel identity which might be expected given their difference in discard storage capacities. However, when the data for Vessel A and Vessel B were analysed separately, batch discarding significantly reduced BBA on the water within the 40 m area at Vessel B where all discards were stored ($z = -2.83$; $p = 0.005$), but not at Vessel A where intermittent discarding of viscera continued during storage periods ($z = -0.80$, $p = 0.424$). For GP, the batch treatment also reduced abundance in all count areas, and significantly so for birds on the water (40 m: $z = -2.46$; $p = 0.014$) and within the tori-line area at Vessel B only ($z = -2.43$; $p = 0.015$; **Figure 2.3**).

Other environmental variables that significantly affected bird abundance in at least one of the sweep count areas were wind speed (higher winds increased BBA abundance), relative wind direction (cross winds increased BBA abundance; tail and cross winds increased GP abundance), trawl duration (increased duration decreased BBA abundance), season (increased BBA abundance during winter; increased GP abundance during the egg laying season (October)), and the number of other vessels visible (increased vessel numbers increased BBA abundance) (**Table S2.5**).

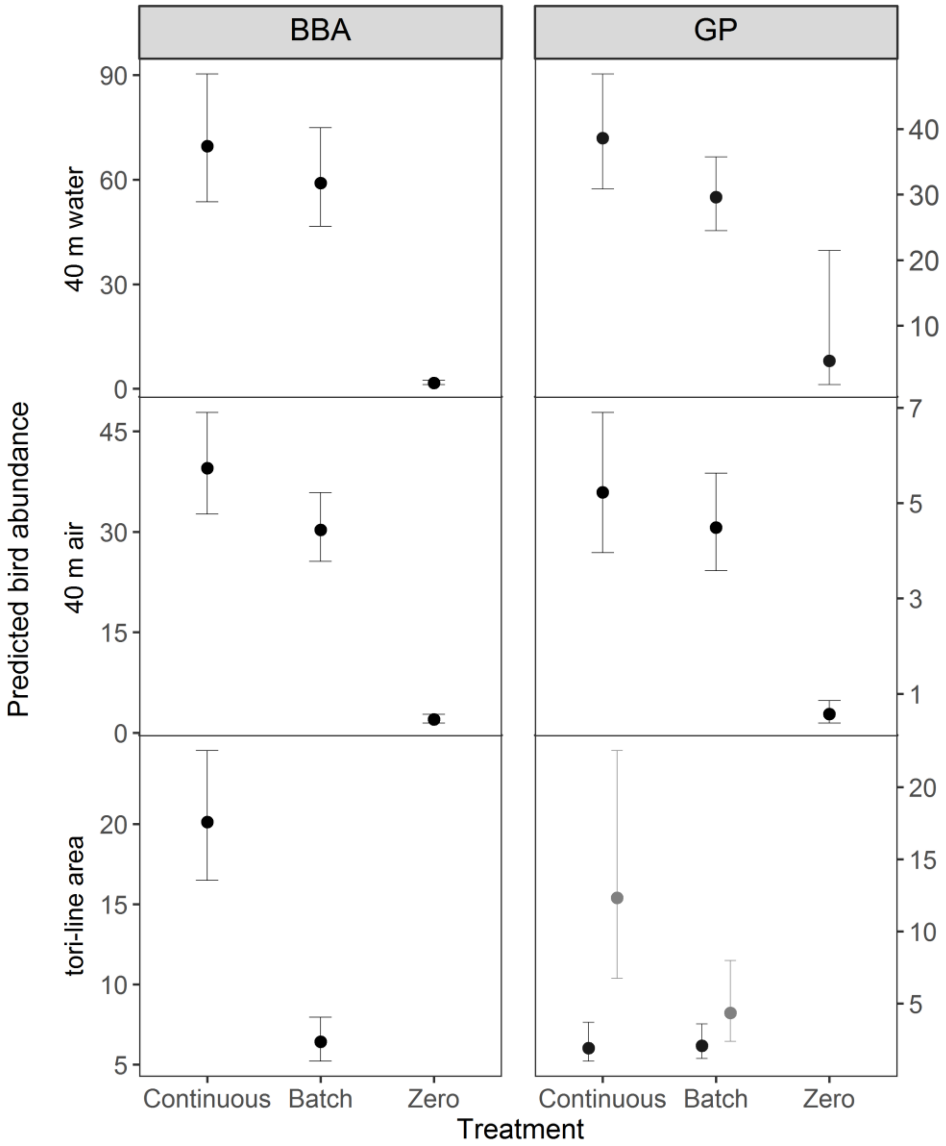


Figure 2.3 Predicted abundance (marginal mean ± 95% CI) of black-browed albatrosses (BBA) and giant petrel species (GP) relative to discard treatments. Bottom right panel shows predictions for Vessel A (black) and Vessel B (grey). Note the difference in scales between panels.

2.3.3. Contact rates

A total of 8,581 contacts by BBA and GP were recorded with warp cables and tori-lines, the majority of which were light, on the water and resulted in no apparent damage (78%; **Table S2.6**). Heavy warp contacts, which have the potential to cause harm, were predominantly incurred by birds on the water (82%). Almost 10% of heavy contacts resulted in harmful or potentially harmful outcomes (death, major injury or unknown fates), although this varied between species (BBA: 11.3%; GP: 7.5%). Sixteen mortalities occurred during experimental observations (15 BBA and one grey-headed albatross), from heavy warp strikes (n = 14) and entanglement with the tori-line (n = 2), as confirmed by observation of carcasses. At least 13 of these mortalities occurred during continuous discarding, whilst two occurred during batch discarding. It is unclear during what treatment the final bird perished, as the trawl contained multiple treatments (Continuous and Batch).

Impact of discarding on seabird collisions

Zero discarding consistently incurred zero contacts (**Table S2.7**), and was therefore not analysed further. Compared to continuous discarding, batch discarding significantly reduced contact rates for both BBA and GP at Vessel B (all $p < 0.001$, **Figure 2.4**). Only heavy contacts of BBA were reduced significantly at Vessel A ($z = -3.02$; $p = 0.003$), although the effect was still significantly stronger at Vessel B where all discards were stored than at Vessel A where viscera continued to be released at an intermittent rate ($z = -2.49$; $p = 0.001$).

During the batch treatment, contact rates of GP and BBA combined declined significantly during storage periods relative to batching events (all contacts: $z = -9.79$; $p < 0.001$; heavy contacts: $z = -3.47$; $p = 0.001$). This difference in contact rates between storage and batching events was significantly greater at Vessel B compared to Vessel A ($p < 0.001$).

Other environmental variables that had a significant effect on contact rates included sea state (heavier sea states increased BBA and GP contacts), relative wind direction (cross and tail winds increased BBA contacts, cross winds increased heavy GP contacts) and season (winter and egg season saw higher overall contacts by both GP and BBA than the chick season), number of vessels visible (fewer heavy contacts of BBA with more vessels), and cumulative trawl duration (fewer contacts of BBA with longer trawl durations) (**Table S2.8**).

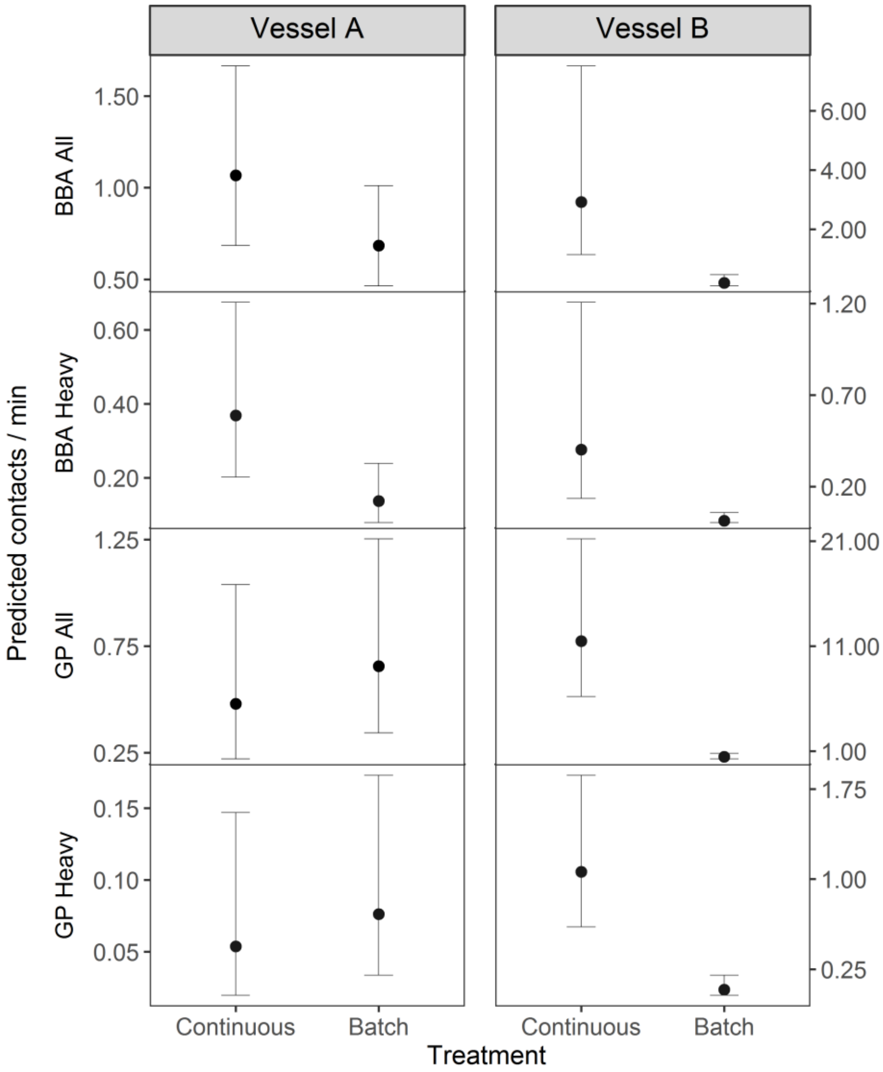


Figure 2.4 Contact rates per min (marginal mean ± 95% CI) for black-browed albatrosses (BBA) and giant petrel species (GP) relative to discard treatments at Vessel A and Vessel B. Note the difference in scales between panels.

Seabird abundance and contact rates

Contact rate was positively correlated with the number of birds attending trawlers for BBA and GP (all minimum: $p < 0.001$; **Figure 2.5**), although this relationship was not significant for GP in the air. Birds inside the tori-line area generally best explained contacts rates, except for heavy contacts of GP, where there was no significant difference between alternative models (**Table 2.4**).

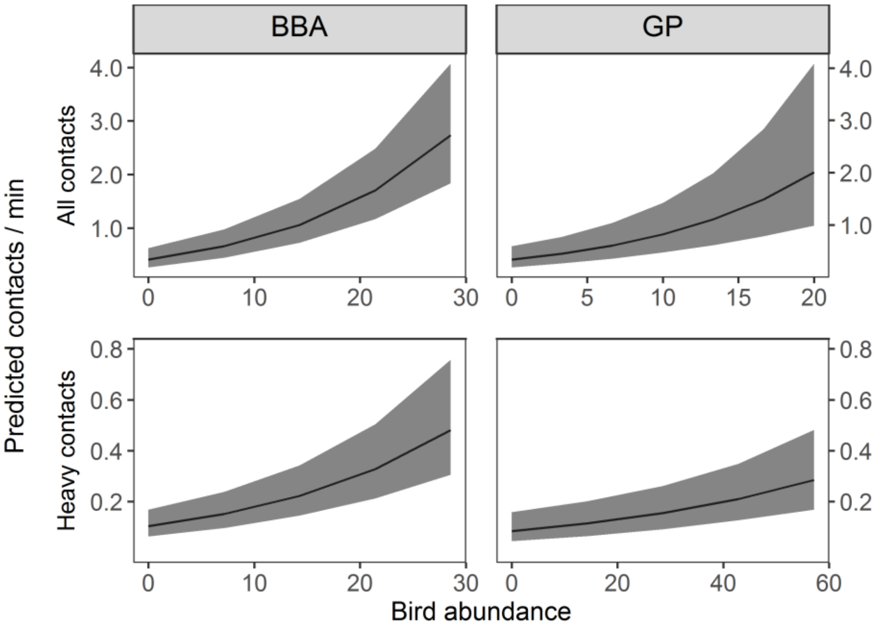


Figure 2.5 Contact rates per min (marginal mean ± 95% CI) in relation to bird abundance of black-browed albatrosses (BBA) and giant petrel species (GP). Best fit models (lowest AIC) shown only, hence bird abundance relates to the tori-line sweep count area in all plots, except for GP heavy contacts (bottom right), where bird abundance relates to the 40 m water sweep count area. The x-axes have been capped to assist visualisation of the relationship at more characteristic abundances of respective sweep count areas. Note the difference in scales between panels.

Table 2.4 Generalised linear mixed model outputs assessing seabird contact rates as a function of seabird abundance. BBA = black-browed albatross; GP = giant petrel species; TL = tori line; AIC = Akaike's Information Criteria. Significance level at α = 0.05.

Response variable	Abundance area	Δ AIC	Estimate	Std. Error	z-value	p-value
BBA all contacts	TL area	0.0	0.948	0.084	11.280	<0.001
	40 m water + air	49.2	1.016	0.125	8.157	<0.001
	40 m air	62.9	0.863	0.122	7.090	<0.001
	40 m water	63.0	0.876	0.122	7.211	<0.001
BBA heavy contacts	TL area	0.0	0.770	0.097	7.952	<0.001
	40 m water + air	14.0	0.893	0.134	6.626	<0.001
	40 m air	20.7	0.803	0.134	5.996	<0.001
	40 m water	23.7	0.749	0.130	5.747	<0.001
GP all contacts	TL area	0.0	0.598	0.113	5.309	<0.001
	40 m water	12.9	0.541	0.130	4.153	<0.001
	40 m water + air	13.5	0.523	0.128	4.080	<0.001
	40 m air	27.3	0.142	0.085	1.664	0.096
GP heavy contacts	40 m water	0.0	0.610	0.134	4.570	<0.001
	TL area	3.4	0.402	0.095	4.229	<0.001
	40 m water + air	6.1	0.486	0.136	3.578	<0.001
	40 m air	16.7	-0.037	0.126	-0.293	0.77

2.4. Discussion

2.4.1. Effect of discard management on seabird-vessel interactions

In the current study, seabird abundance and contact rates were used to test the effect of strategic discard release on seabird-vessel interactions in a Southwest Atlantic trawl fishery. As hypothesised, when discards were absent, there were no birds within the tori-line area, and no contacts with warp cables or tori-lines. In addition, compared to the continuous discharge of discards, batch discarding significantly reduced seabird abundance and contacts. The high interaction rates of BBA is consistent with their dominance at comparable trawl fisheries operating across the wider Patagonian

Shelf (Favero et al. 2011; Seco Pon et al. 2015; Tamini et al. 2015), which in turn suggests our findings should generalise to trawl fisheries operating across the wider Southwest Atlantic.

The extent to which batch discarding reduced seabird interactions varied between the two vessels, being significantly greater at Vessel B where all discards were stored. As simultaneous data collection on Vessel A and Vessel B was not possible, we cannot entirely exclude the possibility of a temporal effect. However, the automatic and intermittent release of hashed viscera during storage at Vessel A provided obvious feeding opportunities thus reducing the effectiveness of batching. Birds are known to increase around vessels when food is present (Pierre et al. 2010, 2012b) and this observation was supported by additional analyses showing an incremental increase in seabird-gear interactions with increasing discard availability (rate and volume) at both vessels (**Table S2.9**). It confirms that reducing the temporal occurrence of discharges is a more effective form of bycatch mitigation for a wider array of seabird species than is discard manipulation such as hashing or mincing (Abraham et al. 2009; Pierre et al. 2010, 2012a).

The significant reductions in seabird-vessel interactions as a result of batch discarding were observed despite relatively short storage periods (average of 18 minutes on Vessel B). This is contrary to findings elsewhere where storage periods of 30 minutes reduced the abundance of small seabirds such as Cape petrels, but not significantly so for larger species such as BBA and GP (Pierre et al. 2012a). This difference may be influenced by operational and environmental factors such as seabird or vessel assemblage, or availability of natural prey. Nonetheless, batch discarding resulted in substantially higher seabird abundance and contact rates compared to the zero-discard treatment, particularly during batching events. Any discards, including factory water, increases bird abundance (Pierre et al. 2010), and the frequent batching events in our study likely contributed to keeping birds closer to vessels. The mitigation potential of discard management can therefore be maximised through prolonged storage periods and by minimising batch-discarding events during trawling activities (Pierre et al. 2010; 2012a,b).

2.4.2. Bird abundance and contact rates

We found that increased bird abundance generally resulted in higher contact rates (**Figure 2.5**). However, this relationship and the extent to which these individual measures of seabird-fisheries interaction were influenced by discard management varied depending on the sweep count area and species considered (**Table 2.4**).

In general, contact rates were most strongly influenced by bird abundance inside the tori-line area where they are closest to warp cables. Within the 40 m count area, the number of GPs in the air had no effect on GP contacts, whilst BBA abundance in the air was as strongly correlated with collisions as was abundance on the water (**Figure 2.5**). This difference may be because, unlike BBA, GP rarely approach discards from the air. Moreover, bird abundance showed a greater response to the batch discard treatment than did contact rates in some instances. Batch discarding reduced BBA abundance inside the tori-line area and in the air, as well as GP abundance on the water (40 m) significantly at both vessels. However, contacts were only significantly reduced at Vessel B for both species. The discrepancy may be the result of other interacting environmental variables influencing abundance and contacts (wind speed, wind direction, season, number of vessels visible, trawl duration), or because contact rates represent a weaker signal.

Our results validate the use of bird abundance as a reliable estimate of collision rates in discard management studies where the latter cannot be collected for reasons of logistics or limited resources. In such events, we emphasise the use of multiple sweep count areas in order to maximise confidence in conclusions drawn about the effectiveness of the discard management system being tested (see also Abraham et al. 2009; Pierre et al. 2010, 2012a,b).

2.4.3. Implementing batch discarding

As well as being reliant on trained and cooperative crew, our study demonstrates that the efficacy of mitigation depends on the discard storage system – i.e. its capacity to store all discards before releasing them systematically in batches. Vessel design differences inhibit a one-size-fits-all solution; close collaboration between scientists, vessel architects and factory crew can help ensure that the deployment of any new equipment can safely and efficiently be built into the processing routine.

Importantly, our study shows that unlike zero discarding, batch-discarding during trawling does not eliminate bird interactions entirely. In addition, discard management cannot completely mitigate against net entanglements that can occur when birds scavenge from the net during shooting and hauling operations. This highlights the benefit of additional mitigation measures during fishing activities, including bird-scaring devices during trawling, effective net cleaning, and efficient deck procedures to minimise the time of shoot and hauling durations (ACAP 2019a).

Finally, while batch discarding does not eliminate this artificial food subsidy for seabirds, it is likely to make it less accessible nonetheless, depending on the sink rate of discards and the competitive abilities of scavenger species. Scavenging seabirds tend to be generalist and opportunist feeders, able to switch prey, so long as alternative prey are available (Bicknell et al. 2013). However, this ability may also be age-dependent, with juvenile and immature birds potentially being affected differently by a discard management policy than more experienced adult birds (Oro et al. 2008; Votier et al. 2008b; Bicknell et al. 2013). Monitoring of natural prey availability as well as dietary and demographic monitoring of affected seabird species will improve our understanding of their dietary flexibility, and potential demographic implications of reduced discard accessibility.

2.4.4. Study limitations

Collecting data aboard commercial vessels often presents logistical challenges that inhibit study design across multiple vessels or seasons (Abraham et al. 2009; Pierre et al. 2010, 2012a; Melvin et al. 2011), but provides a realistic assessment of mitigation measures under operational conditions. More research is required to assess how mitigation is influenced by other factors such as wind, trawl duration and season, probably best achieved via more studies on commercial vessels.

2.4.5. Conclusion

Our study shows that zero-discarding prevented seabird collisions with trawl gears and batch-discarding significantly reduced collisions, particularly when discards were stored effectively between batches. We also found that bird abundance provides a reliable proxy of collision rate, which is important for other studies unable to document bird strikes. Our results provide strong support for discard management as an effective bycatch mitigation tool in the Falklands and demonstrate that batch discarding can significantly reduce bycatch for trawl fisheries where a complete discard ban or prolonged discard storage is unfeasible. Given similar results from New Zealand, this result is likely to apply across a wide range of scenarios, although further research to confirm this is warranted. Differences in discard management designs that limit waste storage may also influence the mitigation potential of batch discarding. Thus, we recommend a re-appraisal of waste management across all fisheries that produce significant amounts of discards.

2.5. Supplementary information

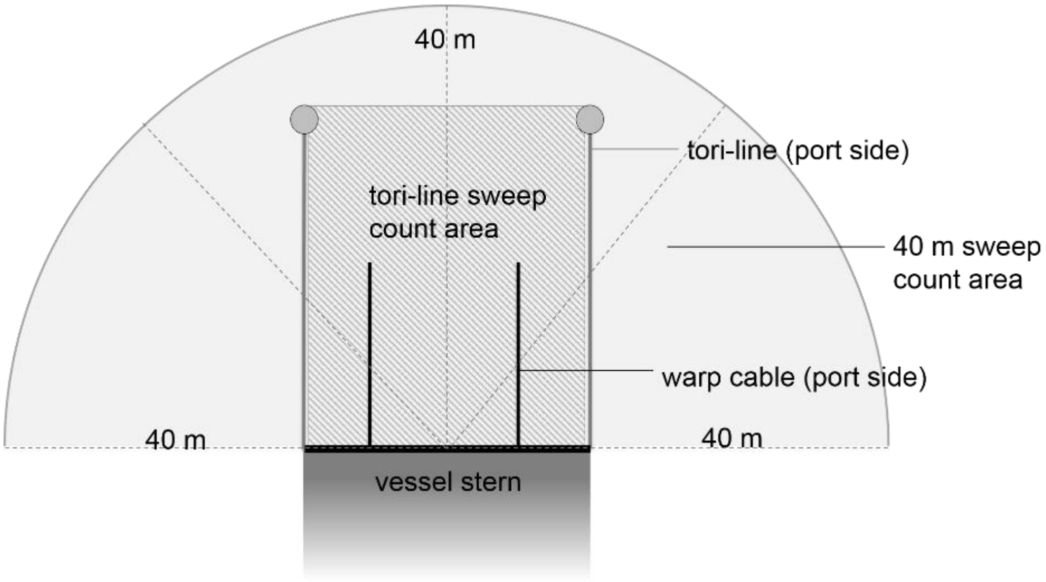


Figure S2.1 Specific areas behind the vessel stern in which black-browed albatross and giant petrel species were counted. Not drawn to scale.

Table S2.1 Sequence of treatments applied and observed across trips 1–4. Original Treatments refers to the original experimental design that included 30-min and 60-min storage periods. However, for practical reasons, it was not possible to adhere to these, and all batch data were eventually pooled.

Trip	Date	Treatments	Original Treatments
1	05/04/2015	Zero, Continuous	Zero, Continuous
1	06/04/2015	Batch	30-min batch
1	07/04/2015	Batch	60-min batch
1	08/04/2015	Batch	30-min batch
1	09/04/2015	Continuous	Continuous
1	10/04/2015	Batch	60-min batch
1	11/04/2015	Batch	60-min batch
1	12/04/2015	Continuous	Continuous
1	13/04/2015	Batch	30-min batch
1	14/04/2015	Batch	60-min batch
1	15/04/2015	Batch	30-min batch
1	16/04/2015	Continuous	Continuous

1	17/04/2015	Batch	30-min batch
1	18/04/2015	Continuous	Continuous
1	19/04/2015	*NA*	*NA*
1	20/04/2015	*NA*	*NA*
1	21/04/2015	Continuous	Continuous
1	22/04/2015	Batch	60-min batch
2	01/06/2015	Continuous	Continuous
2	02/06/2015	Batch	60-min batch
2	14/05/2015	Continuous	Continuous
2	15/05/2015	Batch	30-min batch
2	16/05/2015	Batch	60-min batch
2	17/05/2015	Batch	30-min batch
2	18/05/2015	Continuous	Continuous
2	19/05/2015	Batch	60-min batch
2	20/05/2015	Batch	60-min batch
2	21/05/2015	Continuous	Continuous
2	22/05/2015	*NA*	*NA*
2	23/05/2015	*NA*	*NA*
2	25/05/2015	Batch	30-min batch
2	26/05/2015	Batch	60-min batch
2	27/05/2015	Batch	30-min batch
2	28/05/2015	Continuous	Continuous
2	29/05/2015	Batch	30-min batch
2	30/05/2015	Continuous, Batch	Continuous, 60-min batch
2	31/05/2015	Batch	60-min batch
3	01/11/2015	Batch	30-min batch
3	02/11/2015	Zero, Batch	Zero, 60-min batch
3	05/11/2015	Zero	Zero
3	06/11/2015	Continuous	Continuous
3	07/11/2015	Batch	60-min batch
3	08/11/2015	Zero	Zero
3	09/11/2015	Batch, Continuous	30-min batch
3	10/11/2015	Zero, Continuous	Zero, Continuous
3	29/10/2015	Zero	Zero
3	30/10/2015	Continuous	Continuous
3	31/10/2015	Zero, Batch	Zero, 30-min batch
4	10/04/2017	Zero, Batch	Zero, Batch
4	12/04/2017	Continuous, Zero, Batch	Continuous, Zero, Batch
4	13/04/2017	Batch, Zero	Batch, Zero
4	14/04/2017	Batch, Zero	Batch, Zero
4	15/04/2017	Batch, Zero	Batch, Zero
4	16/04/2017	Continuous, Zero	Continuous, Zero
4	17/04/2017	Batch, Zero	Batch, Zero
4	18/04/2017	Continuous, Batch, Zero	Continuous, Batch, Zero
4	19/04/2017	Continuous, Zero, Batch	Continuous, Zero, Batch

4	20/04/2017	Continuous, Zero	Continuous, Zero
4	21/04/2017	Batch, Continuous, Zero	Batch, Continuous, Zero
4	22/04/2017	Batch, Continuous, Zero	Batch, Continuous, Zero
4	23/04/2017	Continuous, Zero	Continuous, Zero
4	24/04/2017	Continuous, Batch, Zero	Continuous, Batch, Zero

Table S2.2 Seabird-cable /tori-line strike definitions adapted from Sullivan et al. (2006b).

Location	Light strike	Heavy strike
Air	Bird in flight makes contact with the warp cable /tori-line but flies away in a controlled manner.	Bird makes contact with the warp cable /tori-line during flight and is deviated from its natural path or falls into the water.
Water	Bird sitting on the water makes contact with the warp cable /tori-line and, while bird seems unaffected, it may or may not fly away.	Bird sitting on the water makes heavy contact with the warp cable /tori-line and becomes deviated from its natural path, and at least partly submerged.

Table S2.3 Abundance classes for 500 × 500 m counts: 1 (1–10 birds), 2 (11–50 birds), 3 (51–200 birds), 4 (201–500 birds), and 5 (>501 birds). P = present, e.g. when it was too dark to collect abundance data. Modal class (class range), and the percentage of times the species was present during abundance counts are given.

Species	Trip 1 (05/04 - 22/04/2015)	Trip 2 (14/05 - 02/06/2015)	Trip 3 (29/10 - 10/11/2015)	Trip 4 (10/04 -28/04/2017)
Black-browed albatross *Thalassarche melanophris*	5 (1-5), 100%	4 (1-5), 100%	2 (1-5), 100%	5 (1-5), 100%
Giant petrel spp. *Macronectes* spp.	3 (1-5), 100%	2 (1-4), 100%	2 (1-5), 100%	3 (1-5), 100%
Cape petrel *Daption capense*	5 (1-5), 100%	4 (1-5), 100%	2 (1-4), 98%	3 (1-5), 98%
Wilson's storm-petrel *Oceanites oceanicus*	P, 4%	P, 1%	P, 81%	P, 8%
Royal albatross species *Diomedea pomophora/sanfordi*	1 (1), 5%	1 (1-2), 51%	1 (1-2), 71%	2 (1-3), 88%
White-chinned petrel *Procellaria aequinoctialis*	3 (1-4), 68%	3 (1-4), 74%	1 (1-3), 41%	3 (1-3), 84%
Great shearwater *Puffinus gravis*	2 (1-3), 53%	1 (1-2), 28%	1 (1-2), 17%	1 (1), 33%
Kelp gull *Larus dominicanus*	0	1 (1), 24%	1 (1), 6%	0
Wandering albatross *Diomedea exulans*	1 (1), 10%	1 (1), 23%	1 (1), 1%	0
Southern fulmar *Fulmarus glacialoides*	2 (1-3), 59%	P (1-2), 41%	0	1 (1), 11%
Sooty shearwater *Ardenna grisea*	0	0	0	1 (1), 8%
Grey-headed albatross *Thalassarche chrysostoma*	3 (3-4), 71%	P (1), 8%	0	1 (1), 4%
Gentoo penguin *Pygoscelis papua*	0	1(1), 1%	0	0
South American tern *Sterna hirundinacea*	1 (1), 3%	0	1 (1-2), 14%	0
Unidentified giant albatross spp. *Diomedea* spp.	1(1-2), 40%	1(1), 22%	1 (1), 2%	0

Table S2.4 Abundance counts (mean ± standard deviation) of black-browed albatross (BBA) and giant petrel species (GP) in the three count areas during the three discarding treatments.

Vessel	Treatm.	BBA			GP		
		40m water	40m air	tori-line area	40m water	40m air	tori-line area
A	Cont.	97.09 ± 59.3	55.13 ± 31.7	23.89 ± 14.1	28.17 ± 31.3	7.43 ± 13.2	3.75 ± 7.5
	Batch	66.23 ± 48.7	39.14 ± 28.9	12.41 ± 17.8	22.33 ± 22.6	4.64 ± 7.1	2.69 ± 5.9
	Zero	1.20 ± 2.0	2.25 ± 4.1	0.00 ± 0.0	1.18 ± 1.7	1.43 ± 2.0	0.00 ± 0.0
B	Cont.	98.12 ± 47.5	47.57 ± 20.9	24.72 ± 9.7	58.87 ± 26.1	8.67 ± 9.3	15.44 ± 8.6
	Batch	75.68 ± 45.4	37.25 ± 27.0	3.92 ± 4.2	40.62 ± 23.5	6.71 ± 6.7	3.36 ± 3.5
	Zero	2.22 ± 2.8	1.72 ± 1.7	0.00 ± 0.0	0.99 ± 1.7	0.84 ± 1.0	0.00 ± 0.0

Table S2.5 Results of GLMM, with abundance as the selected response variable, and treatment as the primary variable of interest. P = Poisson distribution. NB1 = negative binomial distribution with a log-link function, where the variance increases linearly with the mean as σ 2 = μ(1 + α), with α > 0 (where α is the overdispersion parameter, Hardin and Hilbe 2007; Magnusson et al. 2020). Significance level at *α* = 0.05. BBA = black-browed albatross; GP = giant petrel species.

Model	Effects	Variable	Estimate	Std. Error	z-value	p-value
BBA		(Intercept)	4.03	0.16	25.71	<0.001
(40m water)	Fixed	Ves. id Vessel B	0.30	0.18	1.69	0.092
Model: NB1		Treatment Batch	-0.16	0.12	-1.41	0.159
		Treatment Zero	-3.71	0.22	-16.62	<0.001
		Cum. obs. dur.	-0.08	0.03	-2.94	0.003
		Wind speed	0.14	0.05	2.67	0.008
		Rel. wind 045˚	0.21	0.10	2.12	0.034
		Rel. wind 090˚	0.07	0.11	0.64	0.521
		Rel. wind 135˚	-0.11	0.12	-0.92	0.358
		Rel. wind 180˚	-0.02	0.15	-0.13	0.896
		Season Egg	-0.19	0.24	-0.79	0.427
		Season Winter	0.27	0.17	1.56	0.118
		Vessels visible	0.07	0.04	1.80	0.071
				Var		
	Random	Sample:Trawl	0.41	0.17		
		Trawl	0.55	0.30		
BBA		(Intercept)	3.40	0.12	28.04	<0.001
(40m air)	Fixed	Treatment Batch	-0.27	0.10	-2.64	0.008

Model: NB1		Treatment Zero	-2.97	0.18	-16.58	<0.001
		Wind speed	0.31	0.04	7.21	<0.001
		Rel. wind 045°	0.34	0.09	3.77	<0.001
		Rel. wind 090°	0.21	0.10	2.04	0.041
		Rel. wind 135°	-0.01	0.11	-0.12	0.907
		Rel. wind 180°	0.07	0.13	0.53	0.598
		Season Egg	0.08	0.18	0.46	0.648
		Season Winter	0.37	0.13	2.77	0.006
				Var		
	Random	Sample:Trawl	0.33	0.11		
		Trawl	0.48	0.23		
BBA		(Intercept)	2.76	0.19	14.42	<0.001
(TL area)	Fixed	Treatment Batch	-1.14	0.11	-10.02	<0.001
Model: NB1		Rel. wind 045°	0.48	0.18	2.65	0.008
		Rel. wind 090°	0.58	0.19	3.11	0.002
		Rel. wind 135°	0.20	0.22	0.92	0.358
		Rel. wind 180°	0.51	0.25	2.06	0.039
		Season Egg	-0.42	0.24	-1.79	0.074
		Season Winter	0.10	0.14	0.71	0.478
		Ves visible	0.12	0.05	2.33	0.020
				Var		
	Random	Trawl	0.35	0.35		
GP		(Intercept)	3.65	0.14	25.93	<0.001
(40m water)	Fixed	Ves. id Vessel B	0.47	0.16	2.92	0.004
Model: NB1		Treatment Batch	-0.28	0.11	-2.46	0.014
		Treatment Zero	-2.03	0.84	-2.43	0.015
		Rel. wind 045°	-0.09	0.10	-0.88	0.382
		Rel. wind 090°	-0.22	0.11	-1.95	0.051
		Rel. wind 135°	-0.37	0.13	-2.89	0.004
		Rel. wind 180°	-0.38	0.14	-2.76	0.006
		Treat. Cont.: Cum.obs.dur.	0.23	0.08	2.88	0.004
		Treat Batch: Cum.obs.dur.	0.04	0.03	1.38	0.167
		Treat. Zero: Cum.obs.dur.	1.88	0.96	1.96	0.050
				Var		
	Random	Sample:Trawl	0.29	0.08		
		Trawl	0.59	0.35		
GP		(Intercept)	2.28	0.18	12.79	<0.001
(40m air)	Fixed	Treatment Batch	-0.15	0.15	-1.00	0.320
Model: P		Treatment Zero	-2.21	0.24	-9.26	<0.001
		Wind speed	0.12	0.07	1.82	0.069
		Rel. wind 045°	-0.22	0.15	-1.49	0.136
		Rel. wind 090°	-0.84	0.16	-5.13	<0.001

		Rel. wind 135°	-1.35	0.19	-7.03	<0.001
		Rel. wind 180°	-1.28	0.21	-6.01	<0.001
		Season Egg	0.62	0.23	2.69	0.007
		Season Winter	-0.29	0.19	-1.54	0.122
				Var		
	Random	Sample:Trawl	0.59	0.35		
		Trawl	0.57	0.32		
GP		(Intercept)	0.35	0.40	0.88	0.380
(TL area)	Fixed	Ves. id Vessel B	1.87	0.45	4.12	<0.001
Model: NB1		Treatment Batch	0.07	0.43	0.17	0.863
		Rel. wind 045°	0.53	0.29	1.86	0.063
		Rel. wind 090°	0.06	0.33	0.17	0.866
		Rel. wind 135°	0.33	0.34	0.96	0.338
		Rel. wind 180°	0.55	0.42	1.32	0.187
		Ves. id Ves. B : Treatm. Batch	-1.12	0.46	-2.43	0.015
				Var		
	Random	Trawl	1.05	1.09		

Table S2.6 Summary statistics of seabird contacts with the warp cable and tori-lines during experimental trawls. CP = Cape petrel, GHA = grey-headed albatross, BBA = black-browed albatross, RA = Royal albatross species, SF = southern fulmar, GP = giant petrel species, WCP = white-chinned petrel, GS = great shearwater, SAT = South American tern. TL = tori-line, W = warp, NOA = no apparent damage, PMI = possible minor injury, PMA = possible major injury, D = death, U = unknown.

Spp	Contact Point	Fate	Trip 1				Trip 2				Trip 3				Trip 4			
			Water, light	Water, heavy	Flying, light	Flying, heavy	Water, light	Water, heavy	Flying, light	Flying, heavy	Water, light	Water, heavy	Flying, light	Flying, heavy	Water, light	Water, heavy	Flying, light	Flying, heavy
CP	TL	NOA	1	4	2	4	12			3								
	W	NOA	6	8	1	1	10	8	1	8								
		PMI						1										
GHA	TL	NOA	16	16	7	10	2											
	W	D		1														
		NOA	7	8	9	5												
BBA	TL	D														1		
		NOA	48	23	5	50	182	41	20	121	35	5	6	12		1		
		PMI		1														
	W	D		3				3								4		
		NOA	103	32	47	64	786	105	226	80	116	17	22	18	2332	277	43	23
		PMA		1												3		
		PMI		4						1				1				
		U		9				8				1				79	1	
RA	W	NOA														1		
SF	TL	NOA	2	1		1	4	1	1	1								
	W	NOA					1											
GP	TL	NOA	45	8	1	4	152	4	1	12	288	6	2	2				
	W	NOA	47	18			224	54	2		249	66		4	2146	245	2	1

		PMA														1		
		PMI												1		2		
		U						2				2				30		
WCP	TL	NOA	1	2	1	2	2											
	W	NOA				1			1									
GS	TL	NOA						1										
SAT	W	NOA									1							
Total			276	139	73	142	1375	228	252	226	689	97	30	38	4478	644	46	24

Table S2.7. Contact rates per hour ± standard deviation of black-browed albatross (BBA) and giant petrel species (GP) during the three discard treatments.

