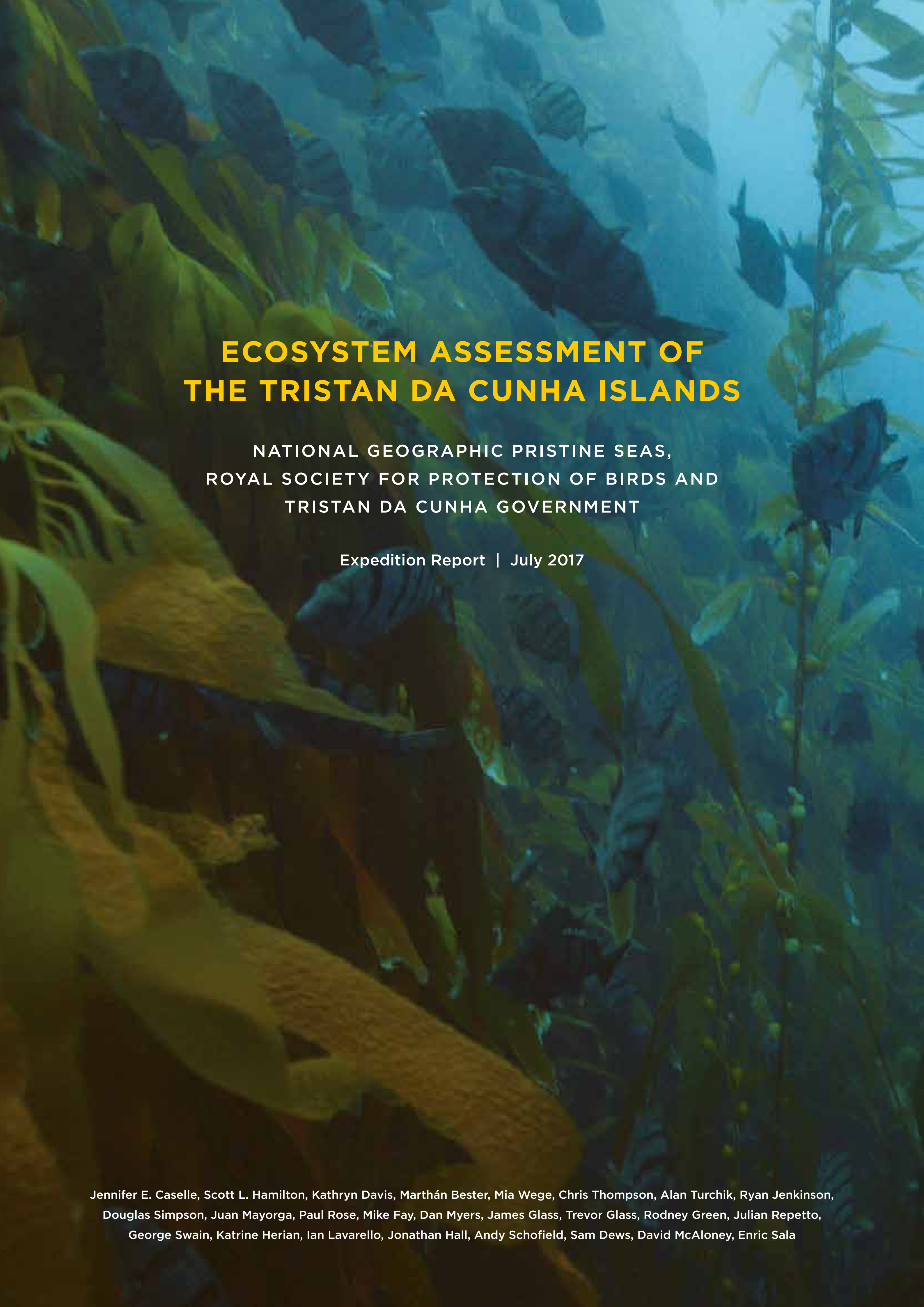

ECOSYSTEM ASSESSMENT OF THE TRISTAN DA CUNHA ISLANDS

NATIONAL GEOGRAPHIC PRISTINE SEAS, ROYAL SOCIETY FOR PROTECTION OF BIRDS AND TRISTAN DA CUNHA GOVERNMENT

Expedition Report | July 2017

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EXECUTIVE SUMMARY

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Tristan da Cunha is a remote group of islands in the South Atlantic, situated approximately 2,700km from South Africa and 3,700km from the nearest shores of South America. The northern island group is composed of Tristan da Cunha, Inaccessible and Nightingale Islands while the southernmost and most isolated island, Gough, lies 380km to the south southeast. Tristan da Cunha is part of the United Kingdom Overseas Territory of Saint Helena, Ascension and Tristan da Cunha. The wide spaces between the islands in the group give Tristan da Cunha a very large Exclusive Economic Zone (EEZ) of approximately 754,000km^2 which includes a broad range of oceanographic features and offshore habitats, such as seamounts. The Tristan Islands sit at the confluence of two major ocean currents, with a resulting temperature boundary or front (Subtropical Front). Consequently, Gough Island sea surface temperature is on average 3–4°C colder than the northern islands and likely experiences enhanced nutrient availability.

The Tristan archipelago is a hotspot of terrestrial endemic biodiversity. Both Gough and Inaccessible Islands have been given UNESCO World Heritage Site status for their near-pristine environments and unique wildlife populations. The marine environment has been less well studied. The marine environment has lower biodiversity than terrestrial habitats but high biomass of several important species, including the Tristan rock lobster (*Jasus tristani*) that is the target of a sustainably certified fishery. The fishery provides an important source of revenue for the islanders as well as providing transportation to and from the island via the larger fishing vessel. While all of the Tristan island's ecosystems appear relatively intact and healthy, there are numerous threats, including introductions of non-native and invasive species, shipping and illegal fishing. Finally, climate change effects on this precarious system are unknown at present but could potentially alter Tristan's marine and terrestrial ecosystems in dramatic ways.

In January–February 2017, National Geographic's Pristine Seas project, in collaboration with the Royal Society for the Protection of Birds (RSPB) and the Tristan da Cunha Government (including the Fisheries and Conservation Departments) conducted a 21-day expedition to Tristan da Cunha, Nightingale, Inaccessible and Gough Islands. The primary goals of the expedition were to conduct comprehensive quantitative surveys of the health of the archipelago's largely unknown marine environment to assist the Government and people of Tristan da Cunha in planning marine protection. The results of the expedition highlight the unique marine ecosystem of this archipelago, particularly the pelagic and deep-sea environments, which were virtually unstudied and documented scientifically. The work presented in this report is meant to complement ongoing research at Tristan da Cunha and provide a springboard to help inform the Tristan Government about potential protection schemes that protect both important fisheries and unique biodiversity of the archipelago.

NEARSHORE ROCKY REEFS AND KELP FORESTS

In general, the Tristan da Cunha Islands have very low faunal biodiversity in nearshore marine habitats, likely due to extreme isolation and lack of any island stepping stones with appropriate habitat between the islands and any continental landmass. We conducted quantitative SCUBA surveys focused on the abundant, conspicuous and strongly interacting members of the kelp forest communities including fishes, kelps, lobsters and any organisms that might interact with lobsters. We targeted kelp-dominated boulder reefs and found that within this habitat type, there was very little variation in substrate type or physical relief with most areas containing high relief, boulders covered in kelps or sandy areas between the boulder reefs.

We measured fish biomass (tonnes/ha) and density (no./100m^2) for all observed fish species at each island. Total fish biomass ranged between 1.5–2.75 tonnes/ha, which indicates a healthy fish assemblage. Five finger (*Nemadactylus monodactylus*) were dominant both numerically and in terms of biomass at all four islands. Interisland differences in biomass were driven by yellowtail amberjack (*Seriola lalandi*) which were abundant at the northern islands but absent at Gough. At Gough, where most fish were larger on average, false jacopever (*Sebastes capensis*) and telescope fish

(*Mendosoma lineatum*) contributed to the differences in the biomass estimates between island groups. The strong differences in species composition between the northern islands and Gough drove statistically significant variation in both biomass and density at the island scale.

Lobster (*Jasus tristani*) biomass and density both varied significantly among islands, sites within island, and biomass also varied by depth. Interestingly, the density and biomass patterns were inversely related across islands. For example, lobsters were most numerous at Tristan and Inaccessible Islands but biomass was largest at Gough and Nightingale. This was due to size structure differences, with larger lobsters at Gough and Nightingale and smaller lobsters at Tristan and Inaccessible. Lobster carapace length averaged 8.1cm at Gough compared to 6.7, 7.5 and 6.6cm at Inaccessible, Nightingale and Tristan respectively. Several fish species including false jacopever and five finger were also much larger at Gough than the other islands.