Vessel	Treatment	BBA		GP	
		All	Heavy	All	Heavy
A	Continuous	28.98 ± 11.0	8.86 ± 4.1	8.15 ± 4.3	1.80 ± 1.5
	Batch	11.67 ± 3.1	2.71 ± 0.9	10.21 ± 5.8	1.29 ± 0.7
	Zero	0.00 ± 0.0	0.00 ± 0.0	0.00 ± 0.0	0.00 ± 0.0
B	Continuous	121.58 ± 31.6	16.29 ± 4.4	102.39 ± 21	11.39 ± 3.2
	Batch	15.92 ± 4.3	2.75 ± 1.1	16.86 ± 3.9	2.19 ± 0.8
	Zero	0.00 ± 0.0	0.00 ± 0.0	0.00 ± 0.0	0.00 ± 0.0

Table S2.8. Results of GLMM, with contact rate (log contacts/min) as the selected response variable, and treatment as the primary variable of interest. P = Poisson distribution, NB1 = negative binomial distribution with a log-link function, where the variance increases linearly with the mean as $\sigma 2 = \mu(1 + \alpha)$, with $\alpha > 0$ (where α is the overdispersion parameter); NB2 = negative binomial where the variance increases quadratically with the mean as $\sigma 2 = \mu(1 + \mu/\theta)$, with $\theta > 0$ (where θ is the overdispersion parameter, Hardin and Hilbe, 2007; Magnusson et al. 2020). Significance level at *α* = 0.05. BBA = black-browed albatross; GP = giant petrel species.

VESSEL A						
Model	Effects	Variable	Estimate	Std. Error	z-value	p-value
BBA	Fixed	(Intercept)	-3.37	0.36	-9.43	<0.001
(All contacts)		Treatment Batch	-0.44	0.25	-1.77	0.077
Model: NB1		Sea state	0.30	0.12	2.38	0.017
		Rel. wind 045˚	0.36	0.36	0.99	0.320
		Rel. wind 090˚	1.31	0.37	3.58	<0.001
		Rel. wind 135˚	0.93	0.38	2.45	0.014
		Rel. wind 180˚	0.93	0.47	1.99	0.046
		Season Egg	1.30	0.45	2.90	0.004
		Season Winter	1.40	0.26	5.39	<0.001
		Vessels visible	-0.33	0.15	-2.28	0.023
				Var		
	Random	Sample:Trawl	1.22	1.49		
		Trawl	0.45	0.21		
BBA	Fixed	(Intercept)	-4.06	0.45	-8.96	<0.001
(Heavy contacts)		Treatment Batch	-0.99	0.33	-3.02	0.003
Model: NB2		Rel. wind 045˚	0.79	0.48	1.66	0.097
		Rel. wind 090˚	1.76	0.47	3.76	<0.001
		Rel. wind 135˚	1.61	0.49	3.27	0.001
		Rel. wind 180˚	1.81	0.59	3.08	0.002
		Vessels visible	-0.48	0.19	-2.60	0.009
				Var		
	Random	Sample:Trawl	0.98	0.97		
		Trawl	0.76	0.58		
GP	Fixed	(Intercept)	-4.28	0.47	-9.13	<0.001
(All contacts)		Treatment Batch	0.31	0.44	0.72	0.471
Model: NB2		Season Egg	3.92	0.66	5.92	<0.001
		Season Winter	1.04	0.45	2.30	0.022
				Var		
	Random	Sample:Trawl	0.63	0.39		
		Trawl	1.22	1.49		
GP	Fixed	(Intercept)	-5.88	0.61	-9.61	<0.001
(Heavy contacts)		Treatment Batch	0.35	0.51	0.68	0.496
Model: NB1		Season Egg	2.39	0.75	3.21	0.001

		Season Winter	0.79	0.54	1.46	0.145
				Var		
	Random	Sample:Trawl	0.98	0.95		
		Trawl	1.21	1.48		
VESSEL B						
Model	Effects	Variable	Estimate	Std. Error	z-value	p-value
BBA	Fixed	(Intercept)	-1.13	0.49	-2.31	0.021
(All contacts)		Treatment Batch	-2.82	0.49	-5.82	<0.001
Model: NB2		Sea state	0.72	0.33	2.20	0.028
		Cum. obs. dur.	-1.61	0.54	-3.00	0.003
		Vessels visible	0.45	0.28	1.61	0.108
		Treatment Batch: Cum. obs. dur.	1.06	0.60	1.78	0.075
				Var		
	Random	Sample:Trawl	0.89	0.79		
		Trawl	1.37	1.88		
BBA	Fixed	(Intercept)	-3.21	0.59	-5.42	<0.001
(Heavy contacts)		Treatment Batch	-3.41	0.64	-5.30	<0.001
Model: NB2		Sea state	0.79	0.39	2.02	0.044
		Cum. obs. dur.	-1.22	0.42	-2.88	0.004
				Var		
	Random	Sample:Trawl	0.50	0.25		
		Trawl	1.54	2.38		
GP	Fixed	(Intercept)	0.17	0.32	0.54	0.592
(All contacts)		Treatment Batch	-3.24	0.38	-8.62	<0.001
Model: NB2		Sea state	0.45	0.20	2.28	0.022
				Var		
	Random	Sample:Trawl	0.79	0.63		
		Trawl	0.84	0.70		
GP	Fixed	(Intercept)	-3.14	0.83	-3.77	<0.001
(Heavy contacts)		Treatment Batch	-2.58	0.45	-5.69	<0.001
Model: NB1		Rel. wind 045˚	0.46	0.92	0.50	0.619
		Rel. wind 090˚	1.72	0.86	2.00	0.046
		Rel. wind 135˚	2.05	0.89	2.31	0.021
				Var		
	Random	Sample:Trawl	0.16	0.03		
		Trawl	0.52	0.27		
COMBINED VESSEL A + VESSEL B						
Model	Effects	Variable	Estimate	Std. Error	z-value	p-value
BBA	Fixed	(Intercept)	-1.65	0.27	-6.12	<0.001
(All contacts)		Ves. id Vessel B	1.30	0.44	2.92	0.003
Model: NB1		Treatment Batch	-0.48	0.34	-1.42	0.156
		Ves. id Ves. B: Treatm. Batch	-2.00	0.53	-3.74	<0.001
				Var		
	Random	Sample:Trawl	0.90	0.82		
		Trawl	0.98	0.95		

BBA (Heavy contacts) Model: NB2	Fixed	(Intercept)	-3.09	0.34	-9.00	<0.001
		Ves. id Vessel B	0.54	0.57	0.94	0.345
		Treatment Batch	-0.87	0.42	-2.06	0.040
		Ves. id Ves. B: Treatm. Batch	-2.49	0.76	-3.27	0.001
				Var		
	Random	Sample:Trawl	0.54	1.06		
		Trawl	-0.87	1.43		
GP (All contacts) Model: NB1	Fixed	(Intercept)	-2.98	0.34	-8.78	<0.001
		Ves. id Vessel B	3.31	0.50	6.65	<0.001
		Treatment Batch	0.16	0.41	0.39	0.690
		Ves. id Ves. B: Treatm. Batch	-3.06	0.56	-5.47	<0.001
				Var		
	Random	Sample:Trawl	0.67	0.44		
		Trawl	1.19	1.41		
GP (Heavy contacts) Model: NB2	Fixed	(Intercept)	-5.31	0.53	-9.96	<0.001
		Ves. id Vessel B	3.27	0.74	4.44	<0.001
		Treatment Batch	0.59	0.61	0.97	0.330
		Ves. id Ves. B: Treatm. Batch	-4.59	0.96	-4.79	<0.001
				Var		
	Random	Sample:Trawl	0.52	0.27		
		Trawl	1.59	2.53		

Table S2.9 Results of GLMM, with the selected response variable being abundance (a) and contact rate (log contacts/min) (b), and discard level and discard rate individually used as the primary variable of interest. Data were combined for black-browed albatross (BBA) and giant-petrel species (GP). NB1 = negative binomial distribution with a log-link function, where the variance increases linearly with the mean as $\sigma 2 = \mu(1 + \alpha)$, with $\alpha > 0$ (where α is the overdispersion parameter; Hardin and Hilbe, 2007; Magnusson et al. 2020).

Abundance						
VESSEL A						
Model	Effects	Variable	Estimate	Std. Error	z -value	Pr(>|z|)
BBA + GP	Fixed	(Intercept)	4.380	0.276	15.87	<0.001
(40m air + water)		Disc_level Medium	-0.098	0.278	-0.35	0.730
Model: NB1		Disc_level Low	-0.221	0.277	-0.80	0.420
		Disc_level Negligible	-0.495	0.292	-1.70	0.090
		Disc_level Nil	-2.473	0.319	-7.76	<0.001
				Var		
	Random	Sample:Trawl	0.24	0.06		
		Trawl	0.20	0.04		
			Estimate	Std. Error	z -value	Pr(>|z|)
BBA + GP	Fixed	(Intercept)	4.392	0.157	28.06	<0.001
(40m air + water)		Disc_rate Batch	-0.263	0.257	-1.02	0.306
Model: NB1		Disc_rate Intermittent	-0.227	0.166	-1.37	0.171
		Disc_rate Infrequent	-0.545	0.204	-2.67	0.008
		Disc_rate None	-2.496	0.231	-10.80	<0.001
				Var		
	Random	Sample:Trawl	0.24	0.06		
		Trawl	0.23	0.05		
VESSEL B						
Model	Effects	Variable	Estimate	Std. Error	z -value	Pr(>|z|)
BBA + GP	Fixed	(Intercept)	5.354	0.107	49.80	<0.001
(40m air + water)		Disc_level Medium	-0.241	0.119	-2.00	0.043
Model: NB1		Disc_level Low	-0.151	0.095	-1.60	0.112
		Disc_level Negligible	-0.388	0.090	-4.30	<0.001
		Disc_level Nil	-3.594	0.169	-21.20	<0.001
				Var		
	Random	Sample:Trawl	0.21	0.04		
		Trawl	0.37	0.14		
	Fixed	(Intercept)	5.159	0.105	49.20	<0.001
BBA + GP		Disc_rate Batch	-0.059	0.117	-0.50	0.615
(40m air + water)		Disc_rate Intermittent	-0.406	0.095	-4.30	<0.001

Model: NB1		Disc_rate Infrequent	-0.808	0.241	-3.30	<0.001
		Disc_rate None	-3.415	0.177	-19.30	<0.001
				Var		
	Random	Sample:Trawl	0.19	0.04		
		Trawl	0.43	0.19		
Contacts						
Vessel A						
Model	Effects	Variable	Estimate	Std. Error	z value	Pr(>|z|)
BBA + GP	Fixed	(Intercept)	0.007	0.162	0.040	0.970
(All)		Disc_level Medium	-1.271	0.132	-9.630	<0.001
Model: NB1		Disc_level Low	-2.210	0.160	-13.820	<0.001
		Disc_level Negligible	-3.895	0.437	-8.920	<0.001
				Var		
	Random	Sample:Trawl	0.51	0.26		
		Trawl	0.92	0.84		
			Estimate	Std. Error	z -value	Pr(>|z|)
BBA + GP	Fixed	(Intercept)	-0.389	0.212	-1.83	0.067
(All)		Disc_rate Batch	0.204	0.264	0.77	0.439
Model: NB1		Disc_rate Intermit.	-1.638	0.223	-7.33	<0.001
		Disc_rate Infrequent	-3.460	0.593	-5.84	<0.001
				Var		
	Random	Sample:Trawl	0.62	0.39		
		Trawl	1.01	1.03		
Vessel B						
Model	Effects	Variable	Estimate	Std. Error	z -value	Pr(>|z|)
BBA + GP	Fixed	(Intercept)	1.453	0.201	7.21	<0.001
(All)		Disc_level Medium	-1.169	0.28	-4.18	<0.001
Model: NB1		Disc_level Low	-2.614	0.271	-9.64	<0.001
		Disc_level Negligible	-3.448	0.199	-17.33	<0.001
				Var		
	Random	Sample:Trawl	0.42	0.18		
		Trawl	0.73	0.53		
BBA + GP	Fixed	(Intercept)	-0.677	0.292	-2.32	0.021
(All)		Disc_rate Batch	2.772	0.189	14.64	<0.001
Model: NB1		Disc_rate Intermit.	-1.027	0.452	-2.27	0.023
		Disc_rate Infrequent	-1.450	1.332	-1.09	0.277
				Var		
	Random	Sample:Trawl	0.76	0.58		
		Trawl	1.11	1.23		

Chapter 3 – Prey-switching to fishery discards does not compensate for poor natural foraging conditions in breeding albatross

This chapter has been published as:

Kuepfer A, Votier SC, Sherley RB, Ventura F, Matias R, Anderson O, Brickle P, Arkhipkin A, Catry P (2022) Prey-switching to fishery discards does not compensate for poor natural foraging conditions in breeding albatross. ICES J Mar Sci. fsac069. doi: 10.1093/icesjms/fsac069

Abstract

Fishery discards supplement food for many seabirds, but the impacts of declining discards are poorly understood. Discards may be beneficial for some populations but have negative impacts by increasing bycatch risk or because they are junk-food. The Falkland Islands support >70% of global black-browed albatross *Thalassarche melanophris* populations, which feed on discards. However, the effect of discards on population demographics, and implications of fishery management changes, are unknown. We analysed stomach contents of black-browed albatross chicks across eight breeding seasons (2004–2020) from New Island, Falkland Islands, to assess variation in discard consumption and how this relates to foraging conditions and breeding success. Across years, 68–98% of samples contained natural prey, whilst 23–88% of samples contained fishery discards. Discard consumption was positively related to fishery catches of hoki *Macruronus magellanicus* and sea surface

temperature anomalies (SSTA) (˚C), and negatively related to breeding success. These results suggest a diet-switching behaviour for Falkland Islands albatrosses, whereby birds switch from preferred natural prey to suboptimal discards when environmental conditions, and hence natural feeding opportunities, are unfavourable. Crucially, this study suggests that fishery discards do not compensate for poor natural foraging conditions for breeding albatrosses in the long term.

3.1. Introduction

Assessing fisheries impacts on marine predators is a key objective in sustainable marine management. Fisheries can negatively impact seabirds via incidental mortality (bycatch) in fishing gear, changes in the community structure, and because they compete for the same stocks (Votier et al. 2004; Wagner and Boersma 2011; Grémillet et al. 2018). Conversely, fisheries discards – unwanted fish and processing offal – can provide a predictable and plentiful food resource for scavenging seabirds (Real et al. 2018; Sherley et al. 2020). Therefore, discarding has implications for marine food-webs, as well as for the ecology and demography of scavenging birds (Bicknell et al. 2013).

Discards can support millions of scavenging seabirds (Sherley et al. 2020) and some seabird populations have increased in areas where discards are abundant, e.g. kelp gulls *Larus dominicanus* on the Patagonian Shelf (Yorio and Caille 2004) and great skuas *Stercorarius skua* in the North Sea (Votier et al. 2004). Accordingly, global changes in catches (Branch et al. 2011; FAO 2020) and discard bans in the European Union, Norway, Chile, and New Zealand may have knock-on effects on discard-dependent species (Votier et al. 2004; Bicknell et al. 2013; Sherley et al. 2020; but see

Le Bot et al. 2019). However, studies assessing this issue tend to focus on European waters (Bicknell et al. 2013; Oro et al. 2013; Le Bot et al. 2019).

Albatrosses regularly follow fishing vessels (Phillips et al. 2016) and, while much research has focussed on how this impacts seabird bycatch risk, little is known about the demographic consequences of discard provision. The black-browed albatross *Thalassarche melanophris* has a southern circumpolar distribution and frequently interacts with fisheries where it scavenges on discards and suffers high bycatch (reviewed in Phillips et al. (2016)).

In the Falkland Islands, where over 70% of its global population breeds (Wolfaardt 2013), discard consumption appears particularly prevalent compared to populations elsewhere (DNA-based study, McInnes et al. 2017a). Indeed, discard consumption has been suggested as a contributing factor to increasing black-browed albatross populations in the Falkland Islands (Thompson and Riddy 1995; Croxall and Gales 1998; McInnes et al. 2017a), which contrasts the declines seen at most other colonies (BirdLife International 2018). Nevertheless, studies testing a direct link between discard consumption and population change in the Falkland Islands are lacking.

Black-browed albatross breeding in the Falkland Islands forage over the Patagonian Shelf throughout the year (Ponchon et al. 2019). During winter, there is evidence that in the northern part of the wintering range they feed extensively on discards, according to gut contents of beached birds and victims of fishery bycatch off southern Brazil (Colabuono and Vooren 2007) and stable isotope analysis from intentionally-captured live birds that foraged near vessels off northern Argentina (Mariano-Jelicich et al. 2013, 2017). However, it is unclear whether birds sampled as a result of interactions with fisheries are representative of the wider population. Breeding adults and chicks in the

Falkland Islands appear to feed predominantly on natural prey (Thompson 1992 (stomach content analysis); Granadeiro et al. 2014 (stable isotope analysis); McInnes et al. 2017b (DNA-based analysis)), and while some tracked individuals associate with fishing vessels at sea, they do not seem to specialise on fisheries' waste (Granadeiro et al. 2014). However, discards during the breeding season may be more important in some years (McInnes et al. 2017a), and more research is therefore required to develop a fuller understanding of how discard feeding varies annually in relation to natural foraging conditions and prey availability. Furthermore, none of the previous studies conducted in the Falkland Islands assessed the relationship between diet and breeding success (but see elsewhere, e.g. Le Bot et al. 2019).

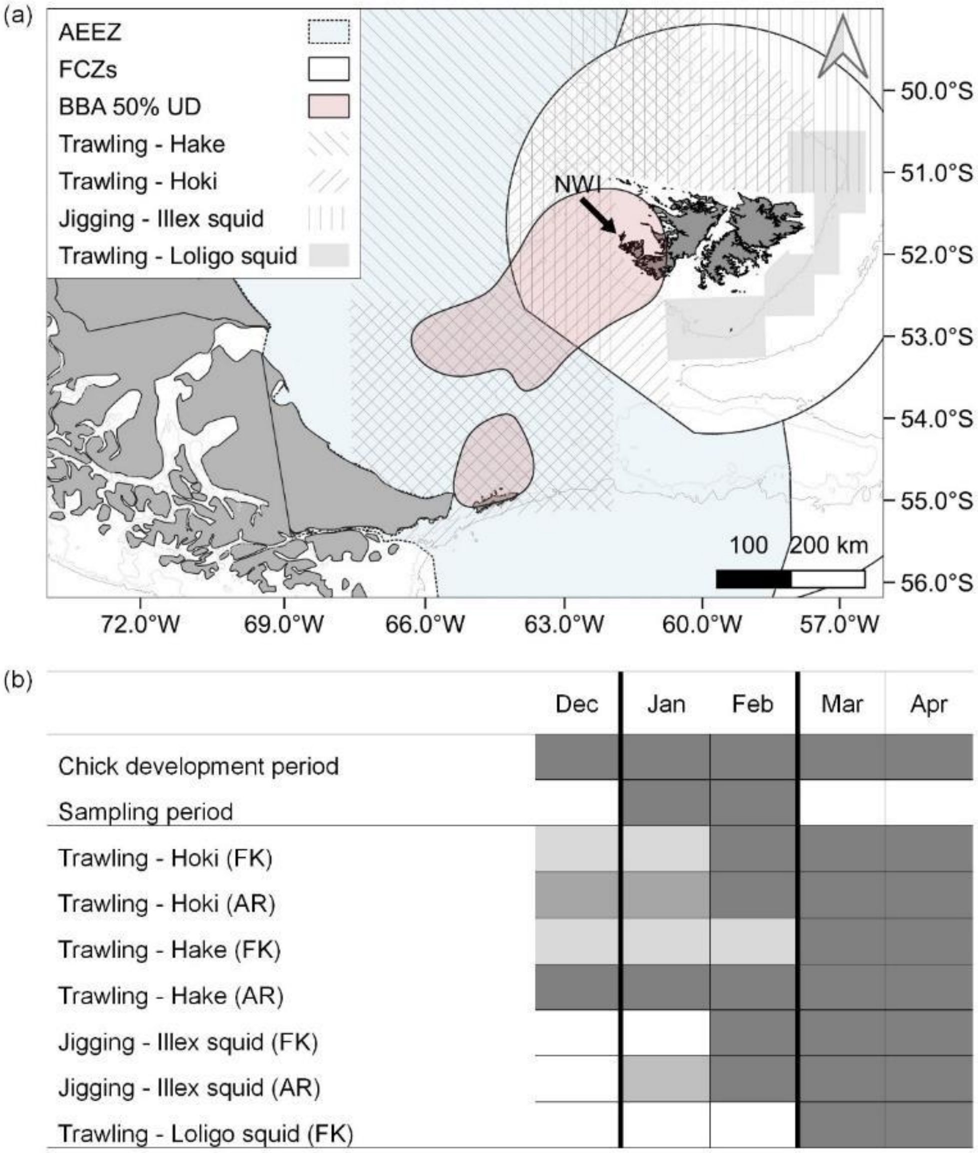
Due to a discard management reform in the Falkland Islands trawl fleet, whereby discards are to be stored temporarily before batch discharging (Kuepfer and Barton 2018; Kuepfer et al. 2022b), an improved understanding of discard use by black-browed albatross is especially pressing. In addition, the black-browed albatross is a sentinel of the Patagonian Shelf Large Marine Ecosystem (Ventura et al. 2021a), and an indicator species used by the Commission for the Conservation of Marine Living Resources (CCAMLR) Ecosystem Monitoring Program (CEMP). Therefore, dietary information will not only help to inform black-browed albatross conservation, but also promote a more complete understanding of fishery impacts in the Southern Ocean marine ecosystem, and further afield (Bestley et al. 2020).

The current study uses multi-annual chick diet and breeding performance data to further our understanding of the importance of discards in supporting the Falkland Islands black-browed albatross population. Our objectives are threefold: (1) quantify chick diet and prey origin (natural vs discards); (2) assess inter-annual diet variability in relation to discard and natural prey availability; and (3) determine the nature of the relationship between discard use and breeding success. In light of available information, we hypothesise that black-browed albatross prefer natural prey for chick provisioning, but that they switch to discards depending on availability and natural foraging conditions. Specifically, we would expect a diet switch to discards when natural foraging conditions are poor, and reproductive performance is stressed.

3.2. Materials and methods

3.2.1. Study area and fleet characteristics

Chick diet was sampled at New Island, Falkland Islands (51˚ 43'S, 61˚ 18'W, **Figure 3.1**), which supports approximately 15,500 breeding pairs (Wolfaardt 2013). Breeding adults concentrate their foraging south-west of New Island over the southern Patagonian Shelf within the Falklands Conservation Zones (FCZs) and the Argentine Exclusive Economic Zone (EEZ) (Ventura et al. 2021a, **Figure 3.1**). The Patagonian Shelf is one of the most productive marine ecosystems in the world, supporting high biodiversity and important fisheries (Baylis et al. 2021), notably demersal finfish freezer trawlers targeting predominantly Argentine hake *Merluccius hubbsi*, hoki *Macruronus magellanicus* and rock cod *Patagonotothen ramsayi* (average annual landings of finfish trawlers 2011–2020: Argentina 155,230 t; Falkland Islands 70,166 t; Ministerio de Agricultura, Ganadería y Pesca Argentina (https://www.magyp.gob.ar/sitio/areas/pesca_maritima/desembarques; Falkland Islands Government 2020; **Figure 3.1**). Finfish fisheries discard bycatch of non-commercial or undersized fish, and also dump offal (head and guts) of commercial fish at sea. Patagonian long-finned squid (*Doryteuthis gahi,* hereafter Loligo), also sometimes caught as bycatch in the finfish fishery, is not discarded. Trawling for Loligo occurs to the east of Falkland Islands, outside the key foraging areas of New Island breeding birds, and outside our sampling period (late February–May; August–October). Finally, between February and May, the Patagonian Shelf also hosts one of the world's largest cephalopod fisheries (Arkhipkin et al. 2015; Harte et al. 2019), jigging for Argentine shortfin squid (*Illex argentinus*, hereafter *Illex*; average annual landings 2011–2020: Argentina 104,166 t; Falkland Islands 114,050 t – Ministerio de Agricultura, Ganadería y Pesca Argentina https://www.magyp.gob.ar/sitio/areas/pesca_maritima/desembarques/; Falkland Islands Government 2020). Discards from these vessels are limited, as the squid is generally packed whole (Laptikhovsky et al. 2006; Arkhipkin et al. 2015).



	Dec	Jan	Feb	Mar	Apr
Chick development period	■	■	■	■	■
Sampling period		■	■		
Trawling - Hoki (FK)	░	░	■	■	■
Trawling - Hoki (AR)	▒	▒	■	■	■
Trawling - Hake (FK)	░	░	░	■	■
Trawling - Hake (AR)	■	■	■	■	■
Jigging - Illex squid (FK)			■	■	■
Jigging - Illex squid (AR)		▒	■	■	■
Trawling - Loligo squid (FK)				■	■

Figure 3.1 (**a**): New Island (NWI) in western Falkland Islands, and its proximity to the South American continent. The 50% Utilisation Distribution (UD) of breeding black-browed albatross tracked from New Island (Ventura et al. 2021a) overlaps with key fisheries on the Patagonian Shelf inside the Falklands Conservation Zones (FCZs) and the Argentine Exclusive Economic Zone (AEEZ). Fisheries areas inside AEEZ are approximate and highlight only areas of key catches (adapted from Irusta et al. 2016; Morsan et al. 2017). The 200-m and 1,000-m isobaths are shown. (**b**): Temporal overlap of chick period, sampling period, darker shading indicating increased catches (relative). FK = Falkland Islands; AR = Argentina.

3.2.2. Sampling

Diet samples were collected during mid-chick rearing (late January and February) across eight seasons between 2003/04 and 2019/20 (hereafter 2004 to 2020) via induced regurgitation. Freshly fed chicks were briefly inverted over a bucket with the stomach and throat gently massaged until the stomach felt empty or regurgitation stopped. The procedure took less than a minute and chicks were never sampled more than once. This procedure has no significant effect on survival or albatross fledging mass (Phillips 2006). Individual diet samples were weighed whole (total mass, g), and again after draining off the stomach oils (drained mass, g).

3.2.3. Diet composition

Prey identification and grouping

Prey items were first separated into one of six dietary categories: (1) fish, (2) cephalopods, (3) crustaceans, (4) gelatinous zooplankton, (5) carrion, and (6) non-food (internal parasites, plant matter, pebbles and plastics), and the drained mass of these groups was recorded. Fish, crustaceans and cephalopod were further identified to the highest possible resolution using reference collections (Falkland Islands Fisheries Department (FIFD) unpubl. data; Xavier and Cherel 2009) and through consultation with local specialists (B. Lee, Z. Shcherbich (FIFD)). The minimum number of individuals was determined through counts of whole animals and fresh loose hard structures (fish = bones, otoliths; cephalopods = squid gladii, beaks (with wings attached), statoliths; crustaceans = carapaces with both eyes still attached to minimise the risk of counting items originating from secondary ingestion). Otoliths and cephalopod beaks were paired based on left and right (otolith), lower and upper (beak), size, and level of erosion. Eroded and accumulated squid beaks and heavily eroded/brittle otoliths were assumed to have originated from previous meals (Furness et al. 1984; van Heezik and Seddon 1989), and were excluded from all analyses. Fresh prey of fish or cephalopods that could not be identified to species level were only included in analyses involving the main prey groups of fish and cephalopods.

Individual prey items identified to species level were then further categorised as either fisheries discard or natural prey (**Table S3.1**). We assumed prey was natural if it met one of the following criteria: (1) the species inhabits depths (including during diel migration) accessible to black-browed albatrosses (~19 m, Guilford et al. 2022) and is not caught in fisheries in the Falkland Islands or the southern Patagonian Shelf; (2) the species is a commercially fished benthopelagic or pelagic fish potentially accessible to black-browed albatrosses, but smaller than the minimum size caught in the Falkland Islands trawl fishery (Falkland Islands Government 2020 and previous annual reports); (3) the fishing period, discard policy or processing makes the species very unlikely to be available from fisheries. The latter is the case for *Illex* and Loligo squid, which could feasibly be obtained both as discards and as natural prey, regardless of size (Prince 1980; Arkhipkin et al. 2013a, 2015). However, we believe that the majority of these squid found in stomach samples were caught naturally because: (1) discards of Loligo and *Illex* are negligible as they are generally packed whole (98% and 96%, respectively in Falkland Islands waters), and discarding Loligo in Falkland Islands waters is illegal (Laptikhovsky et al. 2006; Arkhipkin et al. 2015); (2) whilst squid can be taken directly from the trawl net (A. Kuepfer pers. obs.), the main Loligo fishery operates outside the diet sampling period (**Figure 3.1b**; Falkland Islands Government 2020). Conversely, we assumed that prey originated from discards if at least one of the following three criteria was met: (1) large fish heads were present in the absence of the bodies; (2) the species is a demersal fish of any size that inhabits depths beyond the dive depth of black-browed albatrosses (>19 m, Guilford et al. 2022); or (3) the species is a commercially fished benthopelagic or pelagic fish, which, depending on its age, is potentially accessible to black-browed albatrosses, but is within the size and age class caught in the Falkland Islands demersal-trawl fishery (Falkland Islands Government 2020 and previous annual reports). The latter corresponded specifically to hoki and southern blue whiting (*Micromesistius australis*). Size classes of these species were reconstructed from intact otoliths (maximum sagittal length (mm)) using a microscope equipped with a graticule and by applying species-specific allometric formulae (**Table S3.2**).

3.2.4. Annual indices of prey availability

To estimate discard availability, we used fishery catch data from January and February of each study year from the Argentine EEZ (Sanchez et al. 2012; Navarro et al. 2014; Navarro 2019; Ministerio de Agricultura, Ganadería y Pesca Argentina (https://www.magyp.gob.ar/sitio/areas/pesca_maritima/desembarques/) and the FCZs (Falkland Islands Government 2020 and previous annual reports). The amount of discard available (particularly from processing offal) was assumed to be directly proportional to catches.

Data on natural prey abundance during study years were unavailable, with the exception of just three years of Loligo biomass (Ramos and Winter 2020). Therefore, we used two remotely sensed proxies of foraging conditions: net primary production (NPP, mg/m^2) and sea surface temperature anomalies (SSTA, °C). Primary production can influence prey availability to seabirds through bottom-up processes (Frederiksen et al. 2006; Chassot et al. 2010; Anguita and Simeone 2015); sea surface temperature has been associated with feeding conditions and prey abundance, as it can affect primary and secondary productivity (Behrenfeld et al. 2006; Barbraud et al. 2012). SSTA, in particular, has previously been found to affect demographic aspects of black-browed albatross and other seabirds in the Falkland Islands (e.g. breeding success, breeding probability (Ventura et al. 2021a), breeding pair divorce rates (Ventura et al. 2021b); chick provisioning rates, and chick growth (Quillfeldt et al. 2007, 2010)). Monthly NPP and SSTA were calculated for January and February in the core foraging area (50% UD) of breeding birds tracked between 2008 and 2019 during early breeding (October–December; Ventura et al. 2021a, **Figure 3.1**). The NPP data, produced by Copernicus Marine Environment Monitoring Services were downloaded at a spatial resolution of 0.04° (https://resources.marine.copernicus.eu/?option=com_csw&view=details&product_id=OCEANCOLOUR_GLO_CHL_L4_REP_OBSERVATIONS_009_082). SSTA data were downloaded from NOAA at a spatial resolution of 0.01° (https://coastwatch.pfeg.noaa.gov/erddap/griddap/jplMURSST41anommday.html).

3.2.5. Breeding success

Since 2003/04, 170 to 259 nests have been followed daily from egg-laying through to hatching, and subsequently weekly until chicks reached 60 days of age, at which point they are considered to have survived to fledging (Catry et al. 2010). We define breeding success as the percentage of nests where egg laying occurred that contained a live chick after 60 days.

3.2.6. Statistical analysis

All data analyses were conducted in R version 3.5.3 (R Core Team 2021). Statistical significance was set at $\alpha = 0.05$.

Diet composition

To assess whether the number of sampled chicks was sufficient to describe diet diversity, we randomized annual samples 100 times, and constructed species accumulation curves as a function of sample size (the number of stomach samples collected in a given year – package vegan::accumcomp; (Oksanen et al. 2019). Sample sizes would be considered sufficient to describe the full diversity of the diet if the fitted cumulative prey curves reached an asymptote (**Figure S3.1**).

As samples were all collected within a specific time period (mid-chick rearing), samples from January and February were pooled. Prey were quantified using a range of metrics (**Table 3.1**): (1) percentage drained mass (M%); (2) percentage frequency of occurrence (FO%); (3) percentage numeric frequency (N%); and (4) percentage index of relative importance (IRI%) (Barrett et al. 2007), where:

M%; the percentage of total drained sample mass represented by a prey group;

FO%; the percentage of stomach samples in which a particular prey was present;

N%; the number of individuals of a particular prey present as a percentage of all individual fish and cephalopod species counted;

$\text{IRI\%} = (\text{IRI}/\Sigma\text{IRI}) \times 100$, where $\text{IRI} = \text{FO\%} \times (\text{N\%} + \text{M\%})$.

These different metrics of prey quantification provide complementary dietary information, including variability of prey abundance (FO%), frequency of prey encountered (N%), or approximate nutritional contribution (M%; IRI%) (Duffy and Jackson 1986; Barrett et al. 2007). For practical reasons, however, and because individual crustaceans were not consistently counted across years, it was not possible to obtain all metrics for all taxonomic groups and levels (**Table 3.1**). For example, it was not possible to calculate N% (and hence IRI%) for crustaceans, and M% (and hence IRI%) could not be obtained for individual prey species (and therefore prey origin), because it was impractical to separate out and identify every individual fish bone or flesh remains. Also, we did not attempt to back-calculate the original biomass of individual fish species, because, particularly with discarded fish, it is not always clear whether the chick was fed the entire fish, or just a part of it.

Table 3.1 Metrics used to quantify prey at different taxonomic levels. For practical reasons, it was not possible to obtain M% of individual species. Counts of individual crustaceans were not consistently collected across years. M% = percentage drained mass; FO% = percentage frequency of occurrence; N% = percentage numeric frequency; IRI% = percentage index of relative importance (IRI = FO% × (N% + M%)).

	M%	FO%	N%	IRI%
Main prey groups including…				
Fish	✓	✓	✓	✓
Cephalopod	✓	✓	✓	✓
Crustacean	✓	✓		
Others	✓	✓		
Species of…				
Fish		✓	✓	
Cephalopod		✓	✓	
Crustacean		✓		
Origin				
Discard		✓		
Natural		✓		

Inter-annual variation in diet

Following explorative ordination techniques using non-metric multi-dimensional scaling (nMDS), and testing for equal variance in multivariate dispersion, we used a permutational multivariate analysis of variance (PERMAONVA) (package vegan::adonis, Oksanen et al. 2019) based on the Bray-Curtis similarity index to explore inter-annual trends in diet composition of key prey groups (presence/absence of fish, cephalopods and crustaceans), and prey origin (presence/absence of fisheries discards and natural prey). An analysis of similarity percentage (SIMPER) was employed to calculate the contribution of each prey species (%) to the diet differences observed in the PERMANOVAs (Clarke 1993).

Relationship between discard consumption and prey availability

Annual trends in discard consumption were assessed in relation to commercial fishery catches and environmental indices of natural prey availability using linear regression models on logit-transformed dietary proportions. Our response variables were FO% and N% of hoki and banded whiptail (*Coelorinchus fasciatus*, hereafter grenadier), FO% of fishery discards, and IRI% of fish. IRI% of fish was used due to the high percentage of fish that was categorised as discards (see results). Hoki and grenadier were used because these species represented the most numerous and frequently occurring discard species (see results). For hoki, analyses were based exclusively on specimens categorised as discards. Due to small sample sizes (and to avoid model over-parameterisation), each model was run separately with one of eight alternative explanatory variables. These were hoki catches in (1) Argentina (January), (2) Argentina (February), (3) Falklands (January), and (4) Falklands (February), as well as (5) SSTA (January), (6) SSTA (February), (7) NPP (January), and (8) NPP (February). The decision not to combine January and February values of catch data and of environmental data was because over 70% of samples originated from February. We would therefore assume February prey conditions to be more influential, especially in the case of catch data where no temporal lag would be expected. The decision not to combine Argentine and Falklands catches was based on the changes in commercial species distribution between Argentine and Falkland Islands waters during this period (e.g. Winter and Ramos 2020). In exploratory analyses, we also tested various lags for environmental variables (October–December), but as there was no relationship, we restricted candidate variables to January and February. The candidate models (including a null model) for the same response variable were compared using the Akaike's Information Criterion corrected for small sample size AICc, with lower AICc indicating a better model fit. A model was deemed to have predictive power if the ΔAICc was ≥2 compared to the null model. The significance level was adjusted using the Bonferroni correction with the alpha value of 0.05 divided by the number of models with predictive power to minimise Type 1 error. Model validation involved inspection of residuals for outliers and patterns, and goodness-of-fit was determined using the R-squared value. We did not perform similar analyses to assess trends in natural prey consumption, because we have reasons to believe that our data for this group are underestimated (see results and discussion).

Relationship between discard consumption and breeding success

In order to assess the relationship between breeding success and discard consumption, we fitted generalised linear mixed models (GLMM) in glmmTMB (function glmmTMB, Brooks et al. 2017). We used a binomial distribution (1 = chick fledged or 0 = chick failed) with a logit-link function as our response variable and fitted separate models with each of the following explanatory variables (1) FO% of hoki or grenadier, (2) N% of hoki or grenadier, (3) FO% of fishery discards, and (4) IRI% of fish. All analyses of hoki again only included specimens categorised as discards. We included "year" as a random effect to account for repeat diet measures within each sampling year. This is particularly relevant as we used "year" to account for good and resource poor years, different yearly natural prey availability and discard availability. The AICc was used for model selection, with individual models assessed against the null model. Model fit was evaluated using numerous visual and numerical assessments typically conducted for binary data, including inspection of residuals for normality, patterns and outliers using the DHARMA package (Hartig 2019), and performance of the Hosmer-Lemeshow test (Hosmer et al. 2013). The R-squared values were calculated after Nakagawa and Schielzeth (2013).

3.3. Results

3.3.1. Diet composition

We extracted 2,620 prey items from 282 chick regurgitates during eight seasons between 2004 and 2020 (2004, 2007–2011, 2019, 2020). Overall, 99% of prey items belonged to either fish [85% (FO); 57% (M); 83% (IRI)], crustacean [75% (FO); 26% (M)] or cephalopod [39% (FO); 12% (M); 17% (IRI)] (**Figure 3.2a**). This includes fresh prey that were identified as one of these prey groups, but which could not be identified to any higher taxonomic resolution. Gelatinous zooplankton (medusae and salps) were recorded in four seasons, as well as small numbers of bivalves, gastropods and carrion (penguin feathers).

The asymptotes of species accumulation curves were not fully reached; however, the flattening of the curves suggests that the most important detectable prey had been identified (**Figure S3.1**). Twenty-two taxa were identified (fish: 5 orders, 7 families; 11 species; cephalopod: 2 orders, 5 families, 5 species; crustacean: 4 orders, 6 species; **Table S3.3**). The most frequent species of crustacean were lobster krill *Munida gregaria* [67% (FO)] and *Themisto gaudichaudii* [9% (FO)], both categorised as natural prey. The most important cephalopod was Loligo [20% (FO); 18% (N)], followed by *Illex* [(3% (FO); 2% (N)] and *Moroteuthopsis ingens* [(3% (FO); 1% (N)], also all categorised as natural prey. Fish species were generally dominated by prey categorised as discards: pelagic hoki [29% (FO); 16% (N)], demersal grenadier [23% (FO); 11% (N)], and benthopelagic rock cod *Patagonotothen ramsayi,* [19% (FO); 16%

(N)] (**Table S3.3**). The exception was 2019, when pelagic Fuegian sprat (*Sprattus fuegensis*), categorised as natural prey, was the most frequent and abundant fish species [(35% (FO); 30% (N)] (**Table S3.3**).

Across the years, naturally obtained prey were found in 88.0% of samples [annual range: 67.5–97.5% (FO)] (**Figure 3.2b**). Eighty percent of identified fish were categorised as fishery discards (but see discussion on possible underestimation of natural fish prey such as Fuegian sprat). As such, fishery discards were present in 50% of diet samples across years [annual range: 22.6–87.5% (FO); **Figure 3.2b**]. All discard species corresponded with target or bycatch species of trawl fisheries operating on the Patagonian Shelf (**Table S3.3**). Ninety percent (n = 204) of measurable hoki and 100% (n = 200) of grenadier were categorised as discards, making them the most abundant and frequently present discard species identified.

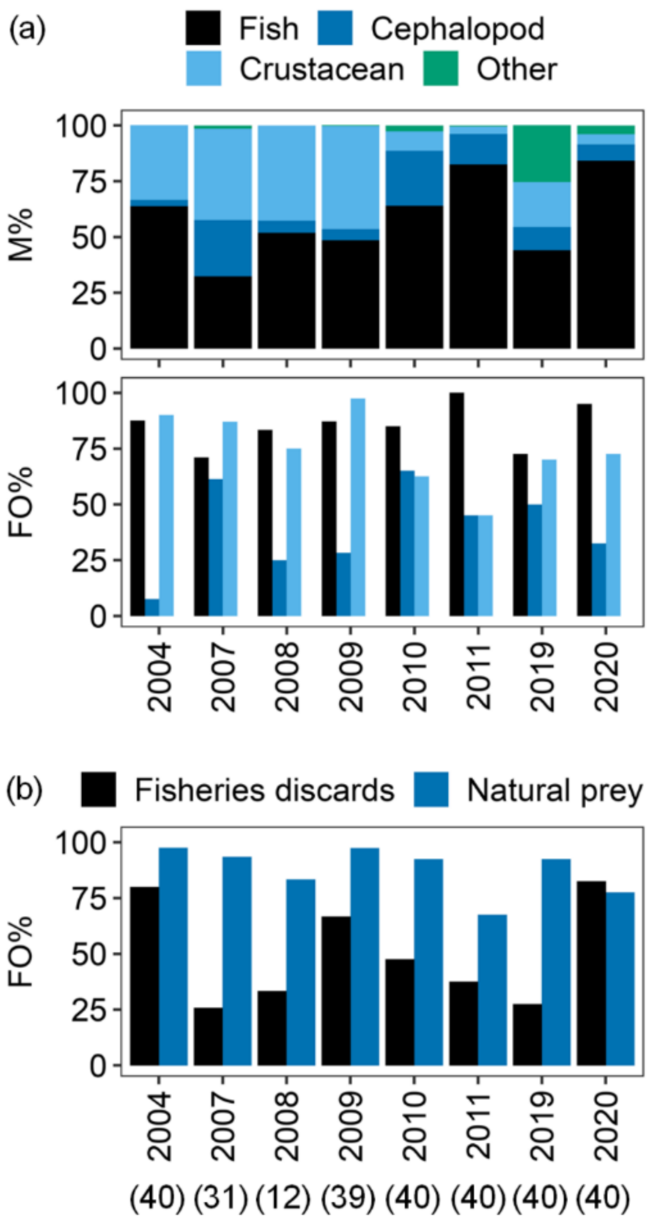


Figure 3.2 Prey found in stomach samples of black-browed albatross chicks at New Island, Falkland Islands, between 2004 and 2020, quantified in (**a**) percentage drained mass (M%) and percentage frequency of occurrence (FO%) of key prey groups; (**b**) percentage frequency of occurrence (FO%) of prey origin. Sample sizes are denoted in brackets underneath sampling year.

3.3.2. Inter-annual variation in diet

Diet composition differed among years in terms of FO% and M% of key prey groups (fish, crustaceans, cephalopods: PERMANOVA, FO%: $R^2 = 0.205$, $P = 0.01$; M%: $R^2 = 0.212$, $P = 0.01$), and FO% of prey origin (fisheries discards and natural prey, PERMANOVA, $R^2 = 0.163$, $P = 0.01$). The SIMPER analysis revealed that, on average, lobster krill*,* hoki, Loligo, rock cod, grenadier, *T. gaudichaudii* and Fuegian sprat contributed 50% to the observed inter-annual differences in diet composition (**Table S3.4**). Lobster krill and hoki were the only two species found in the diet every year, although between 2008 and 2011, rock cod was the most prevalent fish species (**Table S3.3**).

3.3.3. Discard consumption and prey availability

We found a significant positive relationship between consumption of hoki (FO% and N%) and SSTA (February), as well as between consumption of fish (IRI%) and SSTA (February) (**Table 3.2; Figure 3.3**). The trends between grenadier consumption and fishery catches or SSTA were not significant, unless one outlier (year = 2010) was removed (Linear regression, SSTA (February) FO%: $P = 0.005$; N%: 0.003). Consumption of fisheries discards (FO%) was also positively influenced by February hoki catches in the Falkland Islands (**Table 3.2**; **Figure 3.3**). Based on the R-squared values, between 57 and 83% of the variability in the data are explained by the models (**Table 3.2**). Inspection of residuals found no issues of homogeneity, however, outliers were identified for all response variables apart from fish (IRI%). Removal of outliers increased statistical significance of results for hoki (N%) and grenadier (FO% and N%); however, we retained all data points in order to maximise our small dataset. No relationship was found between SSTA (February) and hoki catches (February), which could confound our results (**Table S3.5**). Further, no effect was found between diet and NPP in January or February.

Table 3.2 Linear model outputs and AICc values for model comparison. ΔAICc is the difference in AICc compared to the null model. Coefficient estimates (Est.) and standard errors (Std. Err.) are on the logit scale. Hoki (F) indicates that only specimens categorised as fisheries discards were considered in the analysis. We only include models that showed higher predictive power compared to the null model (ΔAICc ≥ 2).

Response variable	Fixed effect	Est.	Std. Err.	z-value	p-value	Adjusted R^2	AICc	ΔAICc
FO% Hoki (F)	SSTA (Feb)	2.340	0.393	5.947	0.001	0.831	22.8	-9.8
N% Hoki (F)	SSTA (Feb)	1.381	0.391	3.529	0.012	0.621	22.7	-3.4
IRI% Fish	SSTA (Feb)	1.794	0.548	3.275	0.017	0.582	28.1	-2.6
FO% Fisheries discards	Hoki catches FI (Feb)	<0.001	<0.001	3.187	0.019	0.567	25.3	-2.3

3.3.4. Discard consumption and breeding success

We found a significant negative relationship between black-browed albatross breeding success and consumption of hoki (FO%), grenadier (FO% and N%), and fishery discards (FO%; **Table 3.3**, **Figure 3.4**). The relationship with fish (IRI%) was significant only before the Bonferroni correction (**Table 3.3**). Inspection of residuals and individual Hosmer-Lemeshow tests found no indication that model fit was poor (all $p > 0.05$) (Hosmer et al. 2013).

Table 3.3 Outputs of GLMM with logit link function assessing black-browed albatross breeding success as a function of diet estimates and goodness of fit validation statistics. ΔAICc is the difference in AICc compared to the null model. Coefficient estimates (Est.) and standard errors (Std. Err.) are on the logit scale. Hoki (F) indicates that only specimens categorised as fisheries discards were considered in the analysis. We only include models that showed higher predictive power compared to the null model (ΔAICc ≥ 2).

							Conditional model	
Fixed effect	Est.	Std. Err.	z-value	p-value	Marginal R^2	Conditional R^2	AICc	ΔAICc
FO% Fish. discards	-0.024	0.005	-5.24	<0.001	0.073	0.089	2473	-9.8
IRI% Fish	-0.025	0.010	-2.43	0.015	0.040	0.090	2480	-2.4
FO% Hoki (F)	-0.021	0.006	-3.40	<0.001	0.057	0.091	2477	-5.2
FO% Grenadier	-0.024	0.005	-4.41	<0.001	0.066	0.089	2475	-7.8
N% Grenadier	-0.048	0.013	-3.77	<0.001	0.058	0.087	2477	-6.1

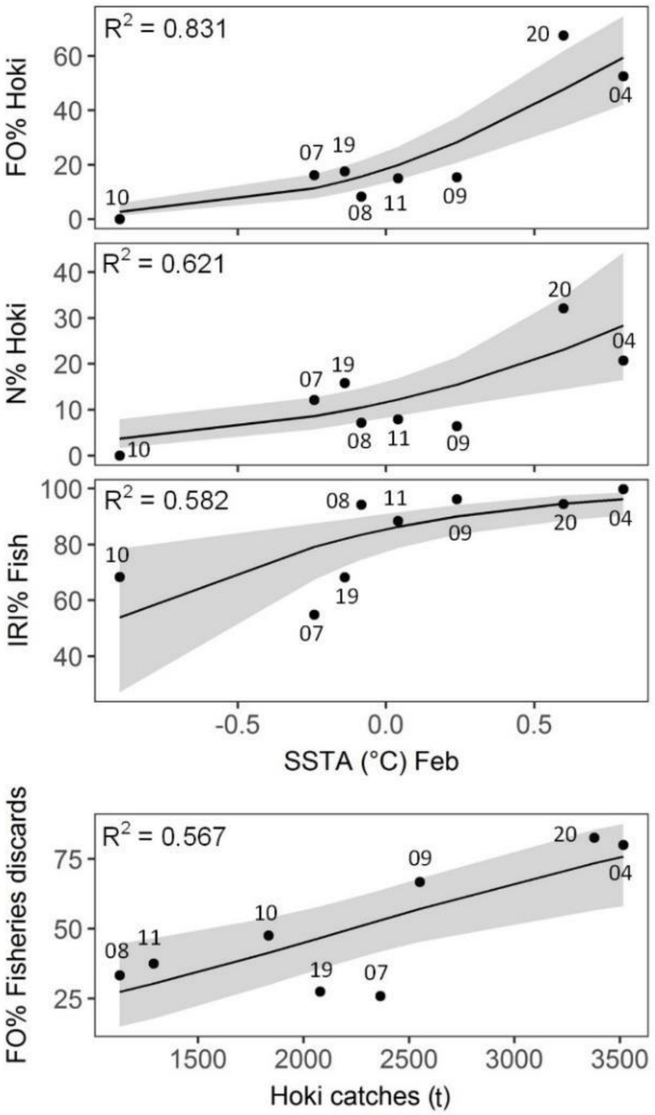


Figure 3.3 Prey consumption ± 95% confidence intervals by black-browed albatross chicks in relation to sea surface temperature anomalies (SSTA) in February, and February commercial catches of hoki in the Falkland Islands. The raw data points are superimposed, with the year indicated (04 = 2004, etc). We only include models that showed higher predictive power compared to the null model (ΔAICc ≥ 2). FO% = percentage frequency of occurrence, N% = percentage numeric frequency, IRI% = percentage index of relative importance.

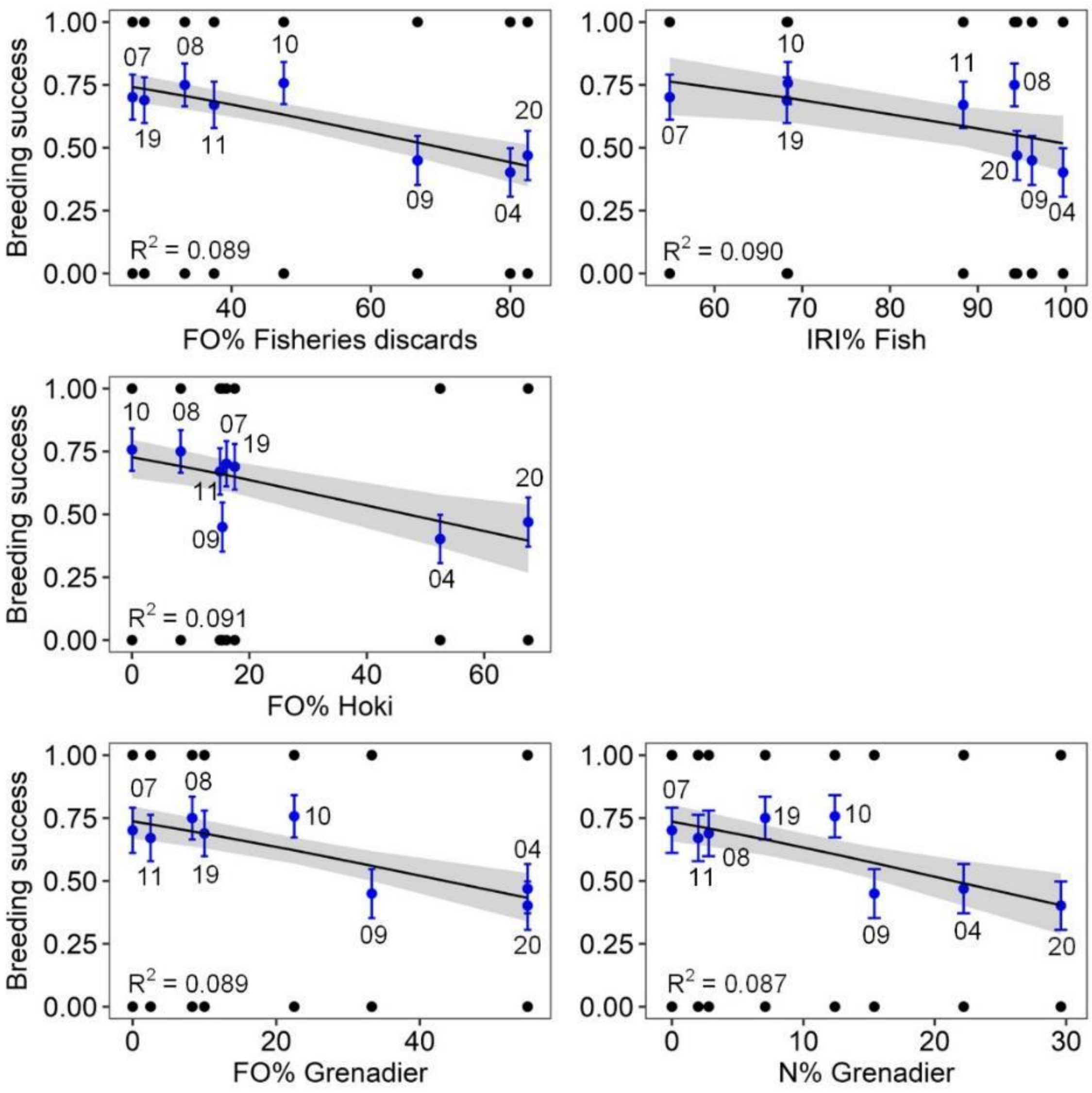


Figure 3.4 GLMM outputs ± 95% confidence intervals assessing breeding success of black-browed albatross at New Island, Falkland Islands, against various metrics of discard consumption. The raw binary data (black dots), average annual breeding success values ± 95% confidence intervals (blue dots), and the conditional R^2 value are provided. The numbers indicate the year (04 = 2004, etc.). We only include models that showed higher predictive power compared to the null model (ΔAICc ≥ 2). FO% = percentage frequency of occurrence, N% = percentage numeric frequency, IRI% = percentage index of relative importance.

3.4. Discussion

Our study represents the first long-term assessment of black-browed albatross chick diets from the world's largest population in the Falkland Islands and provides essential information on fisheries interactions for this species. We found that chicks at New Island were fed predominantly on naturally foraged crustaceans, cephalopods and fish, but also regularly received fishery discards. Moreover, the quantity of discards in the diet was positively related to fishery catches of hoki and SSTA (**Figure 3.3**), and negatively related to breeding success (**Figure 3.4**). Our findings suggest a prey-switching behaviour, whereby black-browed albatross increase discard consumption in years of poor natural foraging conditions when breeding success is reduced.

3.4.1. Diet composition and trends

Throughout the eight study seasons between 2004 and 2020, fish and crustaceans dominated chick diet, while cephalopods represented the lowest proportion of these three top prey groups (**Figure 3.2**). The importance of fish, in particular, corroborates findings for black-browed albatross diets in the Falkland Islands previously (Thompson 1992; McInnes et al. 2017b) and elsewhere, including at colonies in South Georgia, (Prince 1980; Reid et al. 1996; McInnes et al. 2017b; Mills et al. 2020), Kerguelen (Cherel et al. 2000, 2002; McInnes et al. 2017b), Chile (Arata and Xavier 2003; McInnes et al. 2017b), Australia (McInnes et al. 2017a) and New Zealand (McInnes et al. 2017b). Compared to stomach contents examined in 1986/87 (Thompson 1992), the occurrence of crustaceans was higher, whilst fish and cephalopods has remained similar. This may reflect an expansion of pelagic swarms of lobster krill on the Patagonian Shelf (Diez et al. 2016b).

The species diversity of the black-browed albatross diet was lower at New Island compared to other sites such as South Georgia (Mills et al. 2020) or Diego Ramirez, Chile (Arata and Xavier 2003), which likely reflects food web differences or prey availability within the Patagonian Shelf Large Marine Ecosystem. Total species diversity, as well as diversity within certain years, was nonetheless higher than previously identified for New Island using stomach content analysis (Thompson 1992) and DNA (McInnes et al. 2017a). Key prey included species of commercial interest and of ecological importance within the Patagonian Shelf, and were generally comparable to those identified in previous dietary studies from this colony (Thompson 1992; McInnes et al. 2017a).

Across years, between 68 and 98% of samples contained natural prey, of which lobster krill, followed by Loligo, were the most frequent and abundant. In addition to Fuegian sprat, these species form the dominant dietary component of many marine predators breeding in the Falkland Islands, including seabirds (Baylis et al. 2014; Handley et al. 2017). Fuegian sprat, one of the most important pelagic fish in the Southwest Atlantic (Agnew 2002; García Alonso et al. 2020), was scarce in our samples (except in 2019). However, large numbers of unidentifiable small fish lenses were found in the samples, and a previous DNA-based diet study from New Island found Fuegian sprat to be the dominant fish prey in 2013/14 and 2014/15 (McInnes et al. 2017a). Stomach content analysis provides a high level of taxonomic resolution but can result in the underestimation of more fragile and easily digested species (Votier et al. 2003; McInnes et al. 2016) and taxa with small otoliths (van Heezik and Seddon 1989). It is therefore possible that we underestimated the importance of Fuegian sprat, and potentially other natural prey such as jellyfish (McInnes et al. 2017b).

Complementary methods such as DNA metabarcoding or stable isotope analysis could help confirm this (McInnes et al. 2016). The distinct isotopic signature of pelagic Fuegian sprat to that of other key prey can help determine the importance of these fish in black-browed albatross chick diet (e.g. Granadeiro et al. 2014; A. Kuepfer unpubl. data). While the isotopic signature of jellyfish overlaps with that of lobster krill in our system (A. Kuepfer unpubl. data), McInnes et al. (2017b) successfully used DNA to demonstrate high consumption of jellyfish by breeding black-browed albatrosses (though less so at New Island than at Steeple Jason Island).

Fishery discards were present in 23 to 88% of samples across years, illustrating considerable inter-annual variation in discard consumption. Discard species were representative of target or bycatch species of trawl fisheries, with hoki and grenadier being the most consistent and abundant discard species in stomach samples. In summer, Falkland Islands hoki catches tend to be focused in deep waters to the southwest of the Falkland Islands where it is most abundant (Falkland Islands Government 2020). The grenadier species *C. fasciatus*, not previously reported in black-browed albatross diet studies, is a frequent bycatch species in the Falkland Islands hoki fishery (Winter and Lee 2018; Lee et al. 2019). Between 2008 and 2011, rock cod was the dominant fish prey in diet samples, which coincides with the rise and collapse of rock cod biomass and its importance as a commercial species in the Falkland Islands waters between 2007 and 2015. This followed the decline of the Southern blue whiting biomass and catches in this fishery (Laptikhovsky et al. 2013; Baylis et al. 2021; Winter 2021), which, too, is reflected in the chick diet in our study.

3.4.2. Relationship between breeding success, discards and foraging conditions

As hypothesised, the relationship between discard consumption and breeding success in New Island black-browed albatross was negative (**Figure 3.4**). A negative relationship between discard consumption and breeding success has previously been linked to the "junk-food" hypothesis, which suggests that chick development can be adversely affected if chicks are fed on discards that are often lean compared with natural prey (e.g. Grémillet et al. 2008b). Lipid-poor diets can negatively affect chick development (Kitaysky et al. 2006; Mullers et al. 2009). Demersal fish, such as grenadier, tend to have a lower caloric content compared to pelagic prey such as Fuegian sprat (c. 4.8 $kJ.g^{-1}$ vs 7.2 $kJ.g^{-1}$ wet mass, respectively; Ciancio et al. 2007). Hoki, however, has a lipid content comparable to that of sprat due to its fatty liver (6.51 $kJ.g^{-1}$ wet mass, Eder and Lewis 2005), and hence not all discards consumed by New Island chicks can be considered nutritionally poor. Besides the nutritional content of prey, other interacting aspects may make discards suboptimal meals, such as prey abundance and location (e.g. Jodice et al. 2006), or the risk of incidental mortality in fishing gear. Changes in foraging distribution and provisioning rates have been offered as an explanation for relationships between diet and breeding success in the closely related grey-headed albatross *T. chrysostoma* at South Georgia (Xavier et al. 2013; Mills et al. 2020). Such an effect would have to be assessed for Falkland Islands breeding black-browed albatrosses using GPS tracking data.

While discard consumption showed a positive relationship with February hoki catches in the Falkland Islands fishery (**Figure 3.3**), the positive influence of February SSTA on consumption of hoki (N% and FO%) and fish (IRI%) suggests that discard use may not simply be influenced by its availability, but could also be driven by underlying environmental conditions (Wren et al. 2019). High sea surface temperature can in some systems negatively impact primary production, particularly in upwelling and frontal ecosystems like the Patagonian Shelf (Behrenfeld et al. 2006; Barbraud et al. 2012; but see elsewhere e.g. Inchausti et al. 2003; Rolland et al. 2008). In turn, this can result in poor feeding conditions for seabirds (reviewed in Quillfeldt and Masello 2013; Carroll et al. 2015; Furness 2016), and negatively impacts black-browed albatross breeding success at New Island (Ventura et al. 2021a, b). Moreover, and possibly more relevant given the absence in time-lag, SSTA can have a direct effect on prey availability by shifting frontal zones and preferred foraging habitats (see e.g. Wren et al. 2019; Ventura et al. 2021c), potentially driving birds to feed more on discards (see e.g. Votier et al. 2004; Conners et al. 2018). It is possible that our results could be confounded if hoki preferred warmer waters; however, the current and previous research have found no relationship between abundance and temperature in hoki (Alemany et al. 2018), including the closely related *M. novaezelandiae* (Francis et al. 2006). We did not find a relationship between discard consumption and NPP, but a diet shift by black-browed albatross and grey-headed albatross in relation to primary production has been demonstrated in the South Georgia and Scotia Sea marine ecosystem (Mills et al. 2020).

Our findings, though based on a limited number of years, support a diet-switching hypothesis whereby black-browed albatross increase discard consumption during years of poor natural foraging conditions, characterised by high SSTA. Such a diet switch has been recorded in North Pacific black-footed albatross *Phoebastria nigripes* that increased discard consumption during a year of record-high SST (Conners et al.

2018) and also in the Benguela system where Cape gannets *Morus capensis* increased discard consumption when natural prey were scarce (Tew Kai et al. 2013; Cohen et al. 2014). In light of predicted rising sea surface temperatures as a result of climate change (IPCC 2019; Franco et al. 2022), this pattern warrants particular attention in the context of conservation. The impact of environmental and fishery effects on prey availability are likely to be highly complex and system-specific, however (Inchausti et al. 2003; Rolland et al. 2008; Wren et al. 2019). For example, Rolland et al. (2008) found that SSTA and trawling effort were positively correlated with breeding success of black-browed albatross on Kerguelen.

3.4.3. Discard availability and population trends

It has previously been suggested that abundant discards facilitated an increase in the Falkland Islands black-browed albatross population (Croxall and Gales 1998; McInnes et al. 2017a). We confirm that albatross chicks are primarily fed natural prey (see also Granadeiro et al. 2014; McInnes et al. 2017b), although discards are clearly important, particularly during periods of poor natural foraging conditions. Discards may provide a buffering effect of unknown magnitude during such years (Church et al. 2019), and changes to discard availability as a consequence of discard management in the Falkland Islands trawl fleet could reduce such a buffering effect. Importantly, however, our data suggest that discards do not fully compensate for poor natural foraging conditions during the breeding season in the Falkland Islands because the breeding success in these years of increased discard consumption remained low. This is consistent with previous findings that, much like other foods scavenged from humans (e.g. Reusch et al. 2020), fishery discards do not provide an adequate alternative to natural food for birds during breeding (Mullers et al. 2009; Cohen et al. 2014; Le Bot et al. 2019). Critically, however, there remains an important knowledge gap relating to the role of discards in the survival of immatures, and resulting effects on demography. Continued monitoring of black-browed albatross diet in the Falkland Islands, along with an improved understanding of natural prey availability through systematic surveys would provide increased insight into conclusions drawn. Furthermore, additional research is required to understand the incidence and implications of discard consumption during more cryptic life-stages, including non-breeding and juvenile birds.

In a wider context, our study highlights the importance of diet monitoring when assessing population dynamics and effects of changing discard management (Votier et al. 2004, 2008b). In a period of climatic change and a drive towards an ecosystem-based approach to fisheries management, we urge marine management policies to practice caution when assuming that fishery discards are entirely beneficial to seabirds, and recommend improved monitoring of non-commercial, lower and mid-trophic species to enhance our understanding of the ecosystem health and variability.

3.5. Supplementary information

Species accumulation curves suggest that prey diversity was slightly higher than identified from the 30 to 40 samples collected annually (except 2008, n = 12; **Figure S3.1**). However, a flattening of the curve in all years suggests sufficient sampling for the dominant prey species.

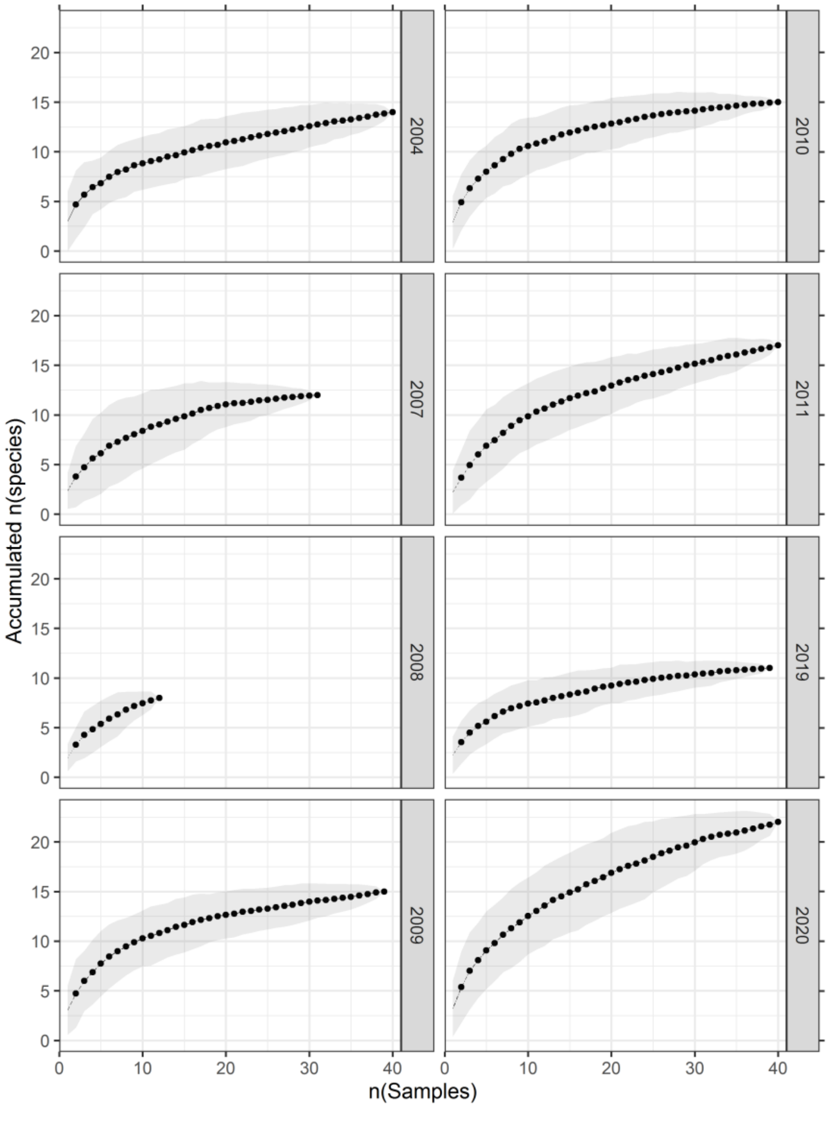


Figure S3.1 Randomised prey species accumulation curves with 95% confidence intervals for diet samples (regurgitates) collected from black-browed albatross chicks at New Island, Falkland Islands, between 2004 and 2020.

Table S3.1 Species and their corresponding habitats. ** = Non-commercial species known to be naturally accessible to black-browed albatrosses during at least part of their life-cycle / diel migration; * = commercially targeted species that are naturally accessible to black-browed albatrosses during at least part of their life-cycle / diel migration; ‡ = commercially targeted species that are not naturally accessible to black-browed albatrosses. ^{x} = non-commercial species that are not naturally accessible to black-browed albatrosses but that are readily made available through discards.

Species	Common name	Habitat	Reference	Categorised as discard or natural prey
Fish				
Coelorinchus fasciatus^x	Banded whiptail	Bentho-demersal	Lee et al. 2019	Discard
Stromateus brasiliensis^x	Butterfish	Demersal-pelagic	Ramilo-Fernández and Sotelo 2020	Discard
Antimora rostrata^x	Blue antimora	Benthopelagic deep-sea	Korostelev et al. 2019	Discard
*Gymnoscopelus nicholsi***	Myctophid sp.	Mesopelagic	Klemmedson et al. 2020	Either (*Not included in analysis, n = 1*)
Salilota australis‡	Red cod	Demersal	Agnew 2002	Discard
Notophycis marginatus^x	Dwarf codling	Demersal	Zapata-Hernández et al. 2014	Discard
Merluccius hubbsi‡	Common hake	Demersal (pelagic in first year)	Costa et al. 2019	Discard
Psychrolutidae indet^x		Demersal, deep-water	Laptikhovsky et al. 2017	Discard
Patagonotothen ramsayi‡	Patagonian rock cod	Benthopelagic	Laptikhovsky et al. 2013	Discard
*Micromesistius australis**‡	Southern blue whiting	Benthopelagic/ pelagic/ mesopelagic	Agnew 2002	Either, size dependent
*Macruronus magellanicus**‡	Hoki	Benthopelagic/ pelagic	Agnew 2002; Riccialdelli et al. 2013	Either, size dependent
*Sprattus fuegensis***	Fuegian sprat	Pelagic	Agnew 2002	Natural
Cephalopod				
*Illex argentinus**‡	Argentine shortfin squid	Benthopelagic/ pelagic	Agnew 2002; Riccialdelli et al. 2013	Natural
*Doryteuthis gahi**‡	Loligo	Demersal/ benthopelagic	Agnew 2002; Riccialdelli et al. 2013	Natural
*Moroteuthis ingens**	Greater hooked squid	Pelagic	Agnew 2002	Natural

Crustacean				
*Munidae gregaria***	Lobster krill (gregaria morph)	Benthopelagic/ pelagic	Agnew 2002	Natural
*Themisto gaudichaudii***		Pelagic	Padovani et al. 2012	Natural
Peltarion spinulosum^{x}	Little purple-back crab	Benthic	Gorny 1999	Discard
*Euphausiids***		Pelagic	Gibbons 1997	Natural

Table S3.2 Allometric formulae used for size-class reconstruction based on sagittal otolith length (OL). TL = total length (mm), PAL = preanal length (mm).

Species	Formula	Reference
Macruronus magellanicus	TL:OL = 35:1	Arata and Xavier 2003
	TL = PAL *0.43	FIFD unpubl. data; Chong et al. 2007
Micromesistius australis	lnTL=-0.259+1.47*(lnOL)	Thompson 1992

Table S3.3 Summary of percentage frequency of occurrence (FO%) and percentage numeric frequency (N%) of individual species found in stomach samples of black-browed albatross chicks at New Island, Falkland Islands. In 2003/04, 2006/07 and 2007/08, only *Munida* sp. was considered, as other crustaceans were residual. Species with ‡ indicate commercially targeted species; species with ^{x} indicate bycatch species. ‡‡ Rockcod is only targeted since 2007; catches of southern blue whiting have been restricted since 2011.

Species	2003/04		2006/07		2007/08		2008/09		2009/10		2010/11		2018/19		2019/20	
	FO%	N%	FO%	N%	FO%	N%	FO%	N%	FO%	N%	FO%	N%	FO%	N%	FO%	N%
CRUSTACEAN																
Decapoda																
Munida sp.	90	-	83.9	-	75	-	94.9	-	47.5	-	22.5	-	65	-	60	-
Peltarion spinulosum	0	-	0	-	0	-	0	-	0	-	0	-	0	-	2.5	-
Amphipoda																
Themisto gaudichaudii	0	-	0	-	0	-	12.8	-	35	-	15	-	5	-	7.5	-
Amphipod sp.	0	-	0	-	0	-	0	-	0	-	0	-	0	-	17.5	-
Euphausiacea																
Euphausiids	0	-	0	-	0	-	0	-	0	-	2.5	-	0	-	0	-
Isopoda																
Isopod sp.	0	-	0	-	0	-	0	-	0	-	5	-	0	-	2.5	-
Unidentified spp.																
Crustacean sp.	0	-	3.2	-	0	-	0	-	0	-	0	-	2.5	-	0	-
FISH																
Moridae																
Antimora rostrata^{x}	20	5.4	0	0	0	0	2.6	0.5	7.5	1.5	0	0	0	0	0	0
Salilota australis‡	2.5	0.5	3.2	1.5	0	0	10.3	2.1	0	0	2.5	1	0	0	2.5	0.6
Notophycis marginatus^{x}	0	0	0	0	0	0	0	0	0	0	0	0	0	0	20	5.6
Gadidae																

Micromesistius australis‡‡	45	20.2	9.7	6.1	0	0	2.6	1.1	2.5	0.5	0	0	0	0	10	1.9
Gadoid sp.	2.5	1.5	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Coelorinchus fasciatus[X]	55	29.6	0	0	8.3	7.1	33.3	15.4	22.5	12.4	2.5	2	10	2.8	55	22.2
Psychrolutidae indet[X]	2.5	0.5	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Merlucciidae																
Merluccius hubbsi‡	7.5	2	6.5	3	0	0	5.1	1.1	0	0	0	0	0	0	0	0
Macruronus magellanicus‡	55	27.1	16.1	12.1	16.7	14.3	17.9	10.6	5	1	20	13.9	32.5	13.5	70	32.7
Merluccidae indet.‡	0	0	0	0	0	0	2.6	0.5	2.5	1	2.5	1	0	0	5	1.5
Clupeidae																
Sprattus fuegensis	0	0	0	0	0	0	0	0	17.5	9.8	2.5	1	35	30.3	17.5	10.2
Myctophidae																
Gymnoscopelus nicholsi	0	0	0	0	0	0	0	0	0	0	0	0	2.5	0.6	0	0
Stromateidae																
Stromateus brasiliensis[X]	0	0	0	0	0	0	0	0	0	0	0	0	0	0	5	0.6
Nototheniidae																
Patagonotothen ramsayi‡‡	20	6.4	9.7	6.1	16.7	21.4	46.2	45.7	40	21.1	22.5	23.8	0	0	0	0
Patagonotothen sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2.5	0.3
Unidentified spp.																
Unidentified fish spp.	17.5	4.4	41.9	19.7	41.7	35.7	30.8	11.2	27.5	6.2	62.5	24.8	5	1.1	7.5	3.7
CEPHALOPODS																
Teuthida (squid)																
Gonatidae																
Gonatus antarcticus	0	0	0	0	0	0	0	0	0	0	2.5	2	0	0	0	0
Loliginidae																
Doryteuthis gahi‡	2.5	1	22.6	27.3	0	0	0	0	55	42.3	17.5	13.9	40	42.1	25	18.5
Ommastrephidae																

Illex argentinus‡	0	0	6.5	7.6	0	0		0	2.5	1	2.5	1	10	4.5	5	1.2
Onychoteuthidae																
Moroteuthis ingens	5	1	6.5	3	0	0	7.7	4.3	0	0	7.5	3	0	0	0	0
Unidentified spp.																
Teuthida indet.	0	0	29	13.6	16.7	14.3	20.5	6.9	10	2.1	20	9.9	5	5.1	5	0.6
Octopoda																
Unidentified spp.																
Octopoda indet.	2.5	0.5	0	0	8.3	7.1	2.6	0.5	5	1	7.5	3	0	0	2.5	0.3

Table S3.4 Output from the SIMPER analysis, averaged across years, for species contributing 50% to the annual differences observed in diet composition of black-browed albatross chicks.

Species	Average dissimilarity contribution (± standard deviation)
Munida sp	0.11 ± 0.14
Macruronus magellanicus	0.09 ± 0.12
Doryteuthis gahi	0.08 ± 0.10
Patagonotothen ramsayi	0.08 ± 0.11
Coelorinchus fasciatus	0.07 ± 0.09
Themisto gaudichaudii	0.04 ± 0.08
Sprattus fuegensis	0.04 ± 0.07

Table S3.5 Linear model assessing the relationship between February hoki catches in the Falkland Islands and February SSTA.

		Conditional model			
Explanatory variable	Fixed effect	Est.	Std. Err.	z-value	p-value
Hoki catches (2004-2020)	SSTA Feb (2004-2020)	425.1	578.6	0.735	0.474

Chapter 4 – Inter-colony and inter-annual variation in discard consumption of albatross chicks revealed using stable isotope and regurgitates

This chapter has been published as:

Kuepfer A, Catry P, Bearhop S, Sherley RB, Bell O, Newton J, Brickle P, Arkhipkin A, Votier SC (2023) Inter-colony and inter-annual variation in discard use by albatross chicks revealed using isotopes and regurgitates. Marine Biology. 170:46. doi 10.1007/s00227-023-04191-7

Abstract

Effective marine ecosystem monitoring is critical for sustainable management. Monitoring seabird diets can convey important information on ecosystem health and seabird-fishery interactions. The diet of breeding black-browed albatross (*Thalassarche melanophris*) has previously been assessed using stomach content analysis (SCA) or stable isotope analysis (SIA), but not both methods together. Combining dietary sampling approaches reduces biases associated with using single methods. This study combines SCA and SIA to study the diet of black-browed albatross chicks, with a specific focus on fishery discard consumption, at two Falkland Islands colonies (New Island 51˚43’S, 61˚18’W and Steeple Jason Island 51˚01’S, 61˚13’W) during two consecutive breeding seasons (2019 and 2020). SCA provided high taxonomic resolution of short-term diet and priors for stable isotope mixing models, with multiple measures of dietary items (e.g. numeric frequency N%, frequency of occurrence FO%). By contrast, SIA of down feathers provided a single and more integrated dietary signal from throughout chick development. Although the two methods disagreed on the dominant prey group (SCA – crustacean; SIA – pelagic fish), the complementary information suggested a chick diet dominated by natural prey (SCA: 74%–93% [FO], 44%–98% [N]; SIA: minimum 87%–95% contribution). Nonetheless, SCA revealed that a high proportion of breeding adults do take discards. We detected consistent colony-specific diets in relation to prey species, but not in relation to higher discard use. Overall, discard consumption was highest in 2020, the year characterised by the poorest foraging conditions. Our results have implications for fisheries management and future dietary studies assessing discard use.

4.1. Introduction

In an era of rapid marine ecosystem change, monitoring is critical for the sustainable management of our oceans. Top predators, such as seabirds, are often regarded as useful indicators of marine ecosystem health (Velarde et al. 2019; Bestley et al. 2020), but indices require careful consideration (Durant et al. 2009). For example, changes in seabird demography, distribution, or breeding biology can reflect environmental change (e.g. Votier et al. 2005; Cury et al. 2011; Sydeman et al. 2021), or impacts of anthropogenic activities such as fishing (e.g. Einoder 2009; Pardo et al. 2017; Sherley et al. 2017). However, seabird populations and distributions alter relatively slowly with time (Nevoux et al. 2010) and non-breeders may buffer the effects of environmental or anthropogenic change (Votier et al. 2008a), making it challenging to detect such impacts (e.g. Sherley et al. 2018). By contrast, dietary variation may be a more sensitive indicator of change.

Seabird diets reflect an individual’s foraging behaviour, including prey preference, foraging distribution and dietary flexibility (e.g. Gaglio et al. 2018; Church et al. 2019), and indicate the quantity, quality, and accessibility of prey at the species and population level (Buren et al. 2012; Velarde et al. 2019). In addition, seabird diets may provide information on fishery interactions, including variation in discard use (Votier et al. 2004, 2008b), or bycatch risk (Einoder 2009; Phillips et al. 2016). Methods to accurately study seabird diets can therefore form an integral part of monitoring strategies and ecosystem-based approaches to fisheries management, such as in the Commission for the Conservation of Marine Living Resources (CCAMLR) Ecosystem Monitoring Programme (CEMP) (also see e.g. Scopel et al. 2019).

A range of morphological and biochemical methods exist for assessing diet, and each provide different information and biases (Votier et al. 2003; Barrett et al. 2007). Stomach content analysis (SCA) and stable isotope analysis (SIA) are common approaches for assessing seabird diet (Duffy and Jackson 1986; Barrett et al. 2007; McInnes et al. 2016). SCA provides high taxonomic resolution, but this tends to reflect short time scales (days to weeks), and can underestimate soft-bodied prey (Votier et al. 2003; Inger and Bearhop 2008; McInnes et al. 2017b). Conversely, SIA typically provides dietary information integrated over longer time scales during tissue growth (weeks to months; Inger and Bearhop 2008; Phillips et al. 2014). However, stable isotope mixing models, which are used to quantify prey source contribution to a diet mixture, rely on prior knowledge of diet and are only informative if key prey differ in their isotopic composition (Inger and Bearhop 2008; Phillips et al. 2014). Combining morphological and biochemical analyses can address the pitfalls of individual methods (Karnovsky et al. 2012; Bonin et al. 2020), and a complementary approach is therefore recommended when using seabird diet for monitoring (Le Bot et al. 2018; Ceia et al. 2022).

The black-browed albatross *Thalassarche melanophris* (hereafter BBA) is the world's most abundant species of albatross, and its diet has been well-studied (see McInnes et al. 2017a). It is used as a bio-indicator in the CEMP, as well as a sentinel of the Patagonian Shelf Large Marine Ecosystem (Ventura et al. 2021a). The population breeding in the Falkland Islands is of particular interest, because it is the world's largest (>70% of breeding populations), and, in contrast to several other populations, is increasing in size (BirdLife International 2018).

BBA breeding in the Falkland Islands forage over the Patagonian Shelf, where large numbers scavenge at fishing vessels for discards, consequently falling victim to bycatch (Granadeiro et al. 2011, 2014; Kuepfer et al. 2022a, b). During non-breeding, discards may be prevalent in the diet of adults (Mariano-Jelicich et al. 2013; although see Granadeiro et al. 2014) and juveniles (Colabuono & Vooren 2007) from the Falkland Islands (but perhaps less so elsewhere, e.g. Petersen et al. 2008). During breeding, however, dietary studies in the Falkland Islands suggest that BBA predominantly consume natural prey (Thompson 1992; Kuepfer et al. 2022a (SCA); Granadeiro et al. 2014 (SIA); McInnes et al. 2017a (DNA)), although discards appear important when natural foraging conditions are unfavourable (Kuepfer et al. 2022a). Discard consumption also varies among colonies, with larger colonies thought to consume more discards, possibly due to increased competition for natural prey during central place foraging (Thompson 1992; McInnes et al. 2017a). However, the previous diet studies all applied different individual methods (SCA: Thompson 1992; Kuepfer et al. 2022a; DNA: McInnes et al. 2017a, b; SIA: Granadeiro et al. 2014), and (apart from

McInnes et al. 2017a) focused on either individual years or individual colonies. It is therefore difficult to distinguish methodological biases from temporal and spatial variation in diets when comparing results. At a time when discard management in the Falkland Islands is changing (through the introduction of batch discarding by trawlers; Kuepfer and Barton 2018; Kuepfer et al. 2022b), and the climate warming across BBAs' range (Franco et al. 2022), it is important to gain a more comprehensive dietary understanding for this globally significant population.

Here we use SCA and SIA to study BBA chick diet at two Falkland Islands colonies from two consecutive breeding seasons. Specifically, we (1) quantify diet, (2) assess the importance of discards and natural prey, and (3) determine how diet varies between colonies and years. Considering previous findings, we hypothesise that (1) natural prey will dominate across years and colonies but that (2) discard consumption will be higher at the larger colony.

4.2. Materials and methods

4.2.1. Study area and fleet characteristics

Fieldwork took place in the austral summer of 2018/2019 and 2019/2020 (hereafter 2019 and 2020) during mid-chick rearing (~ 6–12 weeks of age) at New Island (NWI, 51˚43'S, 61˚18'W) and Steeple Jason Island (SJI, 51˚01' S, 61˚13'W), located in the west and north-west of the Falkland Islands (**Figure 4.1**). NWI and SJI respectively support approximately 16,000 and 210,000 breeding pairs of BBA (Crofts 2020).

Chick-rearing adults from NWI and SJI concentrate their foraging on the southern Patagonian Shelf, although they generally segregate by colony while at sea (NWI: south-west / west; SJI: north-west / north; Catry et al. 2013). The area is extensively fished, predominantly by bottom trawlers for finfish and jiggers for Argentine shortfin squid (*Illex argentinus*, hereafter *Illex*) (Seco Pon et al. 2015; Falkland Islands Government 2021). There is also some longlining (2–4 vessels within the Falklands Conservation Zones (FCZs) and the Argentine Exclusive Economic Zone (AEEZ) combined at <40° S latitude; Seco Pon et al. 2015; Falkland Islands Government 2021)). Discards of demersal and benthopelagic fish are available to seabirds from bottom-trawl vessels throughout the year, including during chick rearing (**Figure S4.1**). Squid discards are limited, as *Illex* and Patagonian long-finned squid (*Doryteuthis gahi*, hereafter Loligo) are generally packed whole, and discarding of the latter is prohibited in Falkland Islands waters (Laptikhovsky et al. 2006; Arkhipkin et al. 2015).

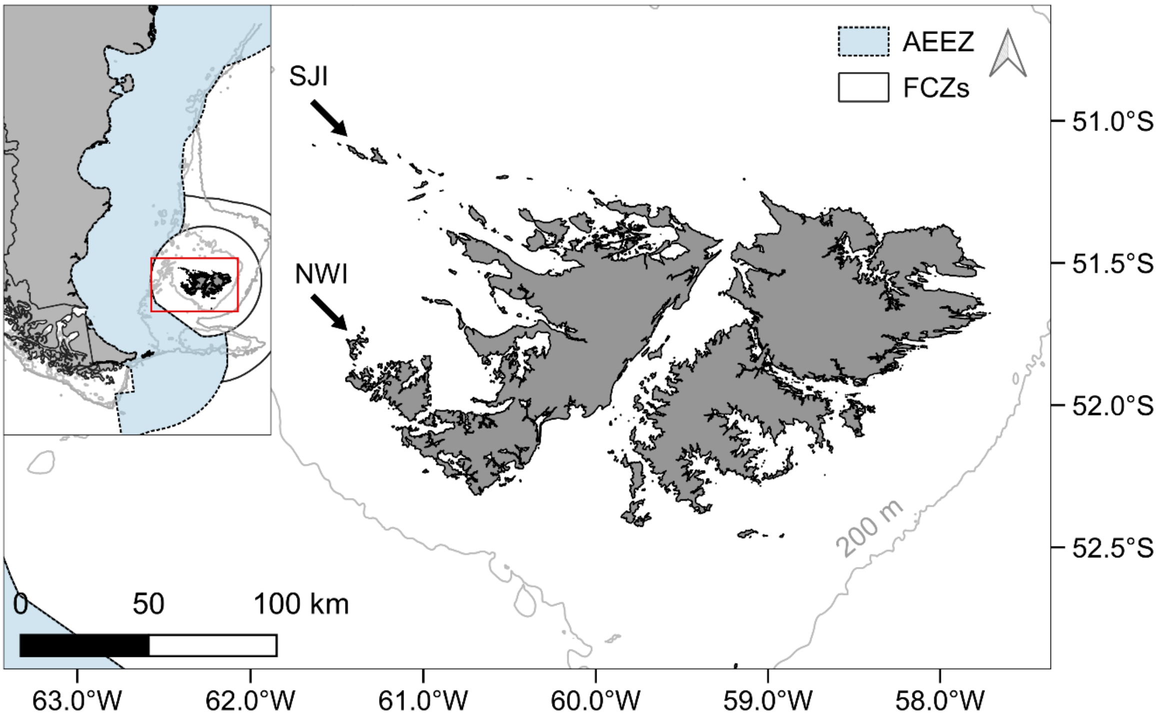


Figure 4.1 Location of New Island (NWI) and Steeple Jason Island (SJI) in the Falkland Islands on the Patagonian Shelf, east of the South American continent. The 200-m and 1000-m depth contours are shown as grey lines. AEEZ = Argentine Exclusive Economic Zone; FCZs = Falklands Conservation Zones.

4.2.2. Sampling

Stomach content analysis

Stomach contents were collected from chicks using induced regurgitation (after Phillips 2006) during late January through to mid-February at NWI, and in mid-March at SJI (NWI 2019: 29 January – 17 February; NWI 2020: 25 January – 23 February; SJI 2019 & 2020: 12 – 16 March). For logistical reasons, it was not possible to sample both colonies simultaneously. Only freshly fed chicks were sampled, and each chick was only sampled once. To obtain a measurement of meal size and stomach oil content, individual stomach samples were weighed whole (total mass, g), and again after stomach oils had been drained (wet mass, g).

Prey were first separated into key prey groups (fish, cephalopod, crustacean, jellyfish, and carrion), and subsequently identified to the lowest possible taxonomic level using reference collections (Falkland Islands Fisheries Department (FIFD), unpubl. data; Xavier and Cherel 2009) and with assistance from specialists (B. Lee, Z. Shcherbich). Minimum number of individuals (MNI) was determined through assemblage of whole animals and fresh loose hard structures. MNI for carrion (penguin feathers) and jellyfish within individual samples was always 1, as it was impossible to determine whether parts originated from a single or multiple individuals. Eroded or brittle structures were excluded from all analyses, as these were assumed to have originated from previous meals.

Prey that were identified to species level were further categorised as fishery discards or natural prey based on numerous criteria relating to natural accessibility of prey to albatrosses, fishing and discard practices, and re-constructed prey size (see Kuepfer et al. 2022a for further details; **Table S4.1**). Large fish heads in the absence of the bodies were categorised as discards because at-sea catch processing generally involves heading and gutting of fish (Kuepfer et al. 2022a). For species where reconstructed size was used to assist with classification (hoki *Macruronus magellanicus* and southern blue whiting *Micromesistius australis*), maximum sagittal length (mm) was measured from intact otoliths (one from each otolith pair as well as unpaired otoliths) using a binocular microscope equipped with a graticule to reconstruct the original size of the prey from allometric formulae (**Table S4.2**). Finally, prey were quantified using four different metrics (see below, **Table 4.1**).

Stable isotope analysis

Multiple down feathers from across the body were collected from a random sample of 55 to 73 chicks at NWI (mid-February, chick age 62 days (Catry et al. 2011; Ventura et al. 2021a)), and at SJI (mid-March, chick age approximately 84 ± 7 days). At these ages, given the large increase in body surface area since hatching, the stable isotope composition of down feathers is no longer expected to reflect egg nutrients. Rather, they provide dietary information from the first two months of chick development and are therefore directly comparable between NWI and SJI, despite the discrepancy in sampling months.

To characterise stable isotopes of potential prey, we sampled specimens collected during the FIFD ground-fish surveys in February 2019 and 2020. Sampling focused south, west, and north of the FCZs, where GPS-tracked BBA breeding on NWI and SJI forage during chick-rearing (Granadeiro et al. 2011; Ventura et al. 2021a). Prey species were chosen based on available knowledge of BBA diet in the Falkland Islands (McInnes et al. 2017a, b; Kuepfer et al. 2022a; current study), and included demersal, pelagic and benthopelagic species of fish, cephalopods, crustaceans and jellyfish.

Prey muscle and albatross chick feathers were processed following Meier et al. (2017). In order to avoid biases in analyses of $\delta^{13}C$ related to the presence of lipids in muscle samples (see e.g. Bearhop et al. 2002; Post et al. 2007), *a priori* lipid extraction was conducted on lipid-rich samples, defined as species with a C:N > 3.5 (Post et al. 2007). The effervescent test (after e.g. Carabel et al. 2006) confirmed the absence of carbonates from all crustacean samples. Samples were weighed into tin capsules (approx. 0.7 mg) and analysed for $\delta^{13}C$ and $\delta^{15}N$ values by continuous-flow isotope ratio mass spectrometry at the National Environmental Isotope Facility Stable Isotope Ecology Laboratory, East Kilbride. Samples were analysed using a Elementar vario Pyrocube elemental analyser (2013), coupled with a Thermo Fisher Delta XP Plus Isotope Ratio Mass Spectrometer (IRMS). Ratios were corrected for instrument drift and linearity using interspersed samples of internal laboratory standards (gelatine, glycine and alanine mixtures) with known stable isotope values (for details see (Jones et al. 2020). Stable isotope ratios were expressed in δ notation in parts per thousand (‰) relative to Vienna Pee Dee Belemnite ($\delta^{13}C$) or air ($\delta^{15}N$). Precision of the measurements was 0.09‰ for $\delta^{13}C$ and 0.17‰ for $\delta^{15}N$, based on the standard deviation of the most common lab standard used (gelatine).

4.2.3. Statistical analysis

All statistical analyses were conducted in R version 4.0.0 (R Core Team 2021). The significance value of frequentist statistical tests was set at $\alpha = 0.05$, unless stated otherwise.

Stomach content analysis

To assess whether the number of stomach content samples obtained from chicks was sufficient to describe dietary species diversity, samples from individual colony/year combinations were randomized 100 times, and an accumulation curve was constructed as a function of sample size (package vegan::accumcomp, Oksanen et al. 2019). Sample sizes would be considered sufficient to describe the full diversity of the diet if the fitted cumulative prey curves reached an asymptote.

Prey were subsequently described using a range of metrics, after Barrett et al. (2007), and following the method of Kuepfer et al. (2022a) (**Table 4.1**). First, key prey groups (fish, crustacean, cephalopod, jellyfish, and carrion) were quantified as a percentage index of relative importance (IRI%) as:

$$IRI\% = \left(IRI_p \Big/ \sum_{i=1}^{n} IRI_{p,i} \right) \times 100,$$

$$IRI_p = FO\% \times (N\% + M\%)$$

(1)

where FO% is the percentage of stomach samples (*i*) in which a particular prey type (*p*) was present; N% is the number of individuals of a particular prey type present expressed as a percentage of the total prey number at that taxonomic level; and M% is the percentage of the total drained sample mass represented by a particular prey type.

Next, individual prey species, and categories constructed from identified species (i.e. fish subgroups and prey origin [discards and natural]) were quantified by FO% and N%. We did not obtain M% (and hence IRI%) at these levels, because it was impractical to separate out, and identify, every individual fish bone or loose flesh to obtain species-specific mass (see also Kuepfer et al. 2022a).

Different methods of prey quantification provide complementary dietary information: M% informs on meal size and can indicate approximate nutritional contribution of similar prey; FO% indicates the variability of prey abundance; and N% measures the frequency of prey encounter (Duffy and Jackson 1986; Barrett et al. 2007). The IRI% is considered the most comprehensive approach, as it integrates information from numerous metrics, thereby reducing biases associated with any one measure (Duffy and Jackson 1986; Liao et al. 2001; Mills et al. 2020). The absence of IRI% of prey origin (as explained above) is therefore unfortunate; however, given that crustaceans and cephalopods were categorised as natural prey, and most fish as discards (**Table S4.1**), the IRI% of key prey groups can serve as a proxy for prey origin.

Table 4.1 Metrics used to quantify prey at different taxonomic levels. M% = percentage drained mass; FO% = percentage frequency of occurrence; N% = percentage numeric frequency; IRI% = percentage index of relative importance.

	M%	N%	FO%	IRI%
Main prey groups				
Fish	✓	✓	✓	✓
Cephalopod	✓	✓	✓	✓
Crustacean	✓	✓	✓	✓
Carrion (penguin feathers)	✓	✓	✓	✓
Jellyfish	✓	✓	✓	✓
Individual species of…				
Fish		✓	✓	
Cephalopod		✓	✓	
Crustacean		✓	✓	
Origin				
Discards		✓	✓	
Natural prey		✓	✓	

Wilcoxon rank sum tests were used to assess (1) differences in meal size, and (2) differences in prey species composition between years and colonies. Year or colony were used as our explanatory variables in individual tests, whilst our dependent variables were either (1) the different meal fractions (total/ liquid/ drained); (2) counts or presence/absence of prey species from either discards or natural prey. To minimise Type 1 errors, the significance level was adjusted using the Bonferroni correction to $\alpha = 0.025$. As stomach content samples were not collected simultaneously at the two colonies, inter-colony comparison warrants some caution due to possible seasonal effects (see Discussion).

Stable isotope analysis

To compare isotopic values ($\delta^{13}C$ and $\delta^{15}N$) of chicks between colonies, we conducted multivariate analysis of variance (MANOVA) with *post-hoc* ANOVAs, and compared isotopic niche using SIBER (Stable Isotope Bayesian Ellipses in R; Jackson et al. 2011). We did not compare niche space between years due to insufficient certainty around the isotopic baseline for the two years, which makes it difficult to discern potential differences in baselines from diet shifts.

To assess the percentage contribution of different prey, a series of Bayesian mixing models with a multiplicative error structure of residual × process error and uninformative Dirichlet priors were applied using the MixSIAR package (Stock and Semmens 2016). Individual mixing models were run for each colony/year combination (NWI 2019, SJI 2019, NWI 2020, and SJI 2020). Mixing model convergence was assessed using the Gelman-Rubin and Geweke's diagnostics (Stock and Semmens 2016).

Uncertainty in SIA can be related to isotopic similarity among prey types (Phillips and Gregg 2001; Phillips et al. 2014). Further, the discriminatory power of mixing models rapidly deteriorates when more than six prey sources are included, so aggregating isotopically and biologically similar sources is recommended (Phillips et al. 2014). We therefore tested the isotopic differences between prey species using MANOVAs, and based on ecological and isotopic similarities, aggregated them into four *a priori* groups: (1) pelagic fish; (2) demersal fish; (3) benthopelagic fish + squid; (4) crustacean + jellyfish.

The decision to include or exclude a particular species as a prey source was informed by our colony/year-specific SCA results (see below, **Table 2**) – only prey species that comprised >10% (FO) of the stomach contents in any colony/year were included, as the exclusion of uncommon prey tends to improve mixing model accuracy (Phillips et al. 2014). In all models, we further included potentially important prey which may easily be missed or underestimated in SCA due to soft body-parts (i.e. jellyfish and Fuegian sprat (*Sprattus fuegensis*); McInnes et al. 2017a, b). Where the isotopic values of individual species did not differ significantly between years (MANOVAs), species were pooled from both years to increase sample size (**Table S4.4, Figure S4.2**). In the absence of stable isotope values for a particular prey species in one year, we used the values available from the other year (**Table S4.4**).

Diet-tissue trophic discrimination factors (TDFs) have not been published for Procellariidae, and we therefore used values estimated using Stable Isotope Discrimination Estimation in R (SIDER, Healy et al. 2018): Feathers: $\delta^{15}N$ 4.09 ± 1.19, $\delta^{13}C$ 2.21 ± 1.25. SIDER is designed to predict TDFs of consumers based on their ecology and phylogenetic relatedness (Healy et al. 2018), and the values determined by SIDER therefore provide for the most up-to-date and biologically justified models (also see e.g. Swan et al. 2020).

4.3. Results

4.3.1. Stomach content analysis

Stomach contents were collected from 143 chicks across the two years and colonies (NWI 2019 & 2020: n = 40; SJI 2019: n = 32; SJI 2020: n = 31). Meals were significantly heavier in 2020 at NWI due to the higher liquid portion (Mann-Whitney U tests; Total:

W = 392, P = 0.004; Liquid: W = 325, P = 0.001; Drained: W = 770, P = 0.924; **Figure 4.2**). At SJI, although the total meal size did not vary between years, liquids were significantly heavier (before Bonferroni correction), and drained samples significantly lighter in 2020 (Mann-Whitney U tests; Total: W = 290, P = 0.138; Liquid: W = 234, P = 0.036; Drained: W = 534, P = 0.019; **Figure 4.2**).

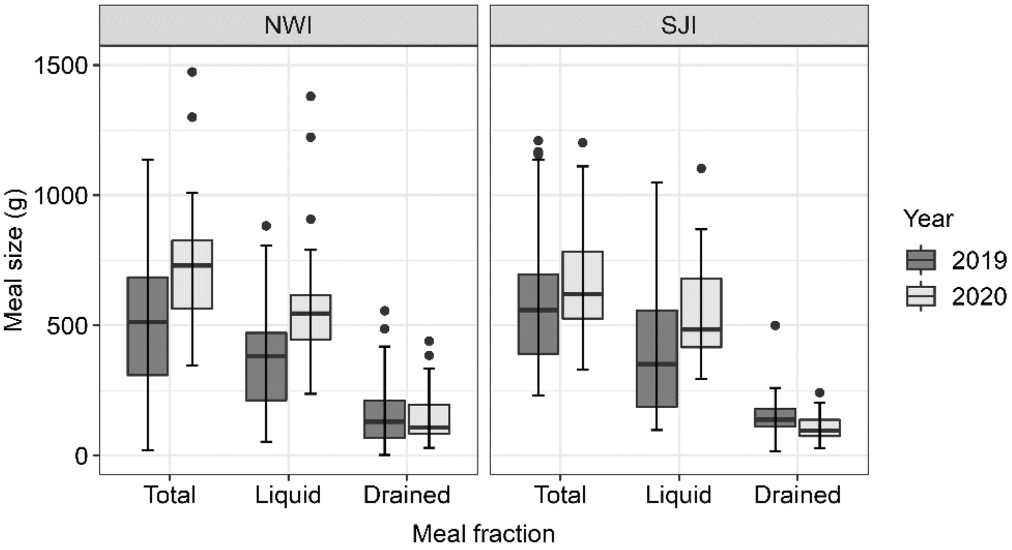


Figure 4.2 Meal sizes (g) as sampled through regurgitation from black-browed albatross chicks at New Island (NWI) and Steeple Jason Island (SJI) in 2019 and 2020. The box shows the interquartile range, with the thick horizontal line representing the median. The whiskers represent the 95% percentiles.

We extracted 9,092 prey items from 143 drained regurgitates. The three dominant prey categories were crustacean, fish and cephalopods (**Figure 4.3a**). Jellyfish and carrion (penguin species) were only found in 2020, but their relative importance remained negligible at both colonies (**Figure 4.3a, Table S4.3**). Crustaceans had the highest relative importance across colonies/years (IRI% = 49–67%), except at NWI 2019, where fish dominated with 80% (IRI). At both colonies, the IRI% of fish was approximately double in 2020 compared to in 2019 (**Figure 4.3a**). Cephalopods, dominated by squid, consistently showed smaller IRI% than fish and crustaceans across colonies and years (IRI% = 4–15%, **Figure 4.3a**).

The asymptotes of species accumulation curves were not fully reached; however, the flattening of the species accumulation curves provides confidence that the most important identifiable prey had been captured (**Figure 4.4**). Species richness was considerably higher in 2020 than 2019 at both colonies (**Figure 4.4**).

The composition of dominant species differed between colonies but was broadly consistent across years (**Table 4.2**). The most frequent and most numerous species encountered at both colonies was lobster krill (*Munida gregaria*) (**Table 4.2**). Individual lobster krill consisted exclusively of adults at NWI, whereas at SJI both adults and juveniles were present at a ratio of approximately 2:5 in 2019, and 1:50 in 2020. In terms of cephalopods, Loligo squid was the most numerous and abundant species at NWI, while *Illex* squid dominated at SJI (**Table 4.2**). In terms of fish, demersal common hake (*Merluccius hubbsi*) was the most frequent and numerous fish species at SJI in both years, followed by benthopelagic hoki (**Figure 4.3b**; **Table 4.2**). At NWI, pelagic Fuegian sprat dominated in 2019, whilst hoki dominated in 2020 (**Figure 4.3b**; **Table 4.2**). Ninety-nine percent (n = 134) of all measurable hoki, and 100% of measurable southern blue whiting (n = 6) fell within the size classes caught in the Falkland Islands fishery in 2019 and 2020 during comparable months, and we therefore classified all hoki and southern blue whiting as discards.

Prey origin varied between years at NWI, but not at SJI. Discard consumption at NWI was significantly higher in 2020 compared to in 2019 (Wilcoxon rank sum test; FO%: W = 360, $P < 0.001$; N%: W = 257, $P < 0.001$), while natural prey declined in terms of

N% (Wilcoxon rank sum test; N%: W = 1143, P < 0.001; **Figure 4.3c**). Discards also varied between colonies, being significantly higher at SJI in 2019 (Wilcoxon rank sum test; FO%: W = 366, P < 0.001; N%: W = 66, P < 0.001) but higher at NWI in 2020 (Wilcoxon rank sum test; FO%: W = 772, P = 0.024; N%: W = 825, P = 0.016; **Figure 4.3c**). Natural prey consumption did not differ between colonies regardless of year (**Figure 4.3c**).

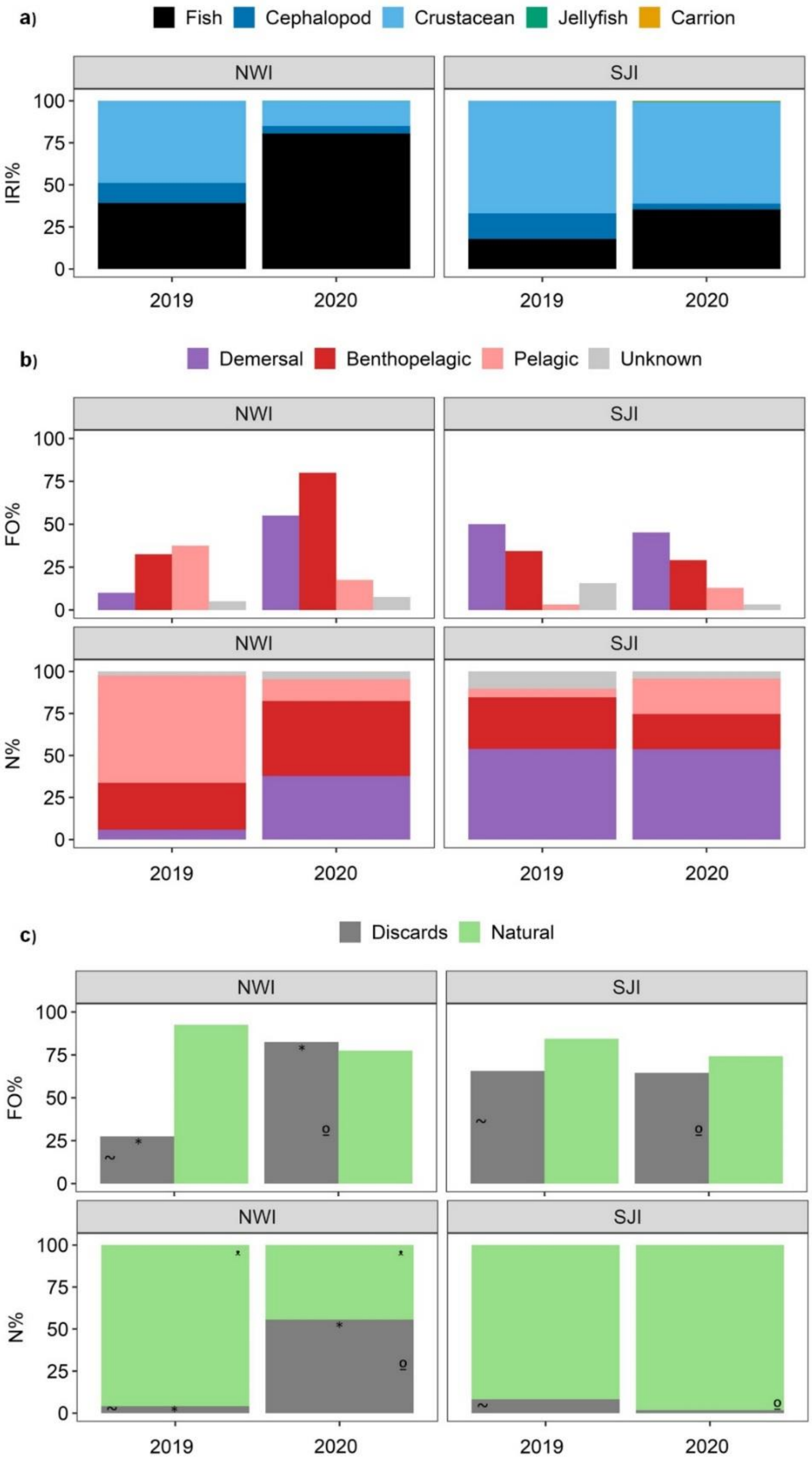


Figure 4.3 Black-browed albatross chick diet based on regurgitation from New Island (NWI) and Steeple Jason Island (SJI) in 2019 and 2020, quantified as (**a**) percentage index of relative importance (IRI%) of main prey groups; (**b**) percentage frequency of occurrence (FO%) and percentage numeric frequency (N%) of fish subgroups; and (**c**)

percentage frequency of occurrence (FO%) and percentage numeric frequency (N%) of discards and natural prey. Note that the negligible IRI% of jellyfish and carrion renders these practically invisible on the plot. The various symbols indicate which groups are significantly different from one another within each metric.

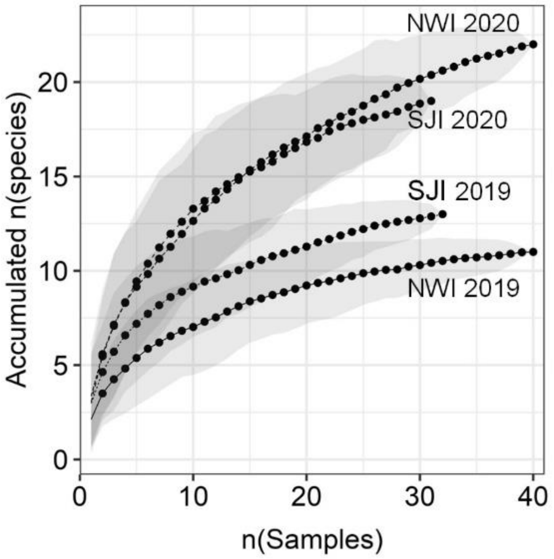


Figure 4.4 Accumulation curves with 95% confidence intervals (shaded area) of prey species extracted from black-browed albatross chick regurgitate samples collected at New Island (NWI) and Steeple Jason Island (SJI) in 2019 and 2020.

Table 4.2 Prey species found in regurgitate samples from black-browed albatross chicks at New Island (NWI) and Steeple Jason Island (SJI) in 2019 and 2020. Only species which occurred at minimum 5% frequency of occurrence (FO%) or numeric frequency (N%) in any given year / colony are presented. Values in brackets represent N% as calculated excluding crustaceans from the total prey count, as very high counts of crustaceans resulted in negligible N% of other species.

	FO%				N%			
	NWI		SJI		NWI		SJI	
Species	2019	2020	2019	2020	2019	2020	2019	2020
Fish								
Demersal Fish								
Banded whiptail grenadier *Coelorinchus fasciatus*§	10.0	55.0		6.5		(19.6)		
Common hake *Merluccius hubbsi*‡			37.5	29.0			(11.0)	(20.1)
Dwarf codling *Notophycis marginata*§		20.0						
Red cod *Salilota australis*‡			12.5					
Kingclip *Genypterus blacodes*‡				6.5				
Benthopelagic Fish								
Hoki *Macruronus magellanicus*‡^	32.5	70.0	34.4	22.6	(13.2)	(28.9)	(10.2)	(9.1)
Southern blue whiting *Micromesistius australis*‡^		10.0						
Butterfish *Stromateus brasiliensis*§		5.0		9.7				
Pelagic Fish								
Fuegian sprat *Sprattus fuegensis*^	35.0	17.5		12.9	(29.7)	(9.0)		(10.8)
Unidentified Fish								
Fish spp.	5.0	7.5						
Crustacean								
Lobster krill *Munida gregaria*^	65.0	60.0	84.4	64.5	66.3	22.7	80.9	97.2
Themisto gaudichaudii^	5.0	7.5	6.2	12.9			(17.0)	(10.2)
Amphipod sp.		17.5				(6.0)		
Cephalopods								
Argentine shortfin squid (*Illex*) *Illex argentinus*‡^	10.0	5.0	59.4	38.7			(25.0)	(20.4)
Patagonian short-finned squid (Loligo) *Doryteuthis gahi*‡^	40.0	25.0	28.1	19.4	(41.2)	(16.4)	(24.1)	(8.6)
Squid spp.	5.0	5.0						

Octopus spp.		9.7
Jellyfish		
Medusa spp.^	5.0	22.6
Carrion		
Spheniscidae spp.^	5.0	12.9

‡ = fishery target species available as discards; § = bycatch species available as discards, ^ = prey naturally accessible to black-browed albatross during at least part of their life history.

4.3.2. Stable isotope analysis

Based on chick down feathers (**Table 4.3**), we found a significant difference in the isotopic niche of chicks from NWI and SJI in 2020, but not in 2019 (MANOVA; 2019: Pillai's Trace = 0.016, F(1,137) = 1.087, P = 0.340; 2020: Pillai's Trace = 0.125, F(1,108) = 7.675, P < 0.001; **Figure 4.5**), with respectively 88% and 75% point-estimate of overlap based on the maximum likelihood SEAc (standard ellipse area corrected for small sample size). *Post-hoc* ANOVAs showed that the difference was due to significantly higher δ15N at SJI in 2020 (δ15N: P < 0.001; δ13C: P = 0.049). As the difference of 0.29‰ in δ15N is relatively close to the limit of the machine reading precision of 0.17‰, some caution is warranted for interpretation. The ratio of C:N in down feathers was significantly lower in 2019 compared to in 2020 at both colonies (Mann-Whitney U tests; W = 1700, P < 0.001), but no difference was found between colonies (Mann-Whitney U tests; W = 7493, P = 0.656).

Table 4.3 Isotopic values (mean ± standard deviation) of black-browed albatross chick down feathers from New Island (NWI) and Steeple Jason Island (SJI) in 2019 and 2020.

Year	Colony	$\delta^{13}C$	$\delta^{15}N$	C:N	N(samples)
2019	NWI	-17.25 ± 0.34	15.53 ± 0.33	3.16 ± 0.03	73
2019	SJI	-17.22 ± 0.31	15.61 ± 0.31	3.16 ± 0.02	66
2020	NWI	-17.13 ± 0.27	15.51 ± 0.38	3.67 ± 0.16	55
2020	SJI	-17.04 ± 0.22	15.80 ± 0.38	3.63 ± 0.29	55

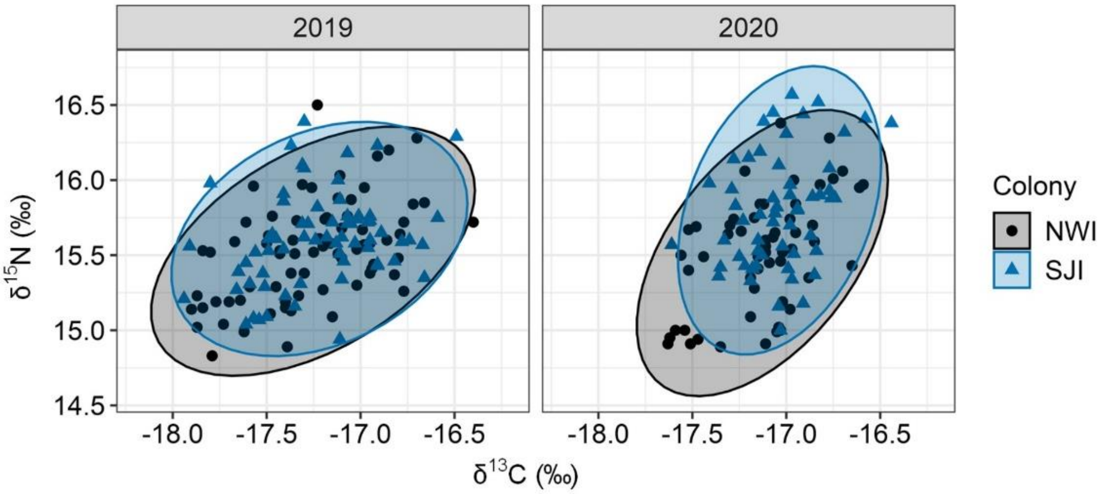


Figure 4.5 Isotopic niche of black-browed albatross chicks from New Island (NWI) and Steeple Jason Island (SJI) based on down feathers in 2019 and 2020 showing SIBER ellipses containing 95% of the data.

Mixing models estimated pelagic fish as the largest contributor to chick diet across both colonies and years (mean = 69–79%; **Figure 4.6**, **Figure 4.7**). Crustacean + jellyfish were the second most important diet source: 14–19% across years and colonies (**Figure 4.7**). Demersal fish, as well as benthopelagic fish + squid, were estimated to contribute relatively little to the diet (**Figure 4.7; Table S4.5**), although note that for SJI 2020, the model struggled to discern between the groups of benthopelagic fish + squid and crustacean + jellyfish (**Figure 4.6**). By summing the percentage estimates of prey groups that are categorised as entirely natural prey (pelagic fish; crustacean + jellyfish), we found that natural prey was the dominant diet for all colony/year combinations (mean minimum estimate 91.4–94.6%, excluding squid). Although the differences were relatively small, consumption of fishery discards was higher in 2020 than in 2019 at both colonies, and was higher at SJI than at NWI in both years (**Table 4.4**).

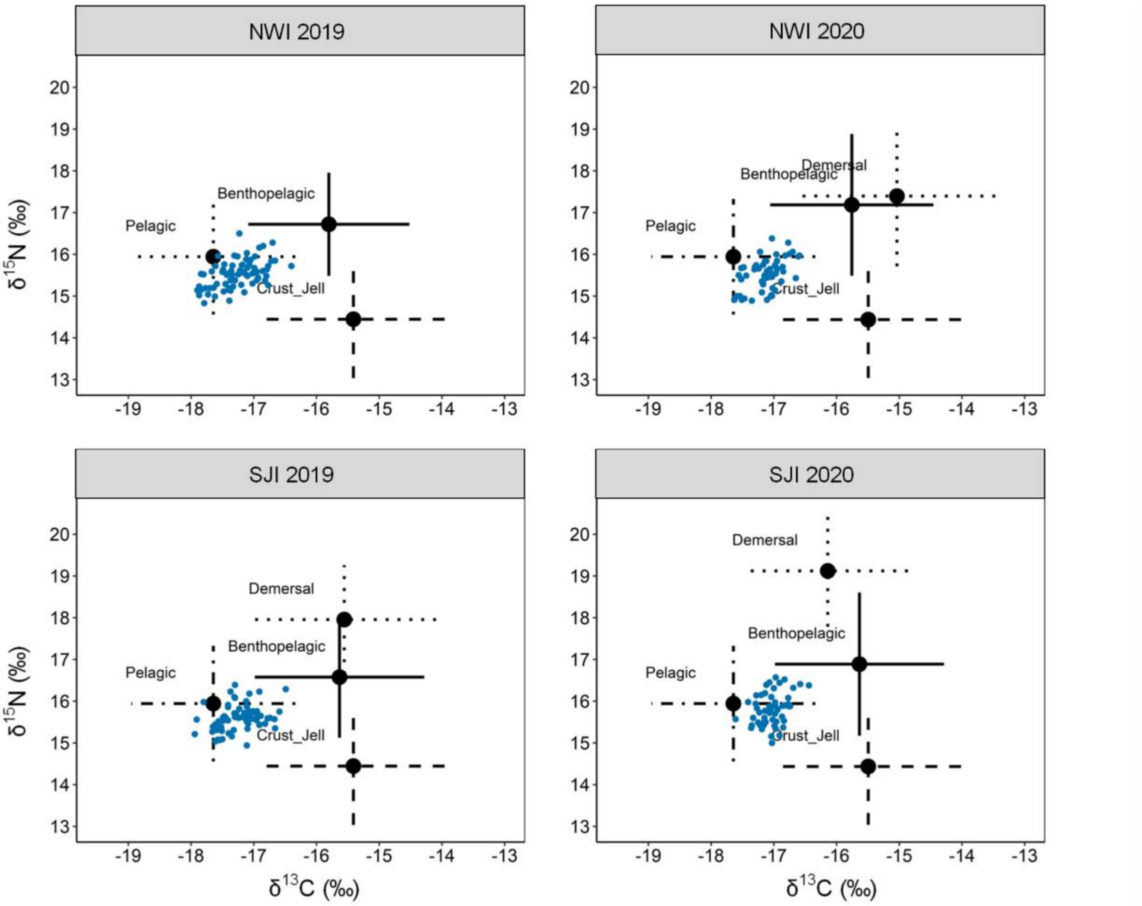


Figure 4.6 Isotopic values of black-browed albatross chick feathers (blue dots) from New Island (NWI) and Steeple Jason Island (SJI) in relation to four prey groups (pelagic fish, demersal fish, benthopelagic prey (fish + squid), crustacean + jellyfish (Crust_Jell)). Chick feather values are presented after correction for diet-tissue isotopic discrimination.

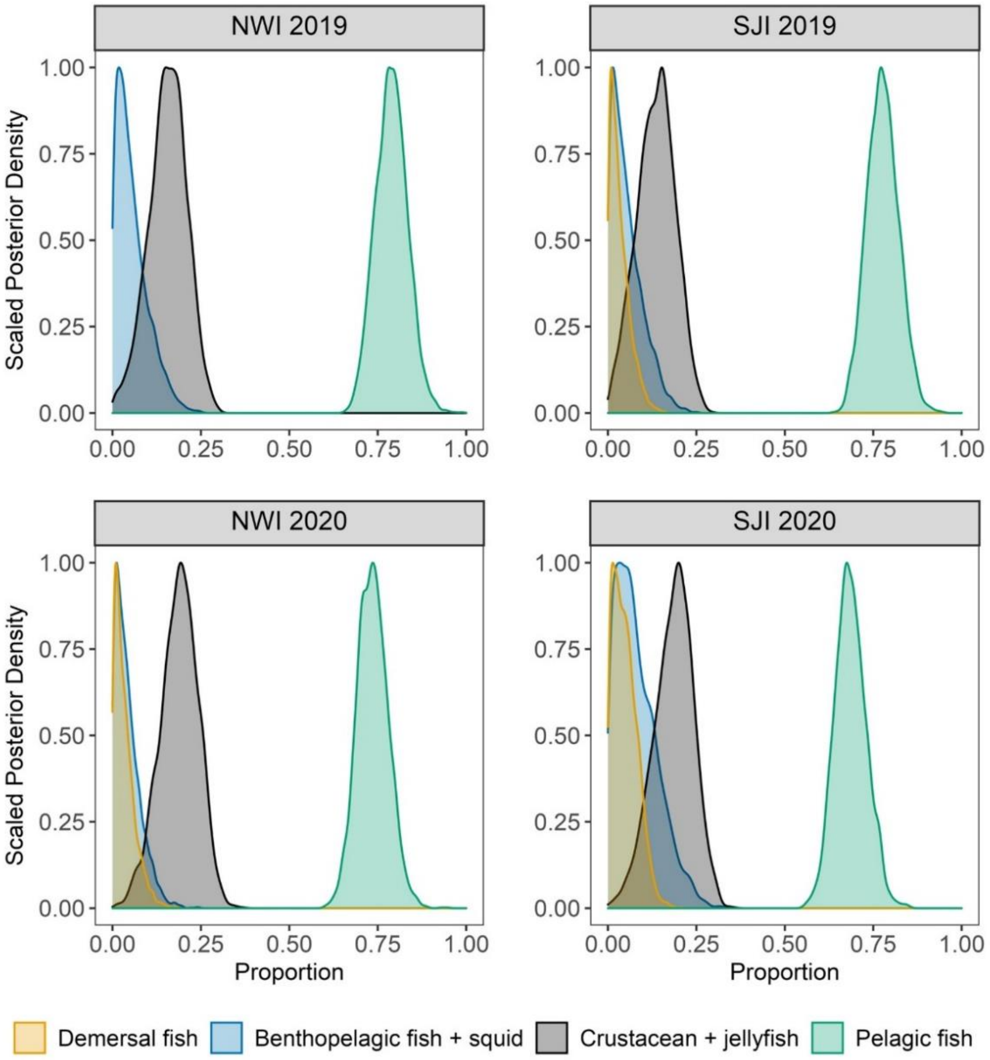


Figure 4.7 Posterior density estimates from MixSIAR models for the contribution of prey sources to the diet of black-browed albatross chicks for 2019 and 2020 at New Island (NWI) and Steeple Jason Island (SJI).

Table 4.4 Percentage contribution of discards and natural prey as estimated from stomach content analysis (SCA) and stable isotope analysis (SIA). Estimates from SIA for discards are the summed mean values of [demersal prey]–[demersal + (squid + benthopelagic prey)]; for natural prey, this represents the summed mean values for [pelagic fish + (crustacean + jellyfish)] – [pelagic fish + (crustacean + jellyfish) + (benthopelagic fish + squid)], as shown in Table S4.5.

	Discards		Natural prey	
	SCA (FO%; N%)	SIA (% contribution)	SCA (FO%; N%)	SIA (% contribution)
NWI 2019	27.5%; 4.2%	0.0–5.3%	92.5%; 95.8%	94.6–99.9%
SJI 2019	65.6%; 8.3%	3.3–8.6%	84.4%; 91.7%	91.4–96.7%
NWI 2020	82.5%; 55.7%	3.4–7.7%	77.5%; 44.3%	92.4–96.7%
SJI 2020	64.5%; 1.8%	4.6–13.2%	74.2%; 98.2%	86.7–95.3%

4.4. Discussion

A complementary approach to diet analyses can help minimise methodological bias, and therefore enable more robust inference (Le Bot et al. 2018). Using SCA and SIA to assess inter-colony and inter-annual variation in BBA diet, we found that chicks were primarily fed natural prey during mid-chick rearing, although dominant prey sources and species identified varied between methods and colonies. Whilst discards contributed less to the diet than natural prey regardless of sampling method, SCA provided higher discard use estimates compared with SIA. Our results have implications for fisheries management and future dietary studies assessing discard use.

4.4.1. Diet composition

Based on SCA, crustaceans, demersal fish and benthopelagic fish were the dominant prey, while SIA suggested that pelagic fish were most important. These results are consistent with SIA studies during early chick-rearing at NWI in 2011 (Granadeiro et al. 2014), and DNA metabarcoding of adults and chicks at NWI and SJI in 2014 and 2015 (McInnes et al. 2017a, b). Both SCA and SIA indicate that cephalopods play a relatively minor role in chick diet, as found previously at these colonies (Granadeiro et al. 2014 (SIA); McInnes et al. 2017b (DNA); Kuepfer et al. 2022a (SCA)), as well as at other BBA colonies (McInnes et al. 2017b).

Jellyfish were scarce in SCA but stable isotope mixing models suggest that jellyfish are more common than this (on average 14–19% across colonies/years). Nevertheless, this should be treated with caution given their isotopic similarity to crustaceans. However, previous work using DNA metabarcoding suggests that they are much more prevalent than crustaceans (SJI: up to 80% FO, 50% relative abundance (RA); NWI: up to 20% FO, 8% RA; McInnes et al. 2017b), further highlighting the importance of combining techniques.

SCA identified Fuegian sprat as the dominant pelagic fish, while, historically, southern blue whiting was more common in the diet of BBA (Thompson 1992). Increased Fuegian sprat consumption was also found in 2014 and 2015 (McInnes et al. 2017a) and likely reflects a shift in the ecosystem following the collapse of the southern blue whiting stock in 2004–2007 (Laptikhovsky et al. 2013). Unlike southern blue whiting, Fuegian sprat is not targeted by fisheries within the foraging range of NWI and SJI breeding albatross, and discards of this species are rare and patchy (Falkland Islands Government 2021).

Despite the close proximity of NWI and SJI (~ 75km), SCA revealed inter-colony diet differences, which appear to be stable over time (Thompson 1992; McInnes et al. 2017a). In particular, the dominant fish and squid were hoki and Loligo at NWI, but common hake and *Illex* at SJI. We cannot entirely exclude an influence of differences in timing between colony visits (NWI: January/February; SJI: March); for example, while hoki catches in 2019 and 2020 peaked in January and February, hake catches peaked in March (Ramos and Winter 2019, 2020; Winter and Ramos 2020). However, Thompson (1992) also found increased hake (and *Illex*) at SJI when sampled in January, compared to at NWI sampled in February/March. Our results may therefore also reflect differences in prey availability at colony-specific foraging sites (see Granadeiro et al. 2011). Indeed, hoki is most abundant in the south-west of the FCZ where NWI birds forage, whereas hake is the dominant catch in the north-west, where SJI birds feed (Catry et al. 2013b; Ramos and Winter 2019; Winter and Ramos 2020; Ventura et al. 2021a).

Finally, there were differences in trophic niche between colonies. The $\delta^{15}N$ values were higher at SJI, although this was only apparent in 2020. This result is suggestive of relatively increased consumption of higher trophic prey (e.g. hake or grenadier).

4.4.2. Relevance of fishery discards

As predicted, albatross chicks were overall fed primarily natural prey; nonetheless, chicks regularly received discards (based on FO%; **Table 4.4**). Previous studies have shown limited fishery interactions during early BBA chick-rearing (December – early January; Granadeiro et al. 2014) – a period of low fishing effort in Falkland Islands and Argentine waters (see also McInnes et al. 2017a). Some studies suggest discard consumption is more frequent during later chick-rearing (from February, McInnes et al. 2017a). Our study supports the high proportion of birds taking discards, but found natural prey to generally remain the main prey source.

Previous multi-colony research suggests higher discard consumption at SJI compared to NWI (Thompson 1992; McInnes et al. 2017a). However, based on SCA, we found that while discard consumption was higher at SJI in 2019, the reverse was the case in 2020. Stable isotope mixing model results, which are temporally more comparable, were inconclusive in this regard, because the models cannot distinguish between benthopelagic squid (natural prey) and benthopelagic fish (discards). However, excluding the confounding prey source, discards were higher at SJI, particularly in 2020, which agrees with the increased trophic niche discussed above.

Discard consumption was higher in 2020, although the extent of this varied between methods. Breeding BBA from the Falkland Islands appear to increase discard consumption during periods of poor natural foraging conditions (Kuepfer et al. 2022a). Indeed, the present study provides some indication of potentially increased nutritional stress in 2020: Prey diversity was higher in 2020, which could result from birds compensating for the scarcity of preferred prey by targeting alternative species (Quillfeldt et al. 2010; van Donk et al. 2017). Significantly higher C:N ratios in 2020 could suggest prey being of inferior protein quality (Robbins et al. 2005). Further, meals contained more stomach oil in 2020, which could be an effect of prolonged foraging trips (supported also by GPS tracking from P. Catry, unpubl. data; also see Warham et al. 1976; Chaurand and Weimerskirch 1994; Connan et al. 2005). The season 2020 also saw reduced chick weight at NWI (on average -7.3%), and reduced breeding success at both NWI (-22%) and SJI (-25%) (Crofts and Stanworth 2021; Kuepfer et al. 2022a). Combined, these indicators support the theory of increased discard consumption during periods of increased nutritional stress in Falkland Islands BBA.

4.4.3. SCA and SIA – a critical evaluation

Our key objectives were to quantify diet (particularly in the context of fisheries), and test for inter-annual and colony-specific differences. Combining SCA with SIA provided us with complementary information and helped with interpretation of results in several aspects (**Table 4.5**).

In contrast to SIA, SCA supplied high taxonomic and morphological details, including information on prey size (age class) and visual confirmation of processed waste from fisheries. This assisted with categorisation and quantification of prey as discards or natural prey. It also enabled us to distinguish between isotopically similar (but from a management perspective, very different) prey (here squid vs benthopelagic fish; crustacean vs jellyfish – although see below). This can be important to complement SIA; if our main prey source was found to be benthopelagic prey, SCA could have helped us determine whether this was dominated by discards (benthopelagic fish) or natural prey (squid). In our case, this confounding prey source only contributed a small amount to the overall diet source of BBA chicks, and therefore bears little influence on the overall result regarding the dominant prey origin.

SCA was also important for selecting mixing model inputs, although our results emphasise that using informative priors from conventional methods in stable isotope mixing models can transfer methodological biases leading to erroneous results (Swan et al. 2020). For example, compared to SCA, SIA suggests the importance of pelagic fish, but more moderate contribution from crustaceans. This difference may reflect differential digestion of prey. While fragile prey like sprat or jellyfish are more easily digested, and hence likely underestimated in SCA, the opposite is true for large or rigid organisms like demersal fish or crustaceans.

We note that differences between methods may in part also be related to short-term differences in the temporal window they represent, particularly for SJI where down growth and stomach content sampling did not overlap (see also e.g. McCluskey et al. 2021). However, this does not appear to be the case at NWI, where pelagic prey also dominated in both years based on SIA of plasma (reflective of the diet from the previous few days (Phillips et al. 2014)) collected in February 2019 and 2020 (A. Kuepfer unpubl. data).

A further potential issue of using SCA in the absence of a complementary method is the large number of ways this type of diet information can be quantified (e.g. FO%, N%, IRI%). These each have their own biases (see Barrett et al. 2007) and care must be taken when selecting a suitable metric to use based on the study objectives. In our case, a multitude of SCA metrics allowed us to draw more comprehensive conclusions in relation to discard consumption, and also gain insight into the proportion of breeders that interact with vessels (although see e.g. Granadeiro et al. 2011, 2014). Meanwhile, SIA provided a cross-check for prey source quantification by providing a single, integrated signal representing multiple meals.

Finally, our SCA was subject to confounding temporal effects due to non-simultaneous sampling. Previous SCA and DNA studies at these colonies suffered from the same limitation (Thompson 1992; McInnes et al. 2017a). The complementary use of feather SIA in the present study assisted with inter-colony comparison due to its more long-term dietary signal. This aspect also makes SIA a suitable candidate for studying diet during the non-breeding period (see e.g. Bugoni et al. 2010; Mariano-Jelicich et al. 2013; Granadeiro et al. 2014), although care must be taken to obtain a meaningful prey-field in time and space (Quillfeldt et al. 2015).

Table 4.5 Advantages and caveats of stomach content analysis (SCA) and stable isotope analysis (SIA) for assessing seabird diet and discard use.

Advantages	SCA	SIA	Caveats
High taxonomic resolution	✓		
Information on meal size	✓		
Information on prey size / age class	✓		
Visual cues of processing waste	✓		
Distinguish isotopically similar prey	✓		Only necessary in ecosystems where key prey are isotopically similar
Informs stable isotope mixing models	✓		Information is biased by differential digestion rate
Provides a single, integrated dietary signal		✓	
Larger temporal window		✓	Different tissues provide different temporal windows (see e.g. Inger and Bearhop 2008; Phillips et al. 2014)
Independent of digestion rate		✓	

4.4.4. Seabird diet in the context of fisheries management

Our dietary results highlight important aspects in terms of fisheries management. Firstly, natural prey are important for BBA chicks, underlining the importance of understanding pelagic prey distribution and abundance in the Falkland Islands (Kuepfer et al. 2022a) and elsewhere (Barrett et al. 2007; Boldt et al. 2022). Secondly, while discards do not contribute substantially to chick diets, a high proportion of adults deliver discards to their young thus incurring a risk of bycatch. Therefore, we echo previous studies that any management actions aiming to reduce discard availabiliy, and therefore bycatch risk (e.g. batch discharging as currently implemented in the Falkland Islands trawl fleet), will be of conservation benefit during this period (e.g. Granadeiro et al. 2011, 2014; Kuepfer et al. 2022b). However, the wider implications of discard reductions during non-breeding, and effects this may have on the survival of juvenile and immature birds, remains unknown.

Going forward, seabird diet provides a helpful fishery and ecosystem monitoring tool but requires appropriate methodological consideration. Given climatic changes and continued fishery presence across the Patagonian Shelf (and elsewhere), we recommend the use of stable isotopes to capture broader time-scales, but in combination with SCA to allow continued prey species identification in a potentally changing ecosystem. We also recommend multiple SCA metrics be considered (N%, FO%, M%, IRI%), as these can provide complementary information on important prey and the extent of seabird-fishery interaction.

4.5. Supplementary information

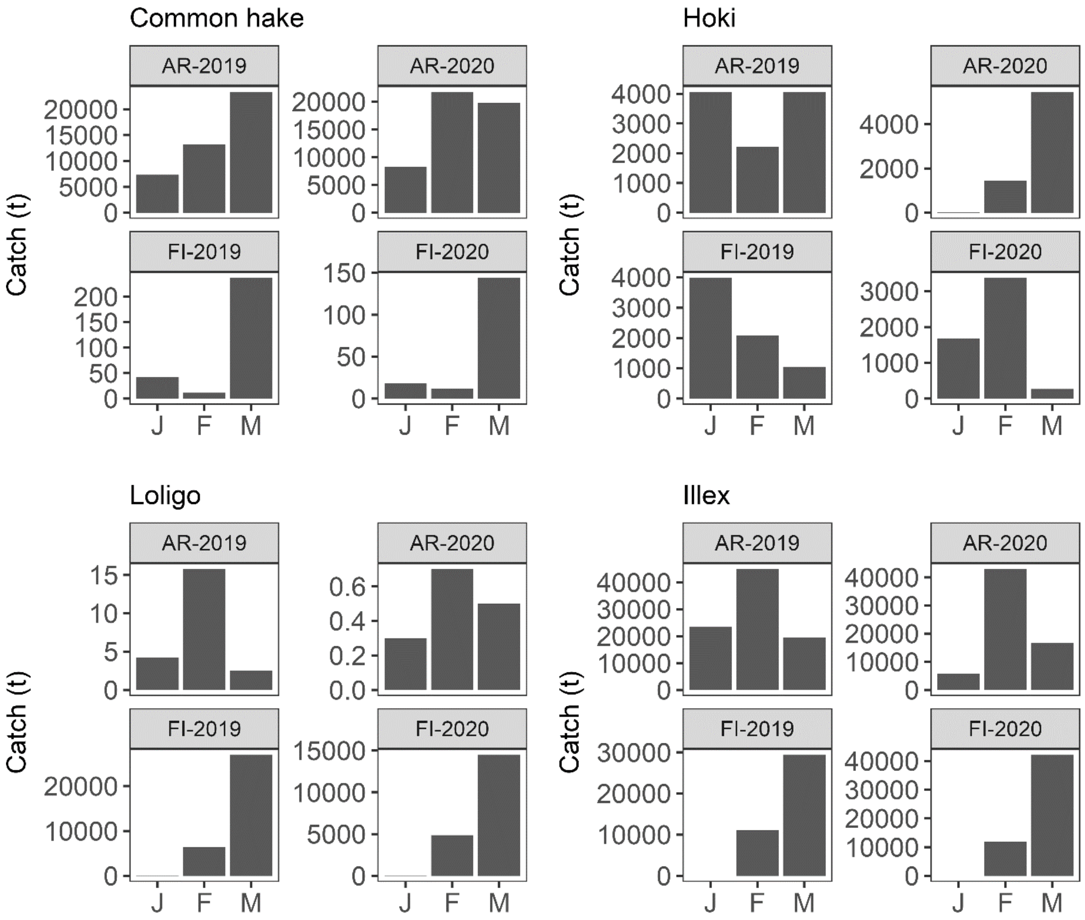


Figure S4.1 Catch statistics of key commercial species in Argentina (AR) and Falkland Islands (FI) in January (J), February (F), and March (M) for the years 2019 and 2020. Note the difference in scale between plots.

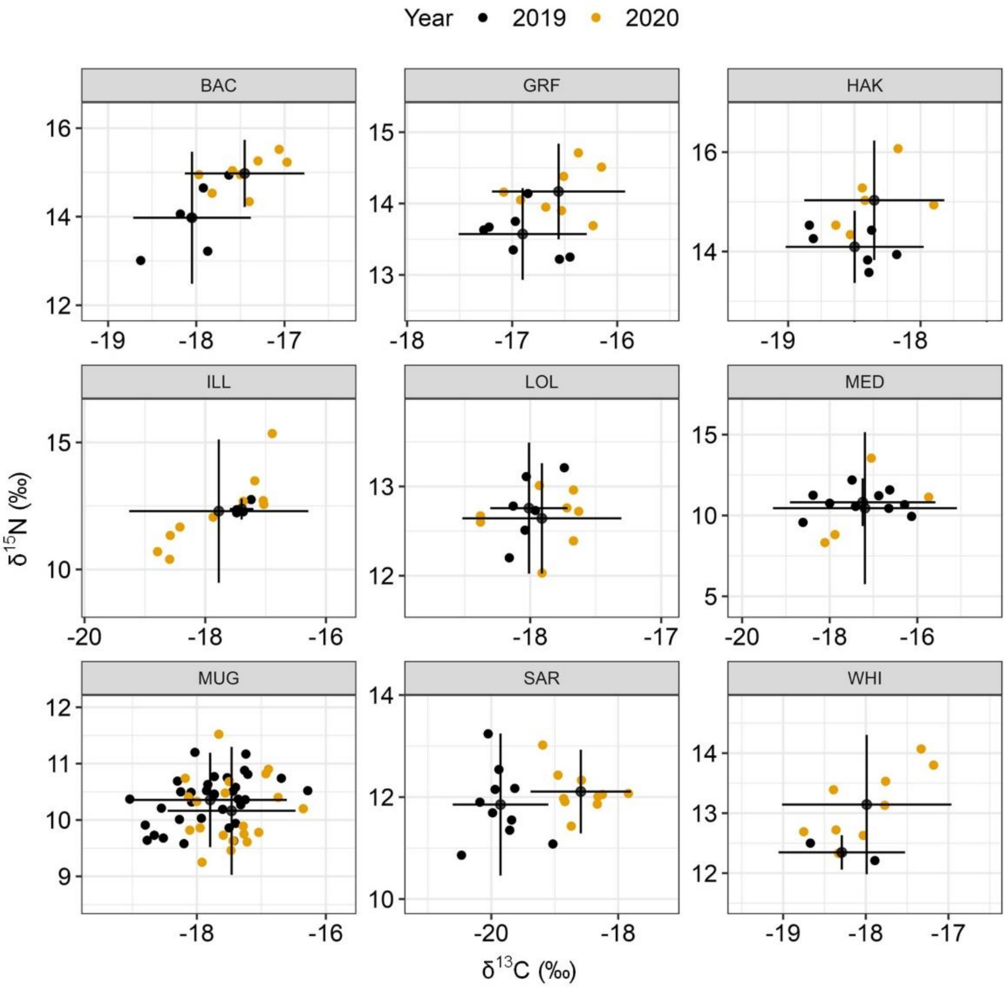


Figure S4.2 Inter-annual comparison of isotopic values of prey species used in our study where multiple year of data were available. BAC = *Salilota australis*; GRF = *Coelorinchus fasciatus*; HAK = *Merluccius hubbsi*; ILL = *Illex argentinus*; LOL = *Doryteuthis gahi*; MED = *Medusa* spp.; MUG = *Munida gregaria*; SAR = *Sprattus fuegensis*; WHI = *Macruronus magellanicus*.

Table S4.1 Species and their corresponding habitats. ** = non-commercial species known to be naturally accessible to black-browed albatrosses during at least part of their life-cycle / diel migration; * = commercially targeted species that are naturally accessible to black-browed albatrosses during at least part of their life-cycle / diel migration; ‡ = commercially targeted species that are not naturally accessible to black-browed albatrosses. ^{x} = non-commercial species that are not naturally accessible to black-browed albatrosses but that are readily made available through discards.

Species	Common name	Habitat	Reference
Fish			
Coelorinchus fasciatus^{x}	Banded whiptail	Bentho-demersal	Lee et al. 2019
Stromateus brasiliensis^{x}	Butterfish	Demersal-pelagic	Ramilo-Fernández and Sotelo 2020
*Gymnoscopelus nicholsi***	Myctophid sp.	Mesopelagic	Klemmedson et al. 2020
Salilota australis‡	Red cod	Demersal	Agnew 2002
Notophycis marginata^{x}	Dwarf codling	Demersal	Zapata-Hernández et al. 2014
Merluccius hubbsi‡	Common hake	Demersal (pelagic in first year)	Costa et al. 2019
Patagonotothen ramsayi‡	Patagonian rock cod	Benthopelagic	Laptikhovsky et al. 2013
*Micromesistius australis**	Southern blue whiting	Benthopelagic / pelagic / mesopelagic	Agnew 2002
*Macruronus magellanicus**	Hoki	Benthopelagic	Riccialdelli et al. 2013
*Sprattus fuegensis***	Fuegian sprat	Pelagic	Agnew 2002
Cephalopod			
*Illex argentinus**	Argentine shortfin squid	Benthopelagic/ pelagic	Agnew 2002; Riccialdelli et al. 2013
*Doryteuthis gahi**	Loligo	Demersal / benthopelagic	Agnew 2002; Riccialdelli et al. 2013
*Moroteuthopsis ingens***	Greater hooked squid	Pelagic	Agnew 2002
Crustacean			
*Munidae gregaria***	Lobster krill	Benthopelagic / pelagic	Agnew 2002
*Themisto gaudichaudii***		Pelagic	Padovani et al. 2012
Peltarion spinulosum^{x}	Little purple-back crab	Benthic	Gorny 1999
Gelatinous zooplankton			

Medusa spp.**		Pelagic	Diez et al. 2012, 2016a, b; Schiariti et al. 2021
Beroe spp.**		Pelagic	Schiariti et al. 2021; pers. obs
Carrion			
Spheniscidae spp.**	Penguins	Marine	Jouventin and Dobson 2018

Table S4.2 Allometric formulae used for size-class reconstruction based on sagittal otoliths length (OL). TL = total length (mm), PAL = preanal length (mm), ML = mantel length (mm).

Species	Formula	Reference
Macruronus magellanicus	TL:OL = 35:1; TL = PAL *0.43	Arata and Xavier 2003 FIFD unpubl. data; Chong et al. 2007
Micromesistius australis	lnTL=-0.259+1.47*(lnOL)	Thompson 1992

Table S4.3 Summary of the multiple metrics used for quantifying key prey groups extracted from stomach contents. NWI = New Island; SJI = Steeple Jason Island. FO% = percentage frequency of occurrence, M% = percentage mass, N% = percentage number, IRI% = percentage index of relative importance (see main text).

Colony	Year	Prey	FO%	M%	N%	IRI%
NWI	2019	Carrion	0.0	0.0	0.0	0.00
NWI	2019	Cephalopods	50.0	12.0	17.0	12.02
NWI	2019	Crustacean	70.0	17.0	67.0	48.76
NWI	2019	Fish	72.5	49.3	15.9	39.22
NWI	2019	Jellyfish	0.0	0.0	0.0	0.00
NWI	2020	Carrion	7.5	0.1	0.6	0.03
NWI	2020	Cephalopods	32.5	8.4	14.1	4.45
NWI	2020	Crustacean	72.5	3.1	30.7	14.95
NWI	2020	Fish	95.0	84.9	54.1	80.55
NWI	2020	Jellyfish	5.0	0.1	0.4	0.02
SJI	2019	Carrion	0.0	0.0	0.0	0.00
SJI	2019	Cephalopods	65.6	18.5	9.6	15.27
SJI	2019	Crustacean	84.4	11.7	84.1	66.96
SJI	2019	Fish	68.8	24.9	6.3	17.77
SJI	2019	Jellyfish	0.0	0.0	0.0	0.00
SJI	2020	Carrion	25.8	1.8	0.1	0.36
SJI	2020	Cephalopods	54.8	7.7	0.9	3.56
SJI	2020	Crustacean	67.7	21.3	97.5	60.55
SJI	2020	Fish	67.7	67.9	1.4	35.29
SJI	2020	Jellyfish	22.6	1.3	0.1	0.24

Table S4.4 Summary of species used as prey sources in individual mixing models.

Model	Species	Sampling year	n	δ^{15}N (mean ± standard deviation)	δ^{13}C (mean ± standard deviation)	Aggregated group source
NWI 2019	*Doryteuthis gahi*	2019 + 2020	14	12.69 ± 0.33	-17.95 ± 0.25	Benthopelagic fish + squid
	Medusa spp.	2019 + 2020	14	10.71 ± 1.33	-17.23 ± 0.89	Crustacean + jellyfish
	Munida gregaria	2019 + 2020	56	10.27 ± 0.51	-17.72 ± 0.68	Crustacean + jellyfish
	Sprattus fuegensis	2019	10	11.85 ± 0.71	-19.86 ± 0.38	Pelagic fish
	Macruronus magellanicus	2019	3	12.35 ± 0.15	-18.29 ± 0.39	Benthopelagic fish + squid
SJI 2019	*Salilota australis*	2019	6	13.98 ± 0.76	-18.05 ± 0.34	Demersal fish
	Coelorinchus fasciatus	2019	7	13.57 ± 0.33	-16.9 ± 0.31	Demersal fish
	Merluccius hubbsi	2019	6	14.1 ± 0.37	-18.5 ± 0.27	Demersal fish
	Illex argentinus	2019 + 2020	15	12.32 ± 1.16	-17.65 ± 0.64	Benthopelagic fish + squid
	Doryteuthis gahi	2019 + 2020	14	12.69 ± 0.33	-17.95 ± 0.25	Benthopelagic fish + squid
	Medusa spp.	2019 + 2020	14	10.71 ± 1.33	-17.23 ± 0.89	Crustacean + jellyfish
	Munida gregaria	2019 + 2020	56	10.27 ± 0.51	-17.72 ± 0.68	Crustacean + jellyfish
	Sprattus fuegensis	2019		11.85 ± 0.71	-19.86 ± 0.38	Pelagic fish
	Macruronus magellanicus	2019	3	12.35 ± 0.15	-18.29 ± 0.39	Benthopelagic fish + squid
NWI 2020	*Themisto gaudichaudii*	2020	4	10.32 ± 0.24	-18.77 ± 0.44	Crustacean + jellyfish
	Beroe spp.	2020	1	9.8	-19.28	Crustacean + jellyfish
	Coelorinchus fasciatus	2020	8	14.17 ± 0.34	-16.56 ± 0.32	Demersal fish
	Doryteuthis gahi	2019 + 2020	14	12.69 ± 0.33	-17.95 ± 0.25	Benthopelagic fish + squid
	Medusa spp.	2019 + 2020	14	10.71 ± 1.33	-17.23 ± 0.89	Crustacean + jellyfish
	Munida gregaria	2019 + 2020	56	10.27 ± 0.51	-17.72 ± 0.68	Crustacean + jellyfish
	Notophycis marginata	2019		11.92 ± 0.42	-18.35 ± 0.2	Demersal fish
	Sprattus fuegensis	2019	10	12.11 ± 0.42	-18.59 ± 0.4	Pelagic fish
	Macruronus magellanicus	2020	10	13.66 ± 1.73	-17.99 ± 0.49	Benthopelagic fish + squid
SJI 2020	*Themisto gaudichaudii*	2020	4	10.32 ± 0.24	-18.77 ± 0.44	Crustacean + jellyfish

	Beroe spp.	2020	1	9.8	-19.28	Crustacean + jellyfish
	Merluccius hubbsi	2020	6	15.03 ± 0.61	-18.35 ± 0.27	Demersal fish
	Illex argentinus	2019 + 2020	15	12.32 ± 1.16	-17.65 ± 0.64	Benthopelagic fish + squid
	Doryteuthis gahi	2019 + 2020	14	12.69 ± 0.33	-17.95 ± 0.25	Benthopelagic fish + squid
	Medusa spp.	2019 + 2020	14	10.71 ± 1.33	-17.23 ± 0.89	Crustacean & jellyfish
	Munida gregaria	2019 + 2020	56	10.27 ± 0.51	-17.72 ± 0.68	Crustacean & jellyfish
	Sprattus fuegensis	2019	10	12.11 ± 0.42	-18.59 ± 0.4	Pelagic fish
	Macruronus magellanicus	2020	10	13.66 ± 1.73	-17.99 ± 0.49	Benthopelagic fish & squid

Table S4.5 Mean percentage prey source contribution (with 95% confidence intervals) to the diet of black-browed albatross chicks from New Island (NWI) and Steeple Jason Island (SJI) as predicted by MixSIAR mixing models.

Model	Pelagic fish	Crustacean + jellyfish	Benthopelagic fish + squid	Demersal fish
NWI 2019	79.1% (72–86.7%)	15.5% (6.9–23.5%)	5.3% (0.3–13.6%)	-
SJI 2019	77.6% (70.5–84.9%)	13.8% (4.5–22.3%)	5.3% (0.4–14%)	3.3% (0.3–8.6%)
NWI 2020	73.8% (66.6–81.2%)	18.6% (9.2–27.6%)	4.3% (0.3–11.2%)	3.4% (0.2–9.2%)
SJI 2020	68.6% (61.8–76%)	18.1% (7.4–26.8%)	8.6% (0.8–19.9%)	4.6% (0.4–10.9%)

Chapter 5 – Limited overlap between central-place foraging black-browed albatross and fisheries on the Patagonian Shelf

This chapter has been submitted to Marine Ecology Progress Series as:

Amanda Kuepfer, José P. Granadeiro, Francesco Ventura, Stephen C. Votier, Richard B. Sherley, Paulo Catry (in review) Limited overlap between central-place foraging black-browed albatross and fisheries on the Patagonian Shelf.

Abstract

Understanding marine predator distribution in relation to fisheries is integral for sustainable marine management. The black-browed albatross *Thalassarche melanophris* is an abundant seabird of the Patagonian Shelf Large Marine Ecosystem, and the most common seabird incidentally caught by local longline and trawl fisheries. However, the extent to which they overlap with fisheries while foraging during different phases of the breeding cycle is poorly described. Here we use multi-annual GPS-tracking from three black-browed albatross colonies in the Falkland Islands, together with gear-specific fisheries distribution data, to (1) identify primary foraging areas used during incubation and early chick-rearing (brooding), (2) quantify overlap with fishing activities, and (3) identify areas and gear types where fisheries exposure is highest. Foraging areas were almost entirely within the national jurisdictions of the Falkland Islands and Argentina (99% of raw foraging fixes and core utilisation distribution). They spanned large areas to the north and west during incubation (up to ~40°S; 60°W), and slightly smaller areas south-west during brooding (~55–52°S; 55°W). Fisheries overlap was relatively low and similar during both breeding stages (Incubation: Bhattacharyya’s affinity BA = 0.39 ± 0.10; Brood: BA = 0.37 ± 0.05). Overlap was greatest with Falklands and Argentine bottom and mid-water trawl fleets (Incubation: BA = 0.36 ± 0.13; Brood: BA = 0.37 ± 0.08). While it remains unclear whether the overlap represents vessel attraction or use of shared locations, our results highlight key areas and fleets where limited management resources could provide the highest benefits across and beyond national jurisdictions.

5.1. Introduction

Fisheries can have profound ecological impacts, such as through habitat destruction, stock collapse, competition with piscivorous animals, and incidental catch of non-target species (Pikitch et al. 2004; Skewgar et al. 2007; Cury et al. 2011). Concerns over the sustainability of fisheries has encouraged a shift from single-stock management to an ecosystem-based approach (Garcia et al. 2003; Pikitch et al. 2004; Crowder et al. 2008; FAO 2022), which integrates possible impacts on marine predators. Seabirds are especially valuable in this context as they can convey a range of information on ecosystem health and functioning (Wing et al. 2014; Estes et al. 2016; Velarde et al. 2019). Further, many seabird populations are also directly threatened by fisheries impacts (Dias et al. 2019) highlighting their importance in fisheries management (Fishpool and Evans 2001; Lascelles et al. 2016; Beal et al. 2021a).

Seabird foraging distribution is driven by complex interactions of physical and biological factors such as upwellings and ocean fronts (Scales et al. 2014; Grecian et al. 2016; Serratosa et al. 2020), prey availability (Xavier et al. 2013), and also varying by seabird life-history stages (Riotte-Lambert and Weimerskirch 2013; Votier et al. 2017). Fisheries, too, can affect bird distributions (Votier et al. 2010; Collet et al. 2015; Blanco et al. 2017). During the breeding season, seabird movements are constrained by central-place foraging and increased intra- and inter-specific competition (Wakefield et al. 2013; Oppel et al. 2018). Accordingly, knowledge of, and protection for, foraging areas during breeding is important (Oppel et al. 2018).

The world's largest black-browed albatross population (*Thalassarche melanophris*), which breeds in the Falkland Islands (>70% of global population; Wolfaardt, 2013), forages year-round within the Southwest Atlantic over the Patagonian Shelf and the southern Brazil Shelf (Grémillet et al. 2000; Huin 2002; Catry et al. 2013b; Copello et al. 2013; Ponchon et al. 2019; Paz et al. 2021). The Patagonian Shelf spans over 1 million km^2, encompassing the Uruguayan, Argentinian and Falkland Islands continental shelf and is characterised by numerous upwellings and frontal systems providing nutrient-rich conditions which favour high biodiversity and commercially valuable fisheries (Croxall and Wood 2002; **Figure 5.1**). Due to their high numbers (>1 million), and role as important consumers, black-browed albatross are sentinels for the region's environmental health (Ventura et al. 2021a). However, the species is also the most commonly bycaught seabird in the area (80–93% within the FCZs) (González-Zevallos et al. 2007; Tamini et al. 2015; Kuepfer et al. 2018, 2022b; Jiménez et al. 2022). As such, the black-browed albatross forms a focal species for ecosystem-based management of the Patagonian Shelf Large Marine Ecosystem, both in terms of biological conservation and ecosystem functioning.

When not breeding, black-browed albatross occupy the Patagonian Shelf north and south of the Falkland Islands (GLS technology: Grémillet et al. 2000; Ponchon et al. 2019; satallite tracking: Copello et al. 2016). They use similar areas during incubation (GLS: Grémillet et al. 2000; Furtado et al. 2020; satellite tracking: Huin 2002) but forage closer to their colonies and off Staten Island during chick-rearing (Huin 2002; Granadeiro et al. 2011). However, besides inter-seasonal variation, albatross foraging patterns also vary among colonies and by year (Granadeiro et al. 2011).

Despite their well-known scavenging behaviour, fisheries have a weak effect on the distribution of brooding black-browed albatrosses (Granadeiro et al. 2011; Catry et al. 2013b). Dietary studies suggest that vessel interactions may be higher during incubation than brooding (McInnes et al. 2017a), and can vary among years (Kuepfer et al. 2022a). However, there are no comparable tracking studies on intra- and inter-annual variation of albatross-fishery interactions. Furthermore, the spatio-temporal overlap of Falklands' breeding albatross with fisheries has never been assessed beyond Falkland Islands waters.

Here we use multi-annual GPS tracking from breeding black-browed albatross across three colonies that represent over 70% of the Falkland Islands population (and c. 50% of the world population of the species) and cover their entire Falklands breeding range in a northwest-southeast axis. Specifically, we (1) identify primary foraging areas used during incubation and early chick-rearing (brooding), (2) quantify overlap with fishing activities, and (3) identify areas and gear type where fisheries exposure is highest.

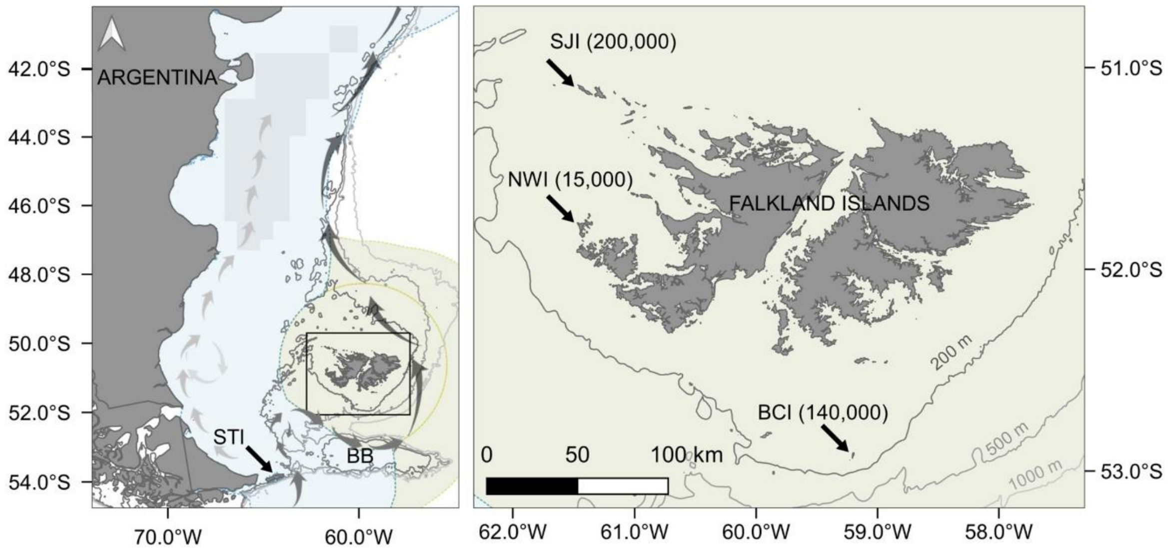


Figure 5.1 Left: Patagonian Shelf Large Marine Ecosystem. The Argentine Exclusive Economic Zone and the Falklands Conservation Zones are shaded in blue and green, respectively. The location of Staten Island (STI) and the Burdwood Bank (BB) are indicated, and the 200-m (shelf break), 500-m and 1000-m depth contours are shown as a grey lines. Northwards flowing arrows indicate the Patagonian Current Front (light grey, west) and the stronger Falklands/Malvinas Current Front (dark grey, east). Adapted from Sabatini et al. (2004). Grey shaded area off Argentina represents the approximate outline of the dynamic hake fishing closure zone (Tamini et al. 2021). Right: Location of the study colonies in the Falkland Islands: Steeple Jason Island (SJI), New Island (NWI), and Beauchêne Island (BCI), with respective number of breeding pairs shown in brackets.

5.2. Materials and methods

5.2.1. Study area

Below 40°S latitude (our area of interest), bottom trawlers target a variety of finfish species year-round, particularly Argentine hake *Merluccius hubbsi* (~33% of total catch) and hoki *Macruronus magellanicus* (~ 7% of total catch) (Seco Pon et al. 2018). In austral spring, ice trawlers operating north of 48°S in Argentinian waters also target pelagic Argentine anchovy *Engraulis anchoita* and chub mackerel *Scomber japonicus* (~2% of total catch; Seco Pon et al. 2018) and an Argentine red shrimp *Pleoticus muelleri* trawl fleet operates inside a hake fishing closure area off San Jose Gulf (e.g. Tamini et al. 2021; **Figure 5.1**). In addition, a large jigging fleet targets Argentine shortfin squid *Illex argentines* (~23% of total catch), representing one of the largest squid fisheries globally (Seco Pon et al. 2018; Harte et al. 2019). Longlining effort on the Patagonian Shelf has declined substantially over the 20 years, but a small fleet of demersal longliners, predominantly targeting Patagonian toothfish (*Dissostichus eleginoides*), continues to operate throughout the year within the Falklands Conservation Zones (FCZs) and the Argentine Exclusive Economic Zone (AEEZ) (**Table S5.1**).

5.2.2. Study colonies

Fieldwork was conducted in the Falkland Islands at three black-browed albatross colonies: (1) New Island (NWI, 51°43′S, 61°18′W); (2) Steeple Jason Island (SJI, 51°01′S, 61°13′W); and (3) Beauchêne Island (BCI, 52°53′S 59°12′W) (**Figure 5.1**). SJI and BCI respectively hold the world's largest and second largest colonies, while the colony at NWI is much smaller (**Figure 5.1**; Wolfaardt, 2013). Long-term data from NWI (2003–2022) suggests that mean egg-laying is ~10 October, chicks mostly hatch for two weeks around ~18 December, and fledging occurs late April to early May.

5.2.3. GPS tracking

Breeding birds were equipped with GPS loggers at all three colonies during incubation (early October – mid-December) and brood-guard (mid-December – early January) in one or more seasons from 2008 to 2019 (**Table S5.2**). The breeding stage allocated to individually tracked birds refers to the breeding stage of individual birds at the point of GPS deployment.

GPS loggers were attached to the scapular feathers or to the four central tail feathers using Tesa® tape and retrieved when the tagged bird had undertaken at least one foraging trip. Bird handling time took generally <10 and <5 min for deployment and retrieval, respectively. GPS loggers weighed between 15 and 30 g, depending on battery size and model used (**Table S5.3**), which represents <1% of the bird's body weight. There was no evidence that logger deployment affected foraging behaviour or success (Phillips et al. 2003; Granadeiro et al. 2014; Campioni et al. 2017; Ventura et al. 2021a). All work was conducted under licences from the Falkland Islands Government.

Because data were collected for various studies, and due to logistical constraints, tracking could not be conducted simultaneously from all colonies in all years, and the available data vary in terms of sample size and temporal resolution (1 s – 60 min, Table S5.3). Trips from brooding birds were collected at <30 min (mean ± sd = 9.6 ± 4.4 min) intervals, with 76% of trips sampled at ≤14 min intervals. Incubation trips were recorded at 14–60 min (28.9 ± 17.0 min) intervals, with 78% of trips sampled at ≤30 min. Sampling rates were chosen based on available knowledge of approximate trip duration and GPS battery life, and to maximise the chances of trips being recorded in their entirety (see e.g. Campioni et al. 2017).

5.2.4. Data analysis

The following steps were undertaken to estimate foraging distributions and overlap with fisheries: (1) cleaning GPS data; (2) interpolating locations to a common time interval and identifying foraging through Hidden Markov Models (HMM); (3) estimating utilisation distributions (UDs) from foraging locations, and assessing the population-level representativeness for each unique combination of year, breeding stage and colony (hereafter referred to as “data group”); (4) estimating fishing UDs from Global Fishing Watch (GFW; https://globalfishingwatch.org/) data, and calculating seabird-fisheries overlap. All data processing and analyses were conducted in R. 4.0.0 (R Core Team 2021). Bathymetry data used for mapping were sourced from https://download.gebco.net/, at a 15 arcsecond resolution.

Step 1: Cleaning GPS data

GPS loggers used had a mean location error of <10 m (Campioni et al. 2017; Morris and Conner 2017). Erroneous locations (i.e. outliers indicating unrealistic ground speeds of >100 $km.h^{-1}$; Wakefield et al. 2009), and locations fixed over land (the breeding colonies) were removed. We also removed consecutive fixes within 2.5 km from respective colonies at the start or end of a trip, as these are likely associated with colony-based behaviour (e.g. rafting; bathing Granadeiro et al. 2018). Based on complete trips only, we calculated the following foraging trip metrics: (1) maximum distance travelled from the colony (km); (2) cumulative distance travelled between all locations (km), and (3) trip duration (days). Incomplete trips did not provide the certainty that the birds had necessarily reached their terminal position (furthest location from the colony) at the point the trips ended. Thus to avoid potential bias, we did not include these trips (22% of all trips) when calculating trip metrics. All remaining analyses utilised the entire dataset to avoid biasing against longer trips, which were more commonly incomplete than shorter trips.

Step 2: Data interpolation and identifying foraging locations

To avoid bias caused by mixed GPS-tracking temporal resolutions, and to allow computation of behavioural states, tracks were either resampled or linearly interpolated to a constant interval of 10 minutes using the *redisltraj* function in the *adehabitatLT* R package (Calenge 2006). These are accepted approaches used for standardising the temporal resolutions of different devices or duty cycles (e.g. Grémillet et al. 2004; Wakefield et al. 2017), and have been successfully applied to a subset of the data used here (Granadeiro et al. 2011, 2014, 2018). We did not interpolate across time gaps >120 min; therefore, after interpolation, each unique track consisted of one or more segments comprising locations of equal time intervals.

Three-state HMM were fitted to interpolated trip segments within the *moveHMM* R package (Michelot et al. 2019), in order to identify three discrete behaviours: resting (small step length/high angle concentration, i.e. few turns), foraging (medium step length/low angle concentration, i.e. many turns) and commuting (large step length/high angle concentration; see **Table S5.4** for details). HMM are considered an effective method for identifying seabird foraging behaviour (Bennison et al. 2018), and the method has previously been applied to a range of Procellariiformes (Bonnet-Lebrun et al. 2020; Conners et al. 2021). Appropriate parameter priors for the final model were selected through a comparison of negative log-likelihood values of a series of candidate models (n = 25) run iteratively using a range of randomly selected mean and standard deviation (sd) parameter values constrained within realistic limits as determined through prior data visualisation (after Michelot and Langrock 2019).

Step 3: Estimating foraging distributions (2008–2019)

Locations were transformed to Lambert Azimuthal Equal Area projection. Kernel density estimations (KDE) were run on foraging locations using the *adehabitatHR* R package (Calenge 2006) by applying a 5-km grid and a fixed smoothing parameter of 18 km following BirdLife International (2004) and Robertson et al. (2014). We first ran KDEs for individual data groups, and subsequently ran an “overall” KDE for each breeding stage containing data from all colonies and years combined. Estimated utilisation distributions (UDs) were limited to at-sea locations only and were rescaled to sum to 1 within each data group. We define the 95% UD as the general foraging area, and the 50% UD as the core foraging area.

Tracking samples are assumed to be representative of their respective colonies. We validated this assumption using bootstrapping via the track2KBA R package (Beal et al. 2021b). Briefly, we randomly selected an increasing number of birds 100 times, calculated the 50% foraging UD as described above, and modelled this as a function of sample size using non-linear asymptotic regression. Samples were assumed to be at least partly representative of the wider population if the representativeness value was ≥70% (Lascelles et al. 2016; **Table 5.1**).

Step 4: Fishing distribution and seabird overlap (2012–2019)

Fishing effort data, based on daily automatic identification system (AIS) location fixes, are available from GFW from 2012 onwards. VMS data are not publicly available for the Argentine fleet; thus using the AIS data had the benefit of spanning our entire study area.

The accuracy of AIS data from GFW, in terms of fishing effort and gear type, is 95% (Kroodsma et al. 2018). Representativity of estimated fishing effort from these data have also previously been validated against those produced using VMS data from the Falkland Islands for the years of 2012–2018 (Bonnet-Lebrun et al. 2020). Finally, Arrizabalaga et al. (2019) found AIS use in our study area to be high, particularly in the southern half of the Patagonian Shelf.

AIS data were downloaded at 0.01° resolution, clipped to our incubation and brooding study areas (as defined by bird track extents) and filtered to only include fishing dates for which we have tracking data (**Table S5.2**). Mean fishing effort (hrs) was summarised by study year/ breeding stage/ gear type across a 50 km^2 grid. Grid squares where no fishing occurred were removed. GFW data were then reformatted to match the structure of the seabird tracking data by weighing individual locations by fishing effort, before running KDEs with a smoother of 50 km. The smoother was chosen based on previous data exploration to avoid over- and under-smoothing.

To assess overlap between albatross foraging and fishing activities, we first re-calculated the albatross KDEs using a 50-km grid that matched the fisheries effort grid. We then used two approaches to quantify overlap between albatross foraging and fishing activities, and to assess where, and with which fleets, overlap was greatest. Firstly, we calculated the Bhattacharyya's affinity (BA; Bhattacharyya 1943; Fieberg and Kochanny 2005) to assess the similarity in foraging and fishing distributions by breeding stage and gear type in individual years for individual colonies. BA represents an index of similarity between the UDs of individual data groups of black-browed albatross (UD_b) and fishing vessels (UD_f), and ranges between 0 (no similarity between the distributions) to 1 (identical distributions):

$$BA = \sum_{x,y} \sqrt{UD_b(x,y)UD_f(x,y)} \tag{1}$$

where x and y represent the spatial domain used by the animals/vessels. Secondly, we plotted fisheries exposure (FE), an index of the relative likelihood of a foraging albatross encountering fishing activity across the entire study area (after Bonnet-Lebrun et al. 2020), where:

$$FE(x,y) = UD_b\left(x,y\right)UD_f(x,y) \tag{2}$$

The FE was assessed for individual breeding stages and gear types (a) within individual years and colonies, and (b) summarised across all years and colonies. Whilst BA provides a numerical indication of overlap (with BA < 0.5 considered low, see e.g. Gulka and Davoren 2019), the FE provides visual information on where this overlap was greatest.

5.3. Results

5.3.1. Albatross foraging trip distribution

Between 2008 and 2019, we obtained 910 foraging trips from 533 birds (mean n(trips/bird) = 1.7), of which 78% were complete (**Table 5.1**). All trips were confined to the Patagonian Shelf between 55°S and 39°S (**Figure 5.2**). During incubation, trips generally extended west and north from the Falkland Islands, regularly reaching north of 45°S (mean ± sd: max. (cum.) dist., duration = 409 ± 259 (1310 ± 1019) km, 4.8 ± 3.3 days) (**Table 5.1**). During brooding, trips dispersed towards the tip of South

America from respective colonies, generally remaining south of 50°S (mean ± sd: max. (cum.) dist., duration = 252 ± 197 (640 ± 463) km, 1.7 ± 1.0 days; **Table 5.1**).

Estimated core foraging (50% UD) occurred almost exclusively within the Falkland Islands and Argentine jurisdictions (area of 50% UD inside FCZs: 51.0 ± 34.1%; AEEZ: 48.5 ± 34.3%; high seas: 0.5 ± 2.8%; **Table 5.1**; **Figure 5.3**). Core foraging areas regularly concentrated around respective colonies during both incubation and brooding (except at BCI 2015). During incubation, core foraging areas of birds from NWI and SJI also occurred in irregular and fragmented areas to the west within Grande Bay (Bahia Grande), and (to a lesser extent) north of the Falkland Islands up to 40°S, and along the AEEZ boundary at ~46–47°S (**Figure 5.3 A**). During brooding, the core foraging area of SJI birds lay exclusively in the proximity of their colony in both years (**Figure 5.3 B**). In comparison, core foraging areas of brooding NWI birds also regularly occurred in an area between the Falkland Islands and the southern tip of South America, and around Staten Island off Tierra del Fuego (**Figure 5.3 B**). At BCI, the single year of tracking revealed core foraging areas of brooding birds stretching southwards onto the Burdwood Bank, as well as around Staten Island (**Figure 5.3 B**).

Table 5.1 Summary of foraging trip characteristics of GPS tracked breeding black-browed albatross at Beauchêne Island (BCI), New Island (NWI) and Steeple Jason Island (SJI). Number of birds and trips are indicated as all trips/complete trips. Cumulative distance (Cum. dist.), maximum distance from the colony (Max. dist.) and trip duration (Dur.) are given as mean ± standard deviation and are based on complete trips only. The remaining statistics are based on all trips. The single data group with low colony-level representativeness (CLR) is highlighted (*); see Materials and Methods for additional details. Core foraging = 50% utilisation distribution. FCZs = Falklands Conservation Zones; AEEZ = Argentine Exclusive Economic Zone.

Island	Year	Breeding Stage	n(birds)	n(trips)	Cum. dist. (km)	Max. dist. (km)	Dur. (d)	% CLR	% core foraging inside FCZs	% core foraging inside AEEZ	Bhattacharyya's affinity with trawlers; longliners
BCI	2015	Brooding	9/8	38/25	921.1 ± 338.4	378.9 ± 111.1	2.0 ± 0.9	88.7	35.1	64.1	0.23; 0.35
NWI	2008	Brooding	39/39	81/72	780.3 ± 576.7	278.8 ± 248.2	2.2 ± 1.3	98.8	100.0	0.0	No fishery data
	2009	Incubation	20/12*	21/12*	2504.1 ± 912.7	618.3 ± 223.9	7.9 ± 2.0	53.9*	15.5	84.4	No fishery data
		Brooding	35/34	70/63	869.2 ± 411.2	386.2 ± 215.5	2.4 ± 1.1	94.5	58.6	41.0	No fishery data
	2010	Brooding	86/79	159/145	339.8 ± 284.8	123.9 ± 120.9	1.2 ± 0.8	99.1	100.0	0.0	No fishery data
	2013	Incubation	23/22	37/35	761.4 ± 795.3	339.9 ± 291.7	3.1 ± 2.7	90.9	60.9	27.6	0.36; 0.13
		Brooding	21/21	49/49	569.5 ± 303.5	238.5 ± 132.1	1.3 ± 0.6	95.2	60.7	39.3	0.32; 0.00
	2015	Brooding	32/20	58/25	870.7 ± 319.6	359.8 ± 115.3	1.9 ± 0.6	90.7	42.9	56.4	0.42; 0.15
	2016	Brooding	17/12	32/19	826.1 ± 418.1	343.8 ± 175.3	2.0 ± 1.0	87.5	28.1	71.8	0.45; 0.16
	2017	Brooding	32/28	32/28	1179.0 ± 442.6	373.0 ± 175.7	1.6 ± 0.5	90.6	17.9	82.0	0.39; 0.12
	2018	Incubation	31/19	53/32	832.1 ± 811.8	307.0 ± 253.4	3.1 ± 2.7	83.9	23.8	75.4	0.32; 0.13
		Brooding	28/18	59/31	755.3 ± 417.4	291.2 ± 179.0	2.0 ± 1.1	95.4	73.4	26.6	0.43; 0.04
	2019	Incubation	29/19	35/21	1662.1 ± 838.6	529.3 ± 182.5	6.4 ± 3.2	73.9	1.0	99.0	0.21; 0.15
		Brooding	22/12	41/20	1005.9 ± 503.0	464.9 ± 211.8	2.1 ± 1.2	82.8	17.8	82.0	0.35; 0.11
SJI	2009	Brooding	25/24	36/33	546.6 ± 404.3	234.5 ± 145.7	2.2 ± 1.1	93.1	100.0	0.0	No fishery data
	2011	Brooding	56/49	70/63	369.1 ± 293.1	144.1 ± 121.4	1.2 ± 0.7	97.2	100.0	0.0	No fishery data
	2012	Incubation	37/36	39/38	1646.3 ± 977.2	425.7 ± 215.4	5.9 ± 3.0	76.5	24.1	75.9	0.53; 0.33

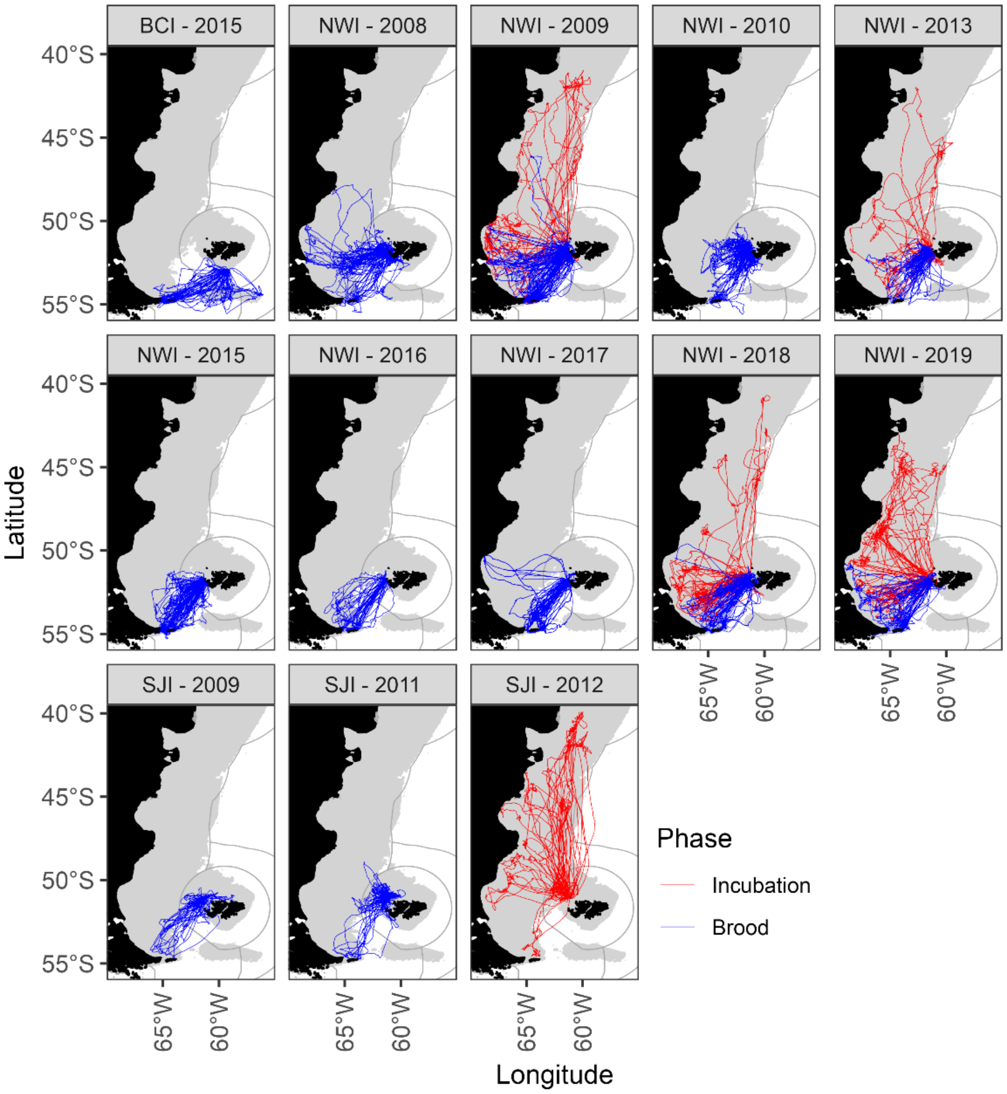


Figure 5.2 GPS tracks from breeding black-browed albatross at Beauchêne Island (BCI), New Island (NWI), and Steeple Jason Island (SJI) between 2008 and 2019. The grey lines represent the Argentine Economic Exclusion Zone and the Falklands Conservation Zones. Grey shading indicates the 200-m shelf break.

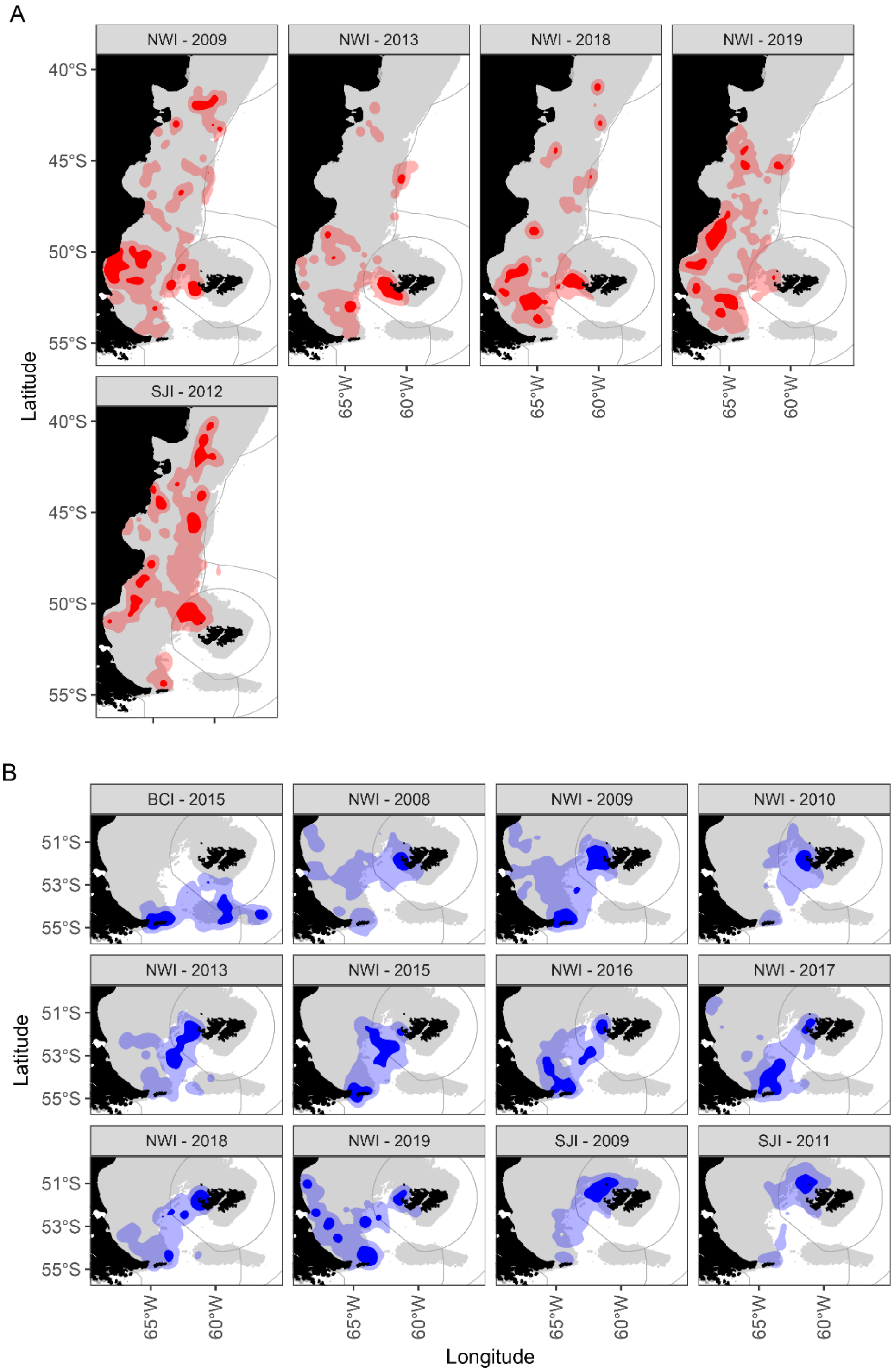


Figure 5.3 Kernel utilisation distributions (UD) of breeding black-browed albatross from Beauchêne Island (BCI), New Island (NWI), and Steeple Jason Island (SJI), during (A) incubation and (B) brooding. Dark red/blue shading = core foraging area (50% UD); lighter red/blue shading = general foraging area

(95% UD). Grey shading = the 200-m depth contour (shelf break). Grey lines = the Argentine Economic Exclusion Zone the Falklands Conservation Zones.

5.3.2. Fishing effort distribution

Between 2012 and 2019, five fishing gear types operated within the birds' distribution extent (95% UD; **Table 5.2**). Trawling made up 87% of total fishing effort (hrs), followed by longline (5%). Fixed gear (traps and pots), jigging, purse seine fishing, and undefined fishing gear accounted for the remaining 7% of overall fishing effort (**Table 5.2**).

Fishing activities during both incubation and brooding were highest north of the FCZs across the shelf break and on the shelf itself within the AEEZ (**Figure 5.4 A**). When limited to the foraging extent of brooding birds, fishing effort was highest within the northern and western part of the FCZs, along the coast of Argentina within Grande Bay, around Staten Island and in the area between the Falkland Islands and the southern tip of South America (**Figure 5.4 B**).

Table 5.2 Gear-specific fishing effort across the black-browed albatross foraging range during incubation and brooding. Fishing effort is total fishing hours and the average daily mean (± sd) over the whole area between 2012 and 2019. Total number of study dates during incubation: n = 101 days; brooding: n = 111 days (see **Table S5.2** for details). Note that the daily mean can be >24 hrs, because it denotes the effort from multiple vessels operating simultaneously.

Gear	Fishing effort (hrs) during incubation	Fishing effort (hrs) during brooding
Trawlers	180348.3 (1803.5 ± 969.3)	12436.5 (112 ± 76)
Longline	9076.9 (90.8 ± 44.6)	2878.5 (25.9 ± 19.7)
Undefined fishing	9293.8 (108.1 ± 87.8)	1159 (21.9 ± 22.4)
Fixed gear	3272.3 (40.4 ± 35.4)	939.8 (14.2 ± 16.7)
Jiggers	1537.8 (17.5 ± 22)	0
Purse seines	200.3 (5.3 ± 7.6)	0

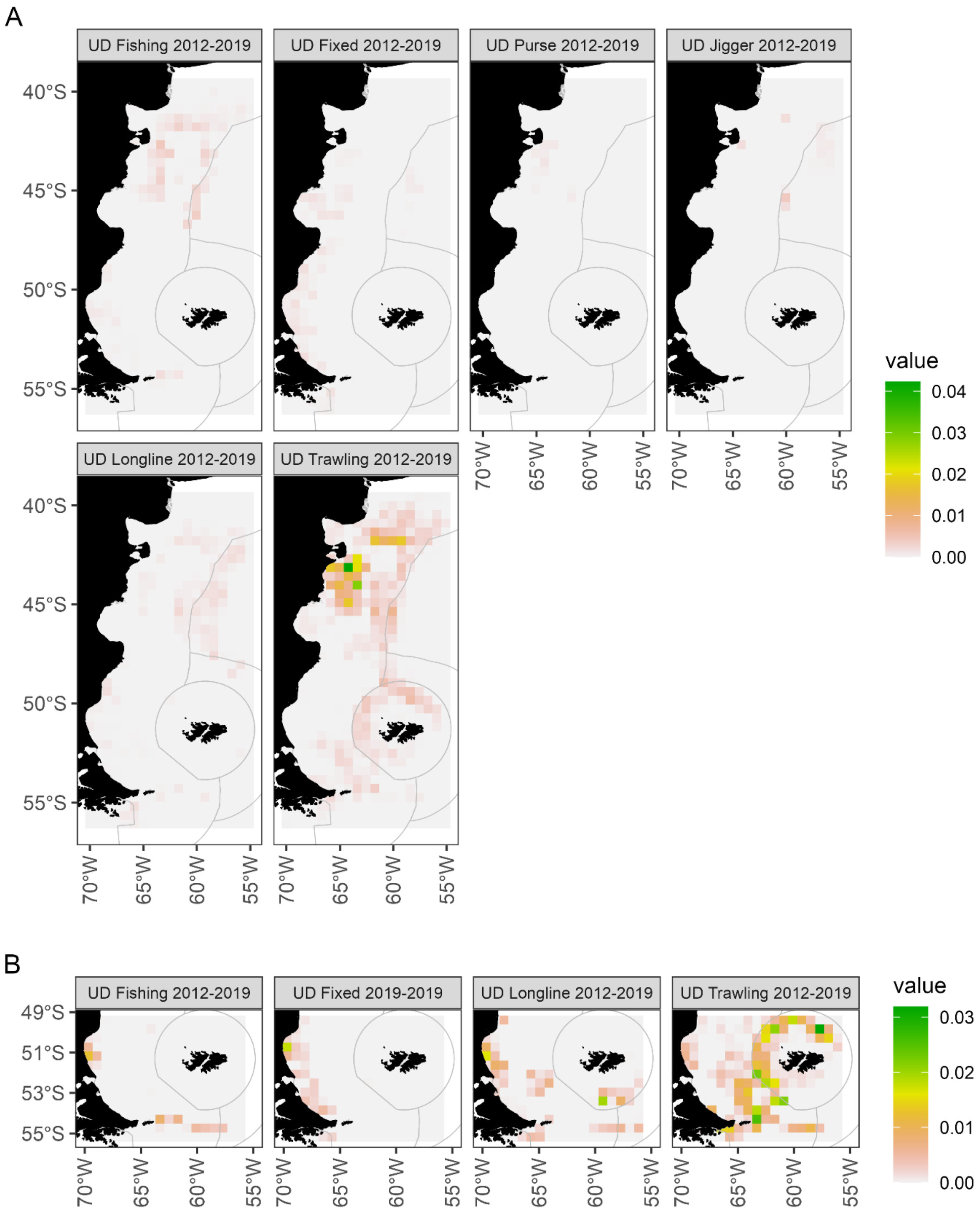


Figure 5.4 Fisheries utilisation distributions (UD) across all study years during (A) incubation (2012, 2013, 2018, 2019), and (B) brooding (2012, 2013, 2015–2019) by gear type. The UD values of all panels combined add up to 1 for each breeding stage, to allow comparison of relative UDs between gear types. Fishing = unspecified gear type. Fixed = fixed gear.

5.3.3. Seabird-fisheries overlap

Albatross overlap with all fisheries was moderately low during both incubation and brooding (mean ± sd, BA = 0.39 ± 0.10; BA = 0.37 ± 0.05, respectively), and highest for trawlers (incubation: BA = 0.36 ± 0.13; brooding: BA = 0.37 ± 0.08; **Table 5.1**; **Figure 5.5**).

During incubation, exposure was highest with trawlers operating ~42°S and 47°S, and, less so, with trawlers operating north-west of the FCZs and in the area southwest of the Falkland Islands towards the tip of South America (**Figure 5.5**). During brooding, the main exposure across years were with trawlers on the border of the FCZs/AEEZ; east of Staten Island, and the area in between (**Figure 5.5**). The single year/stage for which we can simultaneously assess fisheries exposure by different colonies (brood 2015) shows that spatial and gear-specific exposure can be colony-specific; unlike birds from NWI, birds from BCI had higher relative exposure to longline fisheries compared to trawlers (**Table 5.1**; **Figure 5.6**).

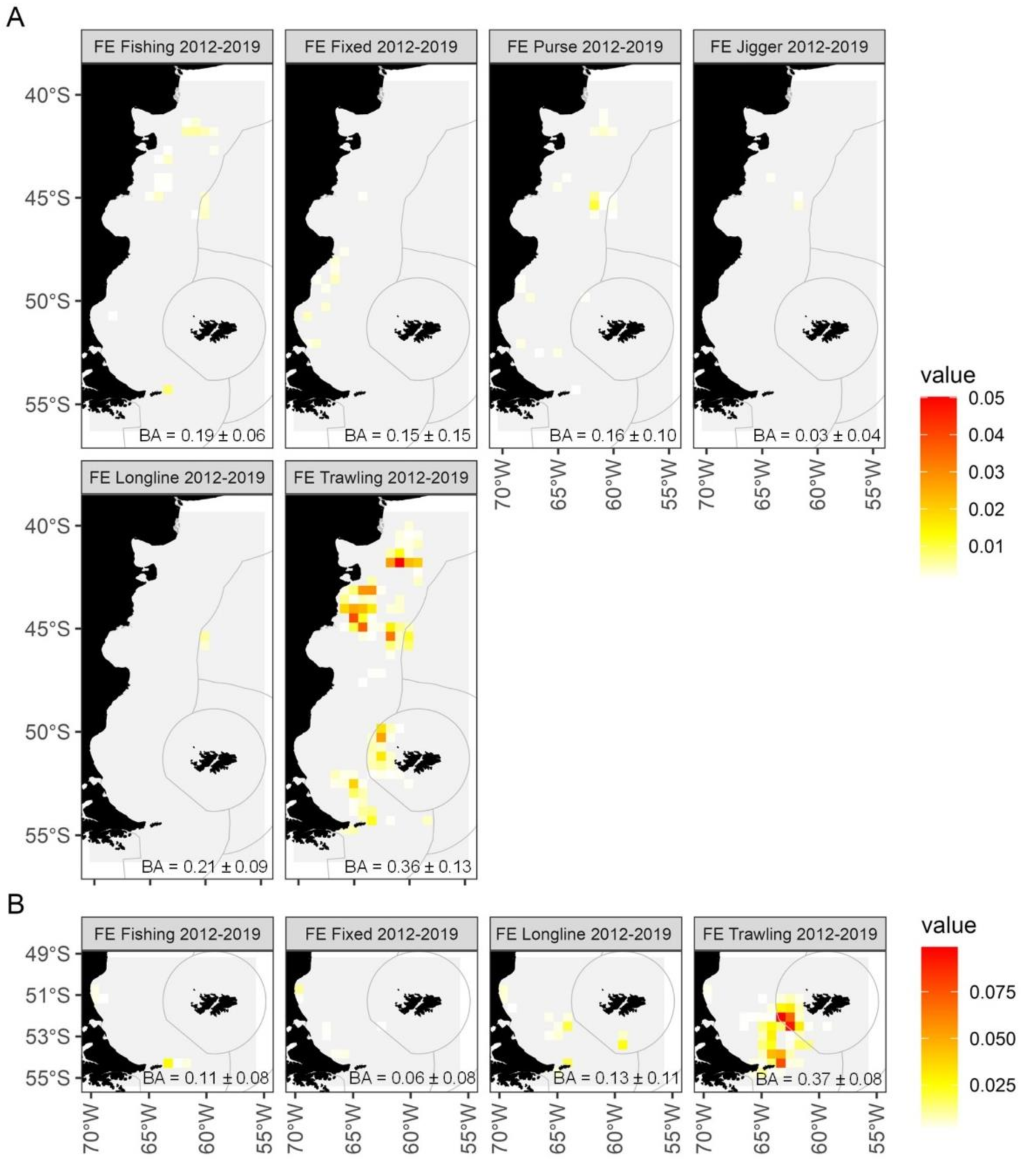


Figure 5.5 Fishing exposure (FE) index for black-browed albatross from all study years and colonies during (**A**) incubation (2012, 2013, 2018, 2019), and (**B**) brooding (2012, 2013, 2015–2019), indicating areas where albatrosses are most likely to encounter fishing vessels. Fishing = unspecified gear type. Fixed = fixed gear, Purse = Purse seine. BA = Bhattacharyya's affinity (mean ± standard deviation) between the fishing and albatross utilisation distributions, providing a numerical indication of total overlap (with BA < 0.5 considered low overlap, see Materials and Methods). FE values have been rescaled to sum to 1 within each breeding phase.

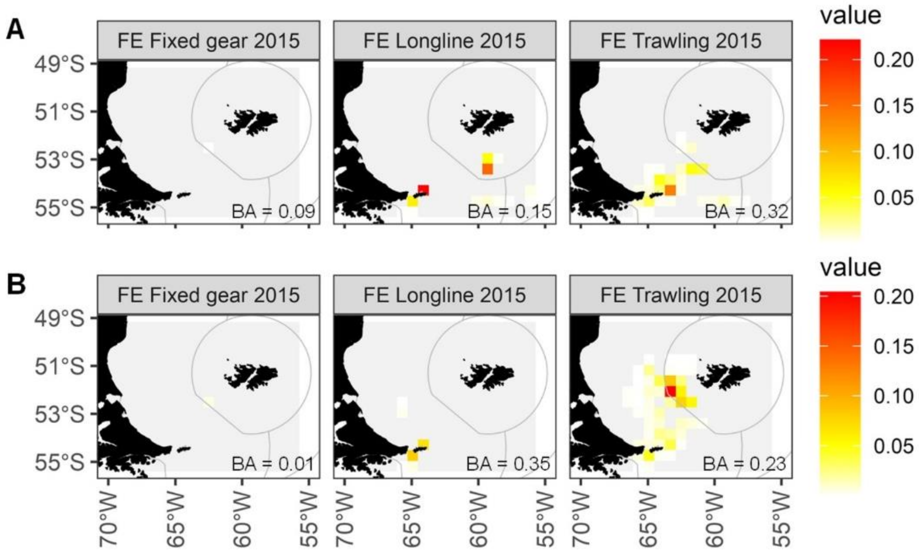


Figure 5.6 Fishing exposure (FE) index during brooding in 2015 for black-browed albatrosses breeding on (**A**) Beauchêne Island, and (**B**) New Island, indicating areas where albatrosses are most likely to encounter fishing vessels. BA = Bhattacharyya's affinity between the fishing and albatross utilisation distributions, providing a numerical indication of total overlap (with BA < 0.5 considered low overlap, see Materials and Methods). FE values have been rescaled to sum to 1 within each colony.

5.4. Discussion

We used bird-borne and vessel-borne tracking data to address key knowledge gaps regarding foraging areas and fisheries overlap by black-browed albatross across the Patagonian Shelf during two breeding phases. Our movement data originates from three colonies that total 50% of the global BBA breeding population, and includes BCI (the world's second largest population) from which published tracking data are scarce. Birds foraged almost exclusively within Falkland Islands and Argentinian jurisdictions where overlap with fisheries (mostly trawlers) was relatively low during both incubation and brooding. Our results have implications for ecosystem-based fisheries management on the Patagonian Shelf Large Marine Ecosystem, both across and beyond national jurisdictions.

5.4.1. At-sea distribution and foraging areas

As found by Huin (2002), core foraging areas differed between incubation and brooding, with the latter generally being closer to the colony, presumably due to the constraints of early chick development (Catry et al. 2013b). Nevertheless, core foraging areas regularly occurred in the immediate vicinity of NWI and SJI colonies during both breeding stages. The waters around SJI (and BCI) are areas of high conservation interest (Baylis et al. 2021).

Besides in the vicinity of colonies, core foraging occurred (1) within Grande Bay/ Bahia Grande to the west of the Falkland Islands (NWI & SJI, incubation); (2) south-east of Mare de la Plata (NWI & SJI, ~40–42°S; incubation); (3) along the Shelf slope and AEEZ boundary immediately north of the FCZs (~46–47°S; NWI & SJI, incubation); (4) the area south-west of the Falkland Islands towards Staten Island/ Isla de los Estados (NWI, incubation and brooding); (5) Staten Island (BCI & NWI, brooding); and (6) the Burdwood Bank (BCI, brooding; also used by these birds during incubation in a previous study; Furtado et al. 2020). Corresponding with habitat associations shown previously (e.g. Catry et al. 2013; Paz et al. 2021), these areas are largely characterised by productive shallow waters with enhanced mesoscale activity and eddies resulting from northwards and eastward current flow (Acha et al. 2004; **Figure 5.1**), and are therefore also hotspots for zooplankton, fish nursery grounds and forage fish (Sánchez et al. 1995; Sabatini et al. 2004; Belleggia et al. 2017).

With the caveat of low temporal overlap in sampling years between colonies, and limited data from SJI and BCI, our study nonetheless provides useful new insight into colony-specific foraging areas, and inter-colony foraging behaviours. For example, NWI and SJI birds share similar foraging areas during incubation, but segregate during brooding (also see Granadeiro et al. 2011). BCI was the only colony to visit the Burdwood Bank during brooding. Their absence from coastal grounds challenges previous hypotheses that brooding NWI birds avoid inshore areas due to possible competition with BCI birds (Catry et al. 2013b). However, concurrent tracking during two year/phase combinations does show evidence of density-dependent, competitive avoidance behaviour during brooding, where inter-colony overlap is low near the colonies (i.e. NWI and SJI 2009), but higher farther from the colonies in an area of high prey abundance and reduced inter-colony competition (i.e. NWI & BCI 2015, with overlap at Staten Island) (Wakefield et al. 2013; Oppel et al. 2018).

5.4.2. Fisheries overlap

Previous studies based on GPS data have found relatively little influence of Falkland Islands fisheries on black-browed albatross foraging behaviour during chick-rearing (Granadeiro et al. 2011; Catry et al. 2013b). However, the increased amounts of discards found in adult diet during incubation by McInnes et al. (2017a) would suggest higher fishery interaction in this breeding stage compared with brooding, although bias in dietary assessments cannot be excluded, particularly given the low sample size in that study. Our research expands on these previous studies, showing that breeding albatross foraging grounds overlapped relatively little with fisheries across their entire range, with overlap being similar during incubation and brooding.

Relative exposure was generally highest with trawlers. Although the GFW data does not denote specific fleets or trawler types, based on vessel information on fisheries in these areas (e.g. González-Zevallos and Yorio 2006; González-Zevallos et al. 2011; Favero et al. 2013; Tamini et al. 2015, 2021; Paz et al. 2018; Seco Pon et al. 2018; Falkland Islands Government 2021), our results suggest relatively higher exposure to (1) the ice trawler fleets operating at ~47–42°S south/south-east of Mar del Plata coast (during incubation), (2) the freezer trawl fleet operating at ~47–45°S along the AEEZ boundary, (3) the finfish freezer trawler fleet operating in west FCZs (incubation & brooding), and (4) the freezer factory fleet operating in the south of the AEEZ (incubation & brooding).

We stress that most of our data originate from a single colony (NWI), and care is warranted when extrapolating results to other colonies. For example, when considering the only available year/phase with concurrent data from two colonies and available AIS data, overlap was highest with trawlers for NWI birds, but highest with the demersal longline fleet for BCI birds. This showcases the importance of colony-specific distribution patterns (also see e.g. Wakefield et al. 2013) to draw robust conclusions about area- and gear-specific exposure (Richards et al. 2022).

Previous studies suggest that fisheries and albatrosses may compete for fish and squid on the Patagonian Shelf (Thompson 1992; Grémillet et al. 2000).

Indeed, overfishing on the Shelf led to the collapse of the pelagic southern blue whiting – a natural prey of albatrosses (Thompson 1992; Laptikhovsky et al. 2013). However, we found little evidence for albatross/fishery competition due to limited spatial overlap during breeding. Moreover, while albatross target pelagic prey (i.e. Fuegian sprat *Sprattus fuegensis*, Lobster krill *Munida gregaria* and squid), the majority of trawl fisheries south of 40°S fish for demersal species which occupy different habitats (e.g. Alemany et al. 2018). Nonetheless, we might expect increased competition further north, where juvenile and immature birds (in particular) disperse to (e.g. Colabuono and Vooren 2007; Carneiro et al. 2020), and where pelagic species (e.g. anchovy) are fished (Skewgar et al. 2007).

5.4.3. Methodological considerations

GPS sampling frequency (i.e. the temporal interval of successive positional fixes recorded) can significantly affect the accuracy of derived foraging trip length, and time spent in a given area (e.g. Ryan et al. 2004; but see Bennet et al. 2019). In this study, while all data were sub-sampled or interpolated to a standard temporal resolution, we did not test for or correct the potential influence of lower sampling frequencies used in incubating birds. However, visual inspection of tracks with different sampling frequencies revealed no obvious variation in direction travelled or key areas utilised (also see Bennet et al. 2019). Further, tracks of mixed sampling frequency (30 and 60-min) from a subset of our data have been used successfully without corrections in a previous study assessing at-sea behaviours (Campioni et al. 2017).

Nonetheless, future studies could benefit from additional analytical procedures in this regard.

Furthermore, approach of assessing seabird-fishery overlap does not allow identification of fine-scale interaction; however, it does provide a proxy for increased vessel interaction (reviewed in Bot et al. 2018; Carneiro et al. 2020). This is a common approach in ecological risk assessments with fisheries for fragile taxa and features (e.g. seabirds: Clay et al. 2019; Carneiro et al. 2020; Pereira et al. 2021; sharks: Queiroz et al. 2016; marine mammals: Baylis et al. 2018; seamounts: Kerry et al. 2022), and highlights fleets where limited resources for effective mitigation deployment and policing could return the highest conservation benefits (see below; e.g. Croxall et al. 2013; Carneiro et al. 2020). Nonetheless, the highly dynamic movement patterns of both seabirds and fishing vessels warrants analyses at multiple scales to fully explore the spatio-temporal variation in their associations (Torres et al. 2013b, a). For example, fine-scale analysis of foraging trips by brooding birds from NWI (2008, 2009) and SJI (2009) indicated that most individuals were usually not close to vessels within the FCZs and expended proportionally little foraging effort near ships (Granadeiro et al. 2011). Further fine-scale analyses covering the whole of the black-browed albatross foraging area would be beneficial.

In addition, using AIS to quantify fishing vessel distribution does not guarantee complete coverage. Not all boats are obliged to use AIS (e.g. artisanal vessels <15 m), while others may illegally disable their transponders (Ford et al. 2018). Artisanal boats are largely limited to <40°S (Oceana 2021), and are therefore outside the range of our tracked birds. However, the area ~47–46°S on the

AEEZ boundary was flagged as a hotspot for illegal fishing in 2021 (https://insightcrime.org/news/gamechangers-2021-iuu-fishing-plundered-latin-americas-oceans/), predominantly by Chinese-flagged jiggers, but also Spanish-flagged trawlers. The extent of fishery overlap in this area may therefore have been underestimated in our study.

Finally, our study focuses exclusively on adult birds during incubation and brooding. However, BBA foraging behaviour varies among seasons (e.g. chick-guard vs post-guard; summer vs winter; Ponchon et al. 2019), breeding status (e.g. failed/successful breeders, Jaeger et al. 2014), and age (juveniles/immature/adults; Carneiro et al. 2020). Important areas of fisheries overlap can be missed when not all life-history stages are accounted for (Carneiro et al. 2020). An improved understanding of this variability is needed for effective conservation management, and should be a priority for future research.

5.4.4. Management implications and future studies

All identified fleets, except for the red shrimp fishery, have reported black-browed albatross bycatch (González-Zevallos and Yorio 2006; Sullivan et al. 2006b; González-Zevallos et al. 2011; Favero et al. 2013b; Tamini et al. 2015, 2021; Paz et al. 2018; Kuepfer et al. 2022b). Mitigation measures are mandatory in some of these fleets, albeit with varying levels of effectiveness, particularly on trawlers (Consejo Federal Pesquero 2016; Kuepfer et al. 2018; Paz et al. 2018; Tamini et al. 2019, 2021; Falkland Islands Government 2021). Our index of relative fisheries exposure does not consider whether the vessels with which the birds interact utilise effective mitigation measures; rather, as mentioned above, it provides a guide for targeted management should these measures currently be absent, ineffective or poorly regulated.

The Falkland Islands black-browed albatross population is currently stable (Stanworth and Crofts 2022), suggesting that any negative fishery interactions are not having substantive population-level effects. Nevertheless, this could change as the climate warms which may negatively influence prey availability (Cury et al. 2011; Weimerskirch et al. 2012; Sydeman et al. 2021a), with the potential for additive or unforeseen fishery impacts (Barbraud et al. 2012; Pardo et al. 2017; Dias et al. 2019). Therefore, effective mitigation measures, coupled with adequate compliance enforcement across high exposure fleets, should remain a critical aspect of ecosystem-based fisheries management (Pardo et al. 2017). More broadly, as marine predators often congregate in multi-species foraging aggregations, localised management and mitigation for black-browed albatross would also benefit other species (Baylis et al. 2019).

As with many larger predators, the distribution of the black-browed albatross is not constrained by political boundaries, and their management requires a multilateral effort across national jurisdictions (Beal et al. 2021a). Seabird interactions on trawlers are more prevalent in the presence of discards (e.g. Kuepfer et al. 2022b). Therefore, to reduce this issue, discard management is being implemented within the FCZs freezer trawlers (Kuepfer and Barton 2018), and this best practice should be considered more widely in trawl fleets across the Patagonian Shelf, and beyond (also see e.g. Copello et al. 2016; Paz et al. 2018; Tamini et al. 2021; ACAP 2021). Where complete removal of discards is impractical, other options such as physical mitigation (e.g. bird scaring lines, see e.g. Tamini et al. 2019), or a dynamic management approach are alternatives worth exploring (e.g. temporary fishery closures, see e.g. Sydeman et al. 2021b, but see Copello et al. 2016).

5.5. Supplementary information

Table S5.1 Characterisation of commercial fleets operating on the Patagonian Shelf within the Argentine EEZ and the Falklands Conservation Zones. Fleet sizes are approximate and can vary annually. “Offal” refers to processing waste. “Discards” refers to both offal and waste from non-commercial catch.

Fishery	Main operation location	Dominant target fish	Fleet size	Annual catch (t)	Discard availability & management	Source
Falklands Conservation Zone (2011-2020)				(2011-2020)		
Demersal trawlers	Mainly north, north-west, west and south-west	Demersal finfish, mainly Argentine hake *Merluccius hubbsi*, hoki *Macruronus magellanicus, Rockcod Patagonotothen ramsayii*	18	70,000	Offal and bycatch; some vessels use batch discarding since 2018	Falkland Islands Government 2021
Demersal trawlers	North east, east, south	Patagonian squid *Doryteuthis gahi*	16-17	56,000	Offal and bycatch; Batch discarding since 2018	Falkland Islands Government 2021
Squid jigging	Mainly north and west	*Illex* squid *Illex argentinus*	106	114,000	Minimal offal available. Crushed and discharged without processing	Falkland Islands Government 2021
Demersal longline	All FCZ	Patagonian toothfish *D. eleginoides*	1	1,200	Crushed discards released	Falkland Islands Government 2021
Argentine EEZ						
Ice trawlers	37–48°S	Demersal finfish (mainly Argentine hake) and pelagic fish (Argentine anchovy *Engraulis anchoita* and Chub Mackerel *Scomber japonicas*)	230	1,123,000	No offal produced, but continuous discharge of all discards of bycatch occurs without processing	Favero et al. 2011; Seco Pon et al. 2015
Bottom freezer trawlers	42°–55°S	Demersal finfish (mainly Argentine hake); Patagonian shrimp *Pleoticus muelleri*	135	250,633 (2000-2012)	Offal and bycatch discards. Equipped with fishmeal plants Discharge occurs without processing	Seco Pon et al. 2015

Demersal longliner	43–55°S	Mainly Patagonian tootfish *D. eleginoides*; Kinclip *G. blacodes*	3 – 4	128,470 (2000-2012)	Fishmeal plants and crushers for non-commercial catch	Favero et al. 2011; Seco Pon et al. 2015
Squid jigging	23–54°S	*Illex* squid	80	104,000	Minimal offal available. Crushed and discharged without processing	Seco Pon et al. 2015; Navarro 2019

Table S5.2 Summary of black-browed albatross tracking dates.

Island	Year	Phase	Start Date	End Date	nDays
BCI	2015	Brood	16/12/2015	06/01/2016	22
NWI	2008	Brood	11/12/2008	01/01/2009	22
	2009	Incubation	18/11/2009	17/12/2009	28
		Brood	11/12/2009	05/01/2010	26
	2010	Brood	15/12/2010	30/12/2010	16
	2013	Incubation	05/11/2013	25/11/2013	21
		Brood	13/12/2013	04/01/2014	23
	2015	Brood	11/12/2015	31/12/2015	14
	2016	Brood	17/12/2016	02/01/2017	17
	2017	Brood	20/12/2017	31/12/2017	12
	2018	Incubation	10/10/2018	09/11/2018	31
		Brood	15/12/2018	01/01/2019	18
	2019	Incubation	12/10/2019	11/11/2019	31
		Brood	15/12/2019	28/12/2019	14
SJI	2009	Brood	17/12/2009	31/12/2009	15
	2011	Brood	15/12/2011	28/12/2011	14
	2012	Incubation	14/10/2012	31/10/2012	18

Table S5.3 Technical details of GPS loggers used across seasons. Accuracy (locational error) for all GPS loggers was <10 m.

Island	Year	Phase	Duty cycles (minutes)	Model	Manufacturer	Weight
BCI	2015	Brooding	10, 14	GPS loggers	Earth & Ocean Technologies, Kiel, Germany	30 g
NWI	2008	Brooding	7, 14	GPS loggers	Earth & Ocean Technologies, Kiel, Germany	25 & 30 g
	2009	Incubation	14	GPS loggers	Earth & Ocean Technologies, Kiel, Germany	30 g
		Brooding	14, 21, 28	GPS loggers	Earth & Ocean Technologies, Kiel, Germany	30 g
	2010	Brooding	3, 5, 7, 10	GPS loggers	Earth & Ocean Technologies, Kiel, Germany	25 g
	2013	Incubation	60	CatTraQ GPS & GPS loggers	Perthold Engineering, Anderson, SC, U.S.A. & Earth & Ocean Technologies, Kiel, Germany	17 g & 15 g
		Brooding	12	CatTraQ GPS & GPS loggers	Perthold Engineering, Anderson, SC, U.S.A. & Earth & Ocean Technologies, Kiel, Germany	17 g & 15 g
	2015	Brooding	7, 10	i-gotU GT-120	Mobile Action Technology, Taiwan	22 g
	2016	Brooding	12, 14, 20	i-gotU GT-120	Mobile Action Technology, Taiwan	23 g
		Brooding	0.02–0.25	i-gotU GT-120	Mobile Action Technology, Taiwan	25 g
	2018	Incubation	14, 20, 30	i-gotU GT-120	Mobile Action Technology, Taiwan	26 g
		Brooding	7 & 14	i-gotU GT-120	Mobile Action Technology, Taiwan	27 g
	2019	Incubation	14, 20	i-gotU GT-120	Mobile Action Technology, Taiwan	28 g
		Brooding	7	i-gotU GT-120	Mobile Action Technology, Taiwan	29 g
SJI	2009	Brooding	14	GPS loggers	Earth & Ocean Technologies, Kiel, Germany	30 g
	2011	Brooding	1, 4, 6	GPS loggers	Earth & Ocean Technologies, Kiel, Germany	25 g
	2012	Incubation	30	GPS loggers	Earth & Ocean Technologies, Kiel, Germany	30 g

Table S5.4 Initial values for the HMM used to identify foraging locations.

State	Step mean (m)	Step sd (m)	Angle mean (rad)	Angle concentration
Resting	250	250	0	0.9
Foraging	1,500	1,500	0	0.5
Commuting	5,500	5,500	0	0.9

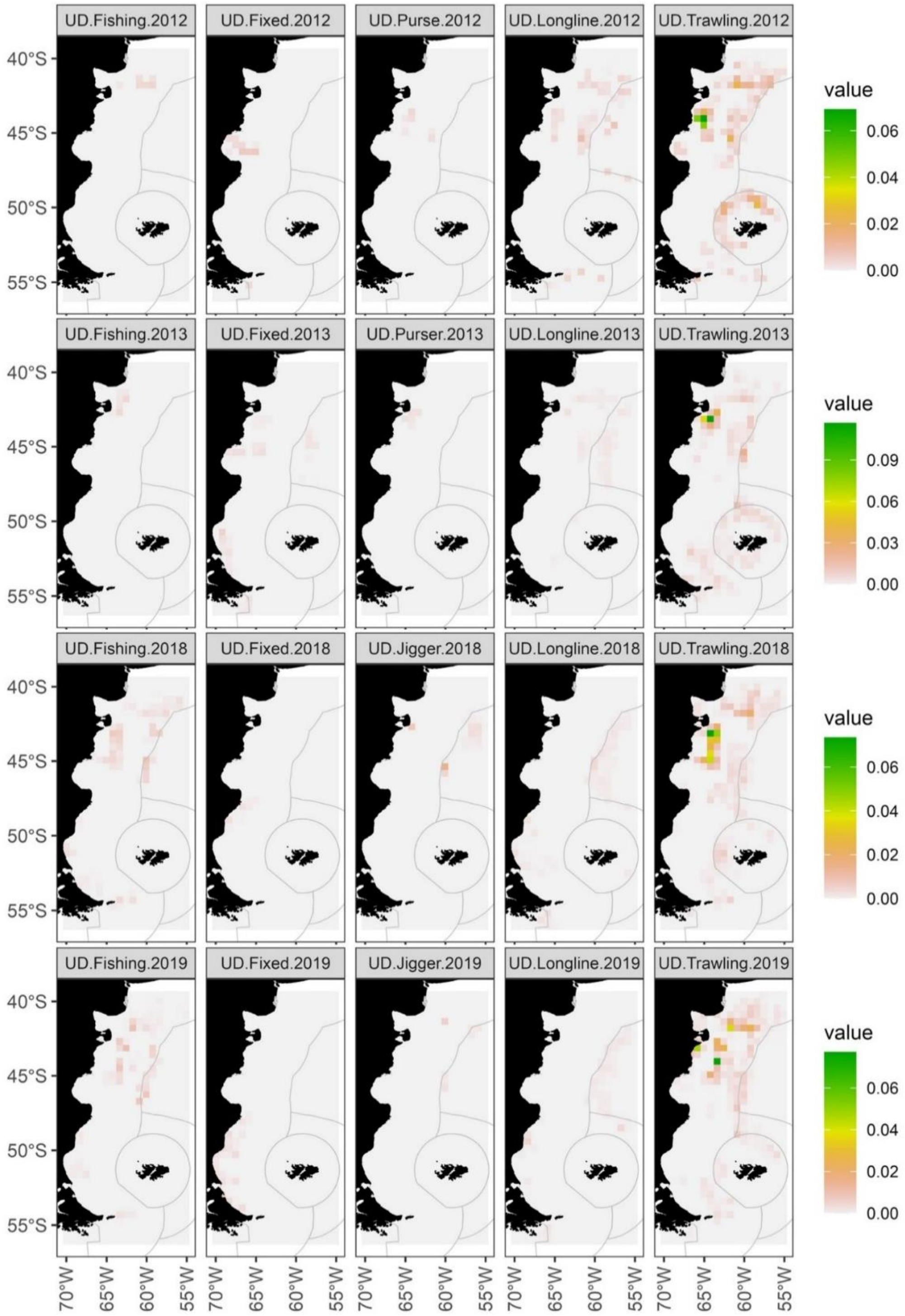


Figure S5.1 Utilisation distribution (UD) of fishing (by year and gear type) during incubation. UD values add up to 1 for each year.

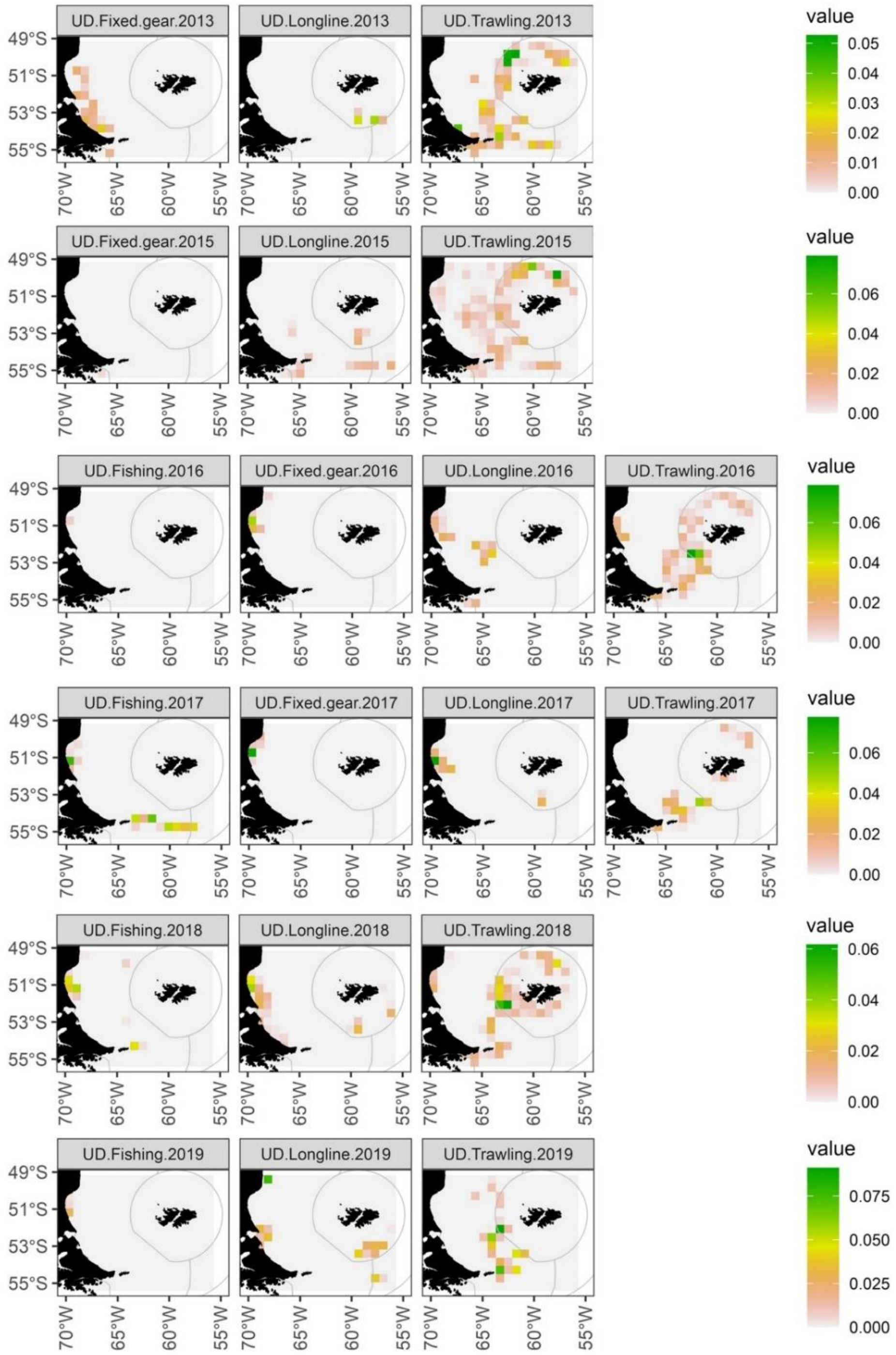


Figure S5.2 Utilisation distribution (UD) of fishing (by year and gear type) during brooding. UD values add up to 1 for each year.

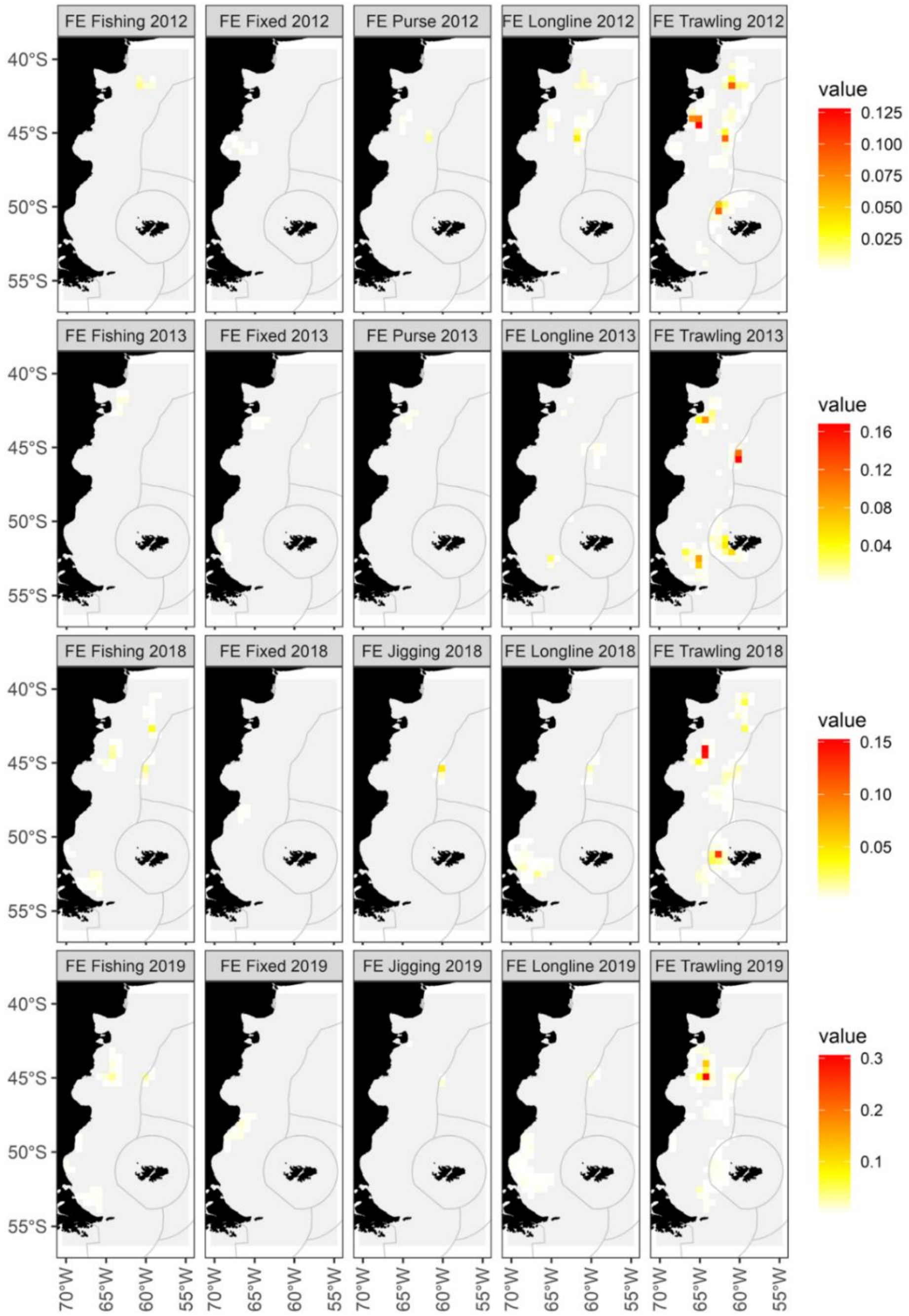


Figure S5.3 Fisheries exposure (FE) index of black-browed albatross from **Steeple Jason Island** (2012) and **New Island** (2013, 2018, 2019). Annual panels are comparable within each year. “Fishing” = unspecified gear type; Fixed = fixed gear; Purse = Purse seine. FE values have been rescaled to sum to 1 within each year. See method in main text for further information.

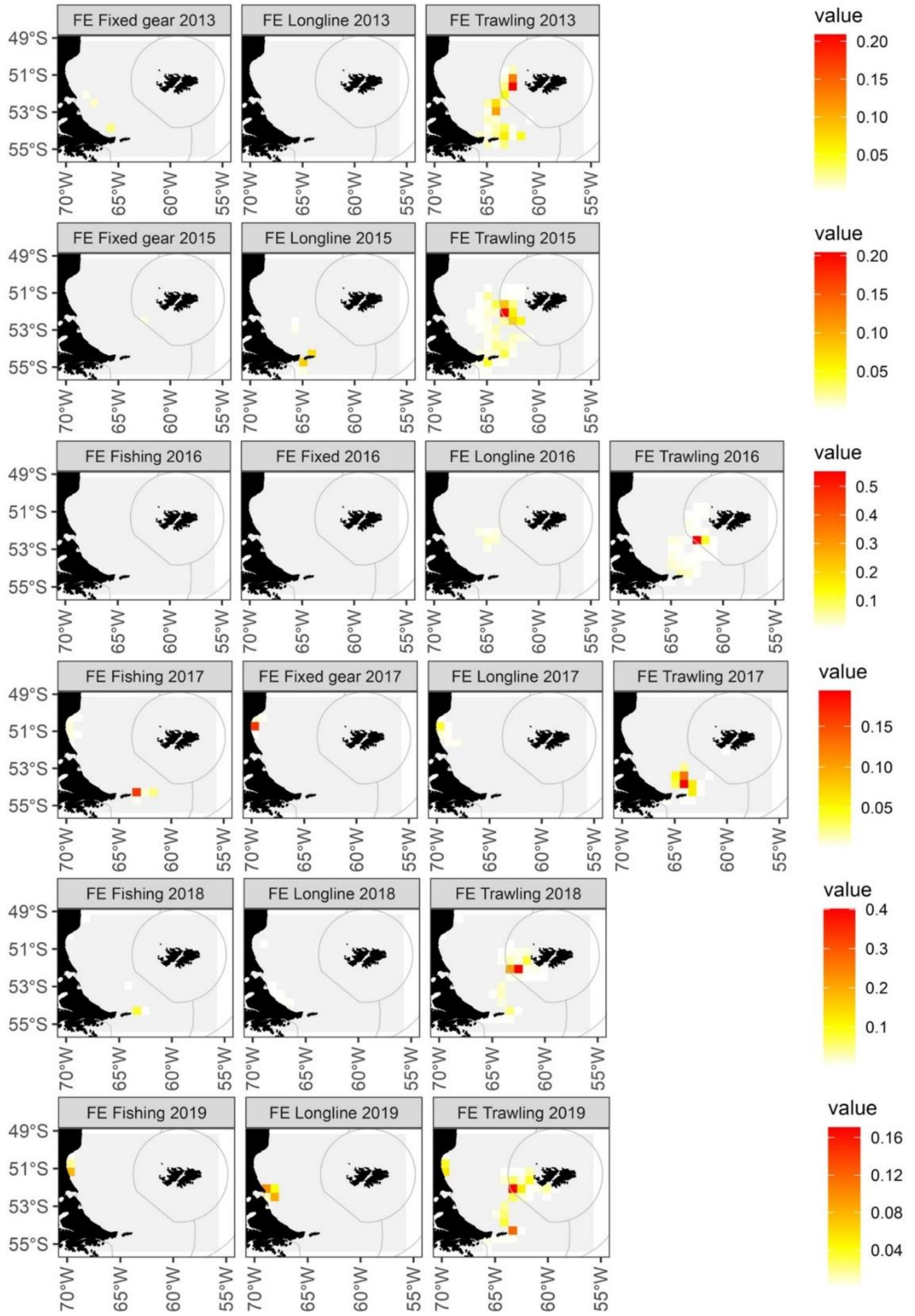


Figure S5.4 Fisheries exposure (FE) index of black-browed albatross from **New Island**. Annual panels are comparable within each year. “Fishing” = unspecified gear type; Fixed = fixed gear. FE values have been rescaled to sum to 1 within each year. See method in main text for further information.

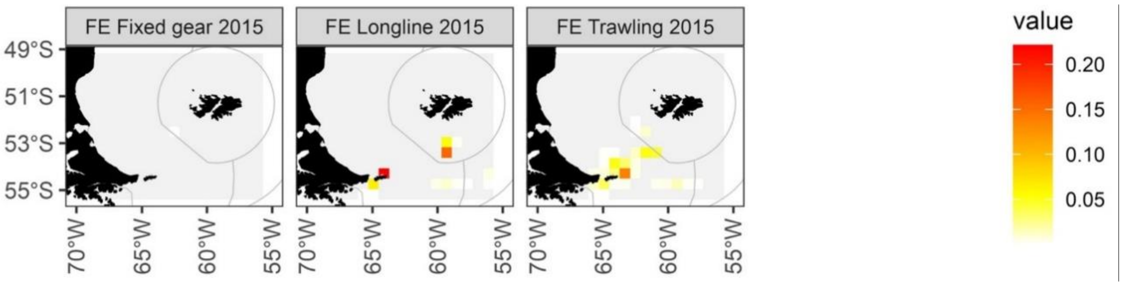


Figure S5.5 Fisheries exposure (FE) index of black-browed albatross from **Beauchêne Island**. Panels are comparable within each year. FE values have been rescaled to sum to 1. See method in main text for further information.

Chapter 6 – General discussion

6.1. Thesis summary

6.1.1. Revisiting thesis rational

Seabirds and fisheries overlap across our world's oceans, and interactions between them are ancestral and complex (Le Bot et al. 2019). In the Southwest Atlantic, where seabird-fishery overlap is amongst the highest globally (Clay et al. 2019), the black-browed albatross *Thalassarche melanophris* (hereafter BBA) represents the main scavenger at fishing vessels, and is also the dominant seabird species of bycatch (>80% in some areas; González-Zevallos et al. 2007; Tamini et al. 2015; Kuepfer et al. 2018, 2022b; Jiménez et al. 2022; Seco Pon et al. 2023). However, the extent to which fisheries influence the ecology and demography of the world's largest population in the Falkland Islands (>70% of breeding pairs; Wolfaardt et al. 2013; Crofts 2020), remains poorly understood. Given the global significance of this population, as well as their importance to the Patagonian Shelf Large Marine Ecosystem (Ventura et al. 2021), gaining an improved understanding of the nature and extent of fishery interaction is integral to biological conservation and ecosystem functioning, and supports fisheries management across the Southwest Atlantic.

6.1.2. Revisiting thesis aims and results

This thesis investigated the influence of the Patagonian Shelf fisheries on the ecology and demography of BBA during breeding, using a multitude of methods and data types.

In **Chapter 2**, we used vessel-based observations of BBA collisions with trawl gear to assess the degree to which a change in discard availability affects BBA (and giant petrel) interactions. A formal assessment of the relationship between seabird abundance and gear collision rates showed that these were positively related, supporting the use of abundance as a proxy for collision rates in other studies. Compared to continuous discarding, batch discarding significantly reduced seabird abundance and gear collisions, while zero discarding eliminated gear collisions altogether. Our findings validate batch discarding as an effective seabird-bycatch mitigation measure in trawl fisheries where zero discarding is not possible, but highlight the importance of complete waste storage between batches.

In **Chapter 3**, we used chick regurgitates collected at New Island across multiple seasons to assess inter-annual trends in diet, and determine the relationship between discard consumption, prey availability and breeding success. Our results showed that BBA chicks are primarily fed natural prey, but that discard consumption can vary significantly between years. Specifically, discard consumption was positively related with increased discard availability, but also increased in years of higher sea surface temperature anomalies and lower breeding success. The findings suggest that although natural prey are the preferred diet, BBA switch to discards when natural foraging conditions are compromised. While fishery discards may act as a buffer, discards do not appear to fully offset poor natural foraging conditions for breeding albatrosses in the long term.

Chapter 4 combined stomach content analysis (SCA) and stable isotope analysis (SIA), using data from two colonies and across two consecutive breeding seasons, to gain a more robust and holistic understanding of the importance of discards to BBA chicks. The study confirmed that chicks are predominantly fed natural prey during mid chick-rearing, but that there can nonetheless be important diet differences between colonies and years. Although discards were less important, we show that they are nonetheless taken regularly, thus exposing a large proportion of breeding adults to a bycatch risk. Combining SCA with SIA provided us with complementary information and helped with interpretation of results, but our study highlights that the optimal methodology is likely to be system- (and season-) specific.

Finally, in **Chapter 5**, GPS tracking and Global Fishing Watch (GFW) data were analysed to determine areas regularly utilised for foraging by breeding BBA, and to assess the similarity in distribution, and areas of overlap, between BBA and fisheries across the entire Patagonian Shelf. We found that BBA foraging areas were generally characterised by productive shallow waters with enhanced mesoscale activity. These areas overlapped only moderately with fishing activity during both incubation and brooding stages, and did so predominantly with trawl fisheries operating within the Falklands Conservation Zones (FCZs) and the Argentine Exclusive Economic Zone (AEEZ).

6.2. Implications for management and future research

The results from this thesis provide direction for management and future research.

6.2.1. Seabird bycatch mitigation

Chapter 2 demonstrated the potential of batch discarding as an effective bycatch mitigation tool in the Southwest Atlantic trawl fisheries. This method had previously only systematically been tested in New Zealand, where fleet composition and seabird assemblages differ to those encountered in the Southwest Atlantic or elsewhere. Our results therefore provide strong support that the method is effective across a wide range of scenarios, and certainly across the wider Patagonian Shelf (see also Jiménez et al. 2022).

The importance of effective bycatch mitigation

Although the level of bycatch is not currently to the detriment of the largest BBA population, effective mitigation remains integral to sustainable fisheries. A conservation crisis continues to be faced by the 31 species listed in the Agreement on the Conservation of Albatrosses and Petrels (ACAP) as a result of fisheries operations (ACAP 2019b). On the Patagonian Shelf, this relates particularly to grey-headed albatross (*Thalassarche chrysostoma*, Endangered), southern royal albatross (*Diomedea epomophora*, Vulnerable), white-chinned petrels (*Procellaria aequinoctialis*, Vulnerable) and wandering albatross (*Diomedea exulans*, Vulnerable) (Kuepfer et al. 2018; Tamini et al. 2019; Carneiro et al. 2022).

Furthermore, there is evidence that the additive effects of fisheries bycatch could become more detrimental in the future, in combination with increasing pressure on populations from climate change (Pardo et al. 2017). For example, an increase in extreme weather events linked to climate change will have a negative effect on adult survival in the Falkland Islands BBA population (Ventura et al. 2023). In Kerguelen, increased SST in wintering locations has been linked to reduced juvenile survival and reduced breeding success the following summer (Jenouvrier et al. 2018). Further, a change in natural prey availability mediated by elevated sea surface temperatures (Quillfeldt et al. 2010) may lead to increased scavenging of discards by BBA, therefore increasing the bycatch risk (**Chapter 3**).

Finally, it has previously been speculated that discards may form an important part of BBA diet, and therefore be at least partly responsible for the increase in population seen in the Falkland Islands (McInnes et al. 2017a). Our results from **Chapters 3 and 4**, which are based on a critical stage of the BBA reproductive cycle (Granadeiro et al. 2011), are not supportive of this theory (although see Section 6.2.3). However, they do suggest that a large proportion of breeders take discards and therefore incur a bycatch risk. Therefore, efforts to limit fisheries waste and vessel attractiveness would provide conservation benefits during breeding (Granadeiro et al. 2011, 2014; Catry et al. 2013b, although see discussion in Section 6.2.3.).

Practical implications for discard management

There is no one-size-fits-all discard storage system for trawl vessels, particularly given the diversity of the trawl fleet on the Patagonian Shelf. Various fisheries have already explored and implemented discard management options, such as in New Zealand and the Falkland Islands, with important lessons learned (**Chapter 2**; Kuepfer and Barton 2018; Kuepfer et al. 2018). Therefore, collaboration across institutions and jurisdictions would benefit conservation and fisheries agendas alike.

The retention of fishery waste is not always safely practicable, especially onboard smaller vessels. In such events, other options are worth exploring such as physical mitigation (e.g. bird scaring lines, see e.g. Tamini et al. 2019; Jiménez et al. 2022), or a dynamic management approach (e.g. temporary fishery closures, see e.g. Sydeman et al. 2021b, but see Copello et al. 2016). The latter can be effective if vessel activity is not the main predictor of seabird foraging distribution, as was found for BBA during breeding (**Chapters 3, 4, and 5**; Granadeiro et al. 2011, 2014; Catry et al. 2013).

Further, while discard management is effective against cable collisions, it has limited influence on seabirds scavenging from trawl nets and drowning through entanglement (Løkkeborg 2011). Net mortalities in the Falkland Islands have increased in recent years (Kuepfer et al. 2018; Falkland Islands Government 2020, 2021; Winter 2021), although multiple simultaneous operational changes relating to observer coverage, net modifications and discard management make it difficult to discern the reason for this. Future research should aim at resolving the cause, and accordingly influence appropriate management actions.

Finally, well-designed mitigation measures are only as effective as their correct application. Adequate policing, including with the help of observer programmers, can help ensure that mitigation regulations are followed. However, observer coverage is typically too low (~5 %) to provide an effective compliance tool (Phillips 2013). Increasing observer effort to >20% across fisheries can improve compliance, and is also the minimum recommended level at which mortality levels can be quantified for readily caught species (Babcock et al. 2003; Debski et al. 2016; Cryer et al. 2018; Wolfaardt et al. 2018). The latter is essential for assessing actual fishery impacts on seabird population and effectiveness of mitigation measures (Pardo et al. 2017). Electronic monitoring is increasingly used as an alternative or complementary tool for monitoring compliance (e.g. Glemarec et al. 2020). Advantages include reduced statistical bias and the opportunity to monitor almost continuously even on vessels where observer placement may be difficult (Gilman et al. 2021). However, effectiveness depends on numerous factors, including adequate placement and maintenance of the cameras, and how much of the digital imagery is analysed (e.g. Glemarec et al. 2020).

6.2.2. The importance of natural prey

Chapters 3 and 4 underline the importance of natural prey for BBA during breeding. Natural prey also appear the main prey for adult and immature BBA elsewhere (e.g. Benguela Upwelling System; Petersen et al. 2008). This agrees with findings elsewhere that the availability, abundance and quality of mesozooplankton and small fish from the epipelagic zone are important for seabird breeding productivity (Cairns 1987; Piatt et al. 2007; Cury et al. 2011; Kowalczyk et al. 2014). Overall, this emphasises the significance of lower and mid-trophic fish to the sustainability of many seabird populations, including the BBA.

Key natural prey of breeding BBA foraging on the Patagonian Shelf include small pelagic fish like Fuegian sprat (*Sprattus fuegensis*), lobster krill (*Munida* *gregaria*), squid (predominantly Patagonian long-finned squid (*Doryteuthis gahi*) and Argentine shortfin squid (*Illex argentinus*)), and jellyfish (**Chapters 3 and 4**; Thompson 1992; McInnes et al. 2017a, b). Like many forage fish, these species occupy a central position in the food-web by transferring energy and organic matter from lower to higher trophic levels (Laptikhovsky et al. 2013; Riccialdelli et al. 2013), forming an integral diet source of many seabirds and marine mammals (Scioscia et al. 2014; Riccialdelli et al. 2020), as well as of commercially important fish (Belleggia et al. 2014). Therefore, natural prey species of BBA represent a critical component of the wider southern Patagonian food-web and ecosystem; however, many are currently understudied.

Important knowledge gaps of natural prey

Many of the species consumed naturally by seabirds, including BBA, are not commercially exploited, and are therefore ignored by surveys and research supporting fisheries management (Barrett et al. 2007; Boldt et al. 2022). For example, while some studies exist on the abundance and distribution of the Fuegian sprat population off the Patagonian coast (Diez et al. 2018; García Alonso et al. 2020), there are none for the second population off the Falkland Islands (Sánchez et al. 1995). Similarly, research on lobster krill is entirely limited to the Beagle Channel and the Argentine coast, with little knowledge of their distribution and abundance patterns elsewhere on the Patagonian Shelf (Diez et al. 2012, 2016b, 2018). This lack of knowledge precludes an understanding of how these species may respond to potential fishing pressures or climate change, the combined effect of which can be complex (Blanchard et al. 2012; Lindegren et al. 2018; Petrik et al. 2020; Ramírez et al. 2022). As a consequence, this also limits a full understanding of the implications of fisheries on population dynamics of higher predators such as BBA.

Direct and indirect implications of fisheries on natural prey

This thesis aimed to better understand the influence of fisheries on the ecology and demography of BBA by investigating direct effects of bycatch risk and discard consumption. However, fisheries may also be exerting indirect effects on BBA, such as through direct and indirect effects on their natural prey.

Overfishing can cause trophic cascades, where the removal of larger predatory fish leads to an increase in biomass of smaller prey fish (Pauly et al. 1998; Frank et al. 2005, 2007; Szuwalski et al. 2017). Cold water ecosystems with simple food webs, such as that of the southern Patagonian Shelf, are particularly vulnerable to irreversible trophic cascades and complete restructuring of the food web (Frank et al. 2005, 2007). In line with global patterns, cephalopod catches have increased in the Falkland Islands (Falkland Islands Government 2021) – a potential sign of overfishing of groundfish stocks (Caddy and Rodhouse 1998; Doubleday et al. 2016). There has also been an increase in lobster krill swarms, although this appears to be climate driven (Diez et al. 2016b, 2018). Unfortunately, insufficient information exists on Fuegian sprat to determine a potential fisheries-mediated increase in this species, but this would be worth investigating.

Contrarily, fisheries may also be impacting some prey fish negatively, such as through bycatch. Although bycatch of Fuegian sprat and lobster krill is considered relatively low and patchy within the Falkland Islands, this may differ spatially and seasonally (e.g. Varisco et al. 2015). Many fisheries suffer from poor reporting of discards, or from omission of this information in stock assessments (Gilman et al. 2012, 2020), and fisheries on the Patagonian Shelf are not immune to this (see e.g. the Falkland Islands rock-cod *Patagonotothen ramsayi* stock assessment; Winter 2021). Such practices can have major implications on estimates of species biomass and ecosystem health (Gilman et al. 2012, 2020). A multi-species, ecosystem-based approach to fisheries management is key to ecosystem sustainability (Pikitch et al. 2004), and for quantifying indirect effects of fisheries on larger predators.

The high abundance of Fuegian sprat and lobster krill also makes these species attractive candidates as future fishery resources (Diez et al. 2018; García Alonso et al. 2020). However, particularly given the lack of available knowledge of these species, any commercial exploitation requires careful consideration (Cury et al. 2011; Smith et al. 2011; Essington et al. 2015). Poorly managed removal of lower trophic level pelagic fish is widespread (Smith et al. 2011), and has also occurred on the Patagonian Shelf (anchovy *Engraulis anchoita* fishery; Skewgar et al. 2007; southern blue whiting fishery *Micromesistius australis*; Laptikhovsky et al. 2013). Forage fish like Fuegian sprat often undergo large and rapid population fluctuations (Schwartzlose et al. 1999; Lindegren et al. 2013), and as schooling fish, they are particularly vulnerable to exploitation by modern mass-capture fishing technologies (Guerra et al. 2020). Removal of lower trophic level species can also be particularly concerning for “wasp-waist” systems where a large part of the plankton production is funnelled through a small number of lower trophic level species (Cury et al. 2000), as is the case for the Patagonian Shelf (Laptikhovsky et al. 2013; Riccialdelli et al. 2013). To gain improved knowledge of the ecosystem health and variability in support of ecosystem-based fisheries management and higher predator conservation, fisheries managers should give higher priority to the strategic and long-term monitoring of lower trophic level species.

Implications of climate change on natural prey

Climate change can affect ecosystems through changes in primary production, increase in temperature and ocean acidification, and alteration of ocean circulation (Brierley and Kingsford 2009). These oceanographic changes can impact the abundance, growth rate and distribution of organisms, thus altering their availability to predators (e.g. Antarctic krill *Euphausia superba*; Flores et al. 2012; Mills et al. 2020), and changing the responses of ecosystems to fishing (Blanchard et al. 2012). Therefore, any assessments on fisheries impacts should not be made in the absence of due consideration to climatic implications (Cleeland et al. 2021). This is particularly relevant for heavily fished ecosystems, such as the Patagonian Shelf Large Marine Ecosystem, which are typically less resilient to climate change compared to unexploited systems (Blanchard et al. 2012; Zhao and Li 2022).

There is evidence of a changing climate on the Patagonian Shelf (Quillfeldt and Masello 2013; Franco et al. 2022). Given the close association of the local food web with the dynamic water masses across the Patagonian Shelf (e.g. Acha et al. 2004; Arkhipkin et al. 2013b), changes in temperatures or currents could affect the distributing of both prey and predators (Diez et al. 2018; Sojitra et al.

2022). Long-term monitoring and physiological experiments could offer useful insight into species-specific climatic responses, and therefore assist with assessment of ecosystem resilience and response to a changing environment (van der Grient et al. 2023).

6.2.3. The importance of discards

Our results from **Chapters 3 and 4** complement previous findings that discards contribute relatively little to the overall energy budget of BBA during breeding (Thompson 1992; Thompson and Riddy 1995). Nonetheless, we demonstrated, for the first time, environmental-dependent levels of discard consumption (**Chapter 3**), and provided evidence for inter-colony dietary differences that are consistent over time (**Chapter 4**; Thompson 1992; McInnes et al. 2017a). Overall, this highlights the critical value of long-term monitoring, and obtaining representative data in space and time for robust inference at the population level.

Variation in discard consumption

Besides annual variation, seabird diets are also influenced by a whole range of intrinsic and extrinsic factors such as season (e.g. Navarro et al. 2009), life-history stage (Votier et al. 2008b, 2011; Riotte-Lambert and Weimerskirch 2013), or sex (Navarro et al. 2009, 2010) – and their interactions with each other and with fisheries (Gianuca et al. 2019) – as a result of the different constraints and demands faced by these groups (Barrett et al. 2007; Riotte-Lambert and Weimerskirch 2013; Campioni et al. 2016). At present, knowledge on BBA dietary requirements on the Patagonian Shelf is almost exclusively limited to breeding adults and chicks (**Chapters 3 and 4**; Thompson 1992;

Granadeiro et al. 2014; McInnes et al. 2017a, b). Information on BBA diet during the cryptic seasons and life-history stages (i.e. when birds are not easily accessible at their colonies, such as during non-breeding, or during juvenile years) is more scarce (but see e.g. Bugoni et al. 2010; Mariano-Jelicich et al. 2013 – winter; Colabuono and Vooren 2007 – winter; juveniles; Campioni et al. 2016 – juveniles). This knowledge gap prevents a more comprehensive understanding of discard consumption by BBA, therefore constraining our understanding of their response to natural and anthropogenic pressures and changes in the environment such as through discard management (Votier et al. 2008a; Bicknell et al. 2013). It also limits quantitative assessments of food consumption by BBA at an ecosystem level (Barrett et al. 2007).

There is evidence that discard consumption by adult BBA on the Patagonian Shelf is higher during non-breeding than during breeding (Bugoni et al*.* 2010; Mariano-Jelicich et al*.* 2013; although see e.g. Granadeiro et al. 2014, and elsewhere, e.g. Petersen et al. 2008). It remains unclear how diet during non-breeding affects demographics, such as in relation to adult survival probability and breeding performance in the subsequent summer. In some wild birds, supplementary food during winter can reduce the risk of starvation, increasing survival and altering movements and reproductive performance (Robb et al. 2008; Cohen et al. 2014). However, discards are predominantly of inferior nutritional quality compared to natural prey of piscivorous marine vertebrates (Jodice et al. 2006; Grémillet et al. 2008), and whilst this may sustain adult survival, a low-quality diet during non-breeding (and breeding) periods can have negative carry-over effects on reproductive success in some seabirds (Sorensen et al. 2009; Cohen et al. 2014).

Further, it is possible that discards affect seabird demographics by influencing the survival of juvenile and immature birds. Younger birds are generally less experienced foragers (e.g. Riotte-Lambert and Weimerskirch 2013) and may be at an elevated risk of bycatch (Weimerskirch et al. 2006; Afán et al. 2019). On the flipside, with their survival heavily constrained by their ability to learn to find food, access to large quantities of discards may artificially increase their survival to recruitment (Oro et al. 2008; Bicknell et al. 2013; but see e.g. Hudson and Furness 1988 regarding their competitive disadvantage as scavengers). In long-lived seabirds like albatrosses, immatures form a substantial component of the population (>50%; Votier et al. 2008a, 2011), and population dynamics can be highly sensitive to juvenile survival (Ventura et al. 2021a). The potential effects that supplementary foods from fisheries waste have on BBA population dynamics therefore warrants further attention.

Challenges and solutions for studying discard consumption

Studying seabird diet and discard consumption when birds are not at their colonies presents obvious challenges. Sampling vessel-following or bycaught birds is an option (e.g. Colabuona & Vooren 2007; Bugoni et al. 2010; Mariano-Jelicich et al. 2013; Jimenez et al. 2017). However, this can introduce important bias when using stomach content analysis where typically only the last meal is assessed (Jiménez et al. 2017), or when using stable isotopes from individuals that specialise in discards (Votier et al. 2010; although see Granadeiro et al.

2014). For example, Colabuono and Vooren (2007) found important diet differences in longline-caught and beached albatrosses.

Using stable isotopes from feathers (e.g. Granadeiro et al. 2014) offers a solution for colony-attending demographic groups, as feathers collected at the colony during summer can reveal information on winter or pre-breeding diet (e.g. Kowalczyk et al. 2015). However, it is important to know the moulting chronology to understand the time period assessed, and this can vary even within a given species (e.g. BBA; Catry et al. 2013a). Further, as with all stable isotope studies, great care is required to obtain a meaningful prey field against which to compare the isotopic signature of the diet, as the baseline can shift in time and space (Bugoni et al. 2010; Quillfeldt et al. 2015). This relies on accurate knowledge of foraging distributions during the period investigated.

Aside from juggling the challenges and biases of various dietary methodologies available, if the aim of the study is to quantify discard consumption, it is important to accurately categorise diet as such. For our studies (**Chapters 3 and 4**), we used a wide range of criteria to categorise a prey item as discards or natural prey, including morphological cues such as prey size and visual evidence of factory processing procedures. However, in some cases, there can be uncertainty, especially if the discarded fish is a natural prey of albatrosses (e.g. southern blue whiting, Thompson 1992; Cherel et al. 1999; jellyfish, McInnes et al. 2017a).

New technology can help overcome the limitations of conventional methods. For example, animal-borne video recorders are increasingly used as a complementary approach to provide copious amounts of novel information on top predator foraging behaviours (e.g turtles, Heithaus et al. 2002; cetaceans, Calambokidis et al. 2007; pinnipeds, Parrish et al. 2000; penguins, Ponganis et al. 2000), and can be particularly valuable in areas where expected fishery overlap and discard consumption is high (Votier et al. 2013; Carneiro et al. 2022). Visual information from the immediate environment around foraging predators can provide insight into previously unknown food-sources (Thiebot et al. 2017), or help distinguish between discards and natural prey (Michel et al. 2022). In addition, bird-borne cameras can provide information on foraging strategies including mechanisms for locating prey (e.g. Ponganis et al. 2000; Tremblay et al. 2014), feeding associations (Sutton et al. 2015; Michel et al. 2022) or kleptoparasitism (Handley et al. 2016), thus providing insight into how surface-feeding birds may access deep-water species like toothfish (Sakamoto et al. 2009) or deep-sea squid (Nishizawa et al. 2018) in the absence of fisheries.

In the Falkland Islands, a pilot study using video cameras attached to BBA revealed imagery of albatrosses actively feeding on jellyfish in the absence of vessels and in association with fur seals. Further footage showed albatrosses pursuit diving after a bait ball (presumably Fuegian sprat; A. Kuepfer unpubl. data). Limited battery life was a key constraint in this study, particularly since albatrosses undertake long foraging trips with relatively short foraging bouts (Weimerskirch and Guionnet 2002). Future technological advances may provide solutions, and these could be explored.

6.2.4. Foraging distribution and fishery overlap

Chapter 5 provides novel broad-scale insight into key foraging areas, and the fishing fleets with which breeding BBA most likely interact during egg incubation and chick brooding. This offers a tool for at-sea risk assessment, and lays important foundations for finer-scale analyses of vessel interactions (see e.g. Croxall et al. 2013; Torres et al. 2013b, a).

Tracking in space and time

Chapter 5 highlights the importance of multi-year and multi-colony tracking data to detect spatio-temporal variations and trends in seabird foraging locations. In particular, movement flexibility driven by extrinsic factors (such as climatic processes or changes in fishing activities) may only be measurable over large time scales (Sojitra et al. 2022; Beal et al. 2023). While fisheries do not appear to govern BBA foraging distributions during breeding (**Chapters 3, 4, and 5**; Catry et al. 2013b), climate change could change this in the future (Barbraud et al. 2012; Pardo et al. 2017). For example, shifts in albatross distribution mediated by changing physiological or biological oceanographic parameters (Sojitra et al. 2022) or wind patterns (Weimerskirch et al. 2012; Gianuca et al. 2019) could increase seabird-fishery resource competition and overlap in target areas (Cury et al. 2011; Sydeman et al. 2021a). Continuing long-term tracking can help detect potential changes in foraging distributions, and simultaneously offers a monitoring tool for environmental change across large oceanic areas.

Besides long-term tracking, it is also useful to understand the factors that drive seabird distributions. Modelling seabird locations as a function of habitat preference, accessibility, competition from neighbouring colonies and fisheries presence can help predict the response of seabirds to future climatic- or anthropogenic-mediated changes (e.g. Wakefield et al. 2009a; Catry et al. 2013b), and also allow extrapolation of distribution patterns to untracked colonies (Ronconi et al. 2022). This offers a potential solution for colonies such as Beauchêne Island, which form a critical component of the total population, but where access for research is difficult.

At present, the vast majority of available tracking data for BBA originates from the breeding season, both in the Falkland Islands (**Chapter 5;** Catry et al. 2013; Ponchon et al. 2019) and elsewhere (Petersen et al. 2008; Wakefield et al. 2011, 2012). Temporal bias in tracking studies is widespread in seabird research (Bernard et al. 2021). However, besides the external drivers mentioned above, offshore movements of seabirds are also influenced by a range of intrinsic factors such as breeding cycle (incubation/ chick-guard/ post chick-guard/ non-breeding; Ponchon et al. 2019), breeding status (failed/ successful; Ponchon et al. 2019) or age (juveniles/ immature/ adults; Carneiro et al. 2020). Distribution segregation within such groups may even vary across populations. For example, BBA demonstrate sex-based foraging segregation at South Georgia (Phillips et al. 2004) and Kerguelen (Desprez et al. 2018), but not in the Falkland Islands (Ponchon et al. 2019; Paz et al. 2021). Conversely, juvenile and adult BBA forage in similar locations at Macquarie Island (Cleeland et al. 2019) and the

Benguela Upwelling System (Petersen et al. 2008), but not on the Patagonian Shelf (Paz et al. 2021).

Unbiased tracking of the distribution of BBA at all life-history stages can lead to important areas of fisheries overlap being missed (Carneiro et al. 2020). For example, in line with age-based differences in foraging areas noted by Paz et al. (2021), juveniles represent 97% of bycaught BBA in longline fisheries off southern Brazil (Neves and Olmos 1997). Juveniles and immatures form a critical but vastly understudied component of seabird populations (Votier et al. 2008a), and knowledge of their at-sea risks are essential for understanding the true influence of fisheries on seabird populations (Gianuca et al. 2017; Sherley et al. 2017; Afán et al. 2019). Therefore, alongside long-term tracking studies, an important next step in understanding seabird-fisheries interactions involves targeted tracking efforts of cryptic life-history stages and outside the breeding season (Bernard et al. 2021).

Scale-dependent overlap

Broad-scale seabird-fishery overlap does not always equate to interaction – and interaction does not always equate to bycatch risk. For example, fine-scale analysis of wandering albatross movements showed that only 23% of all vessel encounters involved vessel visits (Carneiro et al. 2022; see also e.g. Granadeiro et al. 2014). In the Falkland Islands, wandering albatross interact extensively with the longline vessel in some areas; however, due to effective mitigation measures, the bycatch risk is negligible (A. Kuepfer pers. obs.; Falkland Islands Government 2021). Therefore, in areas and seasons where broad-scale spatio-temporal overlap is identified, there is value in assessing fine-scale overlap to gain more accurate indices of interactions that can help fine-tune management measures (Torres et al. 2013b, a).

Various technologies and methodologies exist for establishing fine-scale seabird-fishery overlap. For example, Granadeiro et al. (2014) used time-stamps of vessel positioning (VMS) with BBA tracking and activity data. Carneiro et al. (2022) integrated information from bird-borne GPS and vessel detection loggers. Votier et al. (2013) and Clark et al. (2022) used bird-borne video cameras and GPS-acceleration-dive loggers to reliably quantify the extent to which foraging activity occurs in the presence or absence of vessels.

Technological limitations and progress

Both broad- and fine-scale analyses of seabird-fishery interactions depend on the availability of accurate vessel location data. Access to VMS data is generally more restricted than AIS (Hinz et al. 2013). However, AIS can be intentionally turned off or manipulated (Ford et al. 2018). Using satellite data, Oceana (2021) found that between 2018 and 2021, over 600,000 hours of vessel activities remained hidden at the Argentine EEZ-high sea border due to the disabling of the AIS. Accuracy can vary substantially between fleets and individual vessels. For example, Shepperson et al. (2018) compared VMS and AIS data of individual scallop fishing vessels in the English channel, and found 45–99% of AIS data were missing. In the Southwest Atlantic, the use of AIS is high for vessels flagged to Argentina, Uruguay and the Falkland Islands, and also by foreign fleets fishing across the Patagonian Shelf (Arrizabalaga et al. 2019). However, further north, inside the Brazilian EEZ, AIS provides a poor assessment of fishing activity due to low use and poor detection of AIS from smaller vessels (Arrizabalaga et al. 2019).

Since June 2022, GFW has made available a global layer of radar detections derived synthetic aperture radar (SAR). This new technology does not rely on cooperative tracking systems like AIS and VMS, and aims to offer improved vessel tracking abilities. Future research on seabird-fishery interactions should investigate this technology.

Technological limitations also apply to seabird tracking devices. The balance of battery life, weight/size and accuracy of devices is a major reason for the temporal bias in tracking studies discussed above. The majority of technology used in ecology and conservation stems from adaptations of off-the-shelf devices and technology, which often fail to meet the specialised needs of scientists (Hahn et al. 2022). Further technological advances are needed to provide for more affordable and readily available solutions to studying cryptic life-stages and seasons. Scientists, in partnership with the industry and supported by research funding bodies, should focus expertise and funds into designing the next generation of electronic tags (Bernard et al. 2021; Hahn et al. 2022). In particular, it is important to find ways to provide longer-term energy to the electronic loggers, such as through solar energy or a chemical reaction with seawater (Kim et al. 2016).

6.3. Black-browed albatross – fisheries interactions: Summary of recommendations for future management and research

In view of further improving our understanding and management of seabird-fisheries interactions, we make the following recommendations:

- Effective bycatch mitigation measures are an important component of sustainable fishing practices, and reducing vessel attraction through discard management offers a tested solution for trawl vessels. However, managers should ensure the implementation of appropriate vessel-specific mitigation, and assess their use and effectiveness through adequate observer effort.
- Lower and mid-trophic species are integral to the marine food web. Critical knowledge gaps relating to their abundance and distribution on the Patagonian Shelf (and elsewhere) require urgent addressing to better understand their response to environmental change and fishing pressure, and to thereby help improve our understanding of potential repercussions on higher trophic predators like BBA.
- Cryptic life-history stages and seasons form a critical component of seabird population dynamics, but they remain understudied. Future research should focus on improving our knowledge of these periods and traits to allow for a more comprehensive and non-biased understanding of seabird-fishery interactions and response to environmental change.
- Long-term datasets are invaluable for understanding patterns and relationships in a complex and changing environment, and long-term monitoring of BBA foraging behaviour should continue.

- Seabird-fishery overlap is scale-dependent. Analyses at multiple scales, through use of complementary methodologies should be applied to fully comprehend the spatio-temporal variation in seabird-fisheries associations.
- Affordable and available technology is an important limiting factor for studying cryptic life-history stages and seasons, and there is a need for enhanced cross-discipline collaborations and expanded funding networks that allow rapid technological advances suitable for conservation science.

6.4. Conclusion

To conclude, this thesis provides new insights into fisheries interactions on the Patagonian Shelf by the world's largest population of BBA. The results could only be achieved by combining a range of long-term and high-quality data, and through effective collaboration between scientists, industry and government. Our findings have implications for fisheries and conservation management, and highlight the importance of interdisciplinary collaborations across and beyond national jurisdictions.

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