The major habitat forming species, giant kelp (*Macrocystis pyrifera*) was the least variable among islands with no consistent differences in either plant or stipe density, even between the northern islands and Gough. We did find significant site to site variation and depth variation for kelp plant density and stipe density, however, the range in site to site variation for kelp was less than that for fishes and invertebrates. We found a significant effect of depth on giant kelp which was due to much greater density of both plants and stipes at the shallow zone at Gough Island relative to the deep zone. At the northern islands, this depth stratification was not apparent. Pale kelp (*Laminaria pallida*) was the other major component of the kelp forests at Tristan islands. Unlike giant kelp, pale kelp densities varied significantly across islands, with this warm water tolerant species showing higher densities at the northern islands, relative to colder Gough Island.

Kelp forest communities at the Tristan Islands are not species rich and the trophic structure is simple compared to more diverse kelp forests in other parts of the world. Surprisingly, we found no relationship between giant kelp (primary producer) and urchins (grazer) but a strong inverse relationship between pale kelp and urchins. During survey and non-survey SCUBA dives, we never observed urchins grazing or resting on giant kelp but we did observe urchins both on the holdfasts and blades of the pale kelp. Lobster density and biomass also showed no clear relationship with urchins (their putative prey) or octopus (their putative predators) but did show similarly low levels of variation among islands as giant kelp.

We surveyed the wreck site of the MV Oliva at Nightingale. Surveys took place on natural habitat just inshore of the wreck and deeper habitat directly over much of the structure of the wreck. In general, we saw no dramatic differences relative to other sites at Nightingale (apart from the structure of the wreck itself). The community structure analysis of the benthic community did distinguish this site from the other sites but this was due to high abundance of barnacles (common on the structure of the wreck) and pale kelp and very low abundance of giant kelp and urchins. Most of the remaining structure is covered completely with benthic turfing and foliose algae and pale kelp (*L. pallida*).

DEEP SEA

National Geographic deep-water drop cameras were deployed at a total of 23 sites across all islands to depths ranging from 164m to 1,414m. We observed a total of 21 species or species groups. Common species included: bluenose (*Hyperoglyphe antarctica*); lantern sharks (*Etmopyerus* sp.); bluntnose sixgill shark (*Hexanchus griseus*); grenadiers (family Macrouridae); deepwater cod (*Physiculus karrerae*); and cutthroat eels (*Synaphobranchus* sp.). Interestingly, a southern elephant seal (*Mirounga leonina*) was observed on a camera drop at 190m depth. The benthos varied from sand/mud substrate to moderate and high rock relief and the common biotic habitat forming organisms included whip corals, gorgonians, sea pens and sea fans. We observed a non-significant trend for higher species richness on rocky substrate and a significant positive effect of rock habitat on total N_{max}. We characterized drop camera stations into two different depth zones (0–750m and > 750m depth) and found that the deep-water fish assemblages differed significantly between the shallower and deeper strata, but did not differ among islands. Communities shallower than 750m were characterized by the presence of soldiers (family Sebastidae), bluefish, wreckfish (*Polyprion oxygeneios*), small seabass (*Lepidoperca coatsii*) and roughy (*Beryx decadactylus*), while communities deeper than 750m were composed of deepwater and antimora cods (family Moridae), cusk eels (Ophidiidae), rattails (Macrouridae), cutthroat eels (*Synaphobranchus* sp.), and lantern shark (*Etmopyerus* sp.). Species richness and total N_{max} did not differ between depth zones.

PELAGIC ZONE

Mid-water baited remote underwater video systems (BRUVS) were deployed at 27 sites across all islands at a distance of approximately 5km from shore. We recorded 402 individual pelagic fishes, marine mammals, birds and turtles, representing 14 species from 11 families. The most abundant species was a horse mackerel (*Trachurus* sp. likely *T. longimanus*) with 198 individuals observed across 44.5% of sites and the yellowtail amberjack (*Seriola lalandi*) with 69 individuals observed, although this species was observed primarily in large schools and thus was only present across 7.4% of sites, all of which were in the northern islands. Blue sharks (*Prionace glauca*) were observed at 55.6% of sites both in the northern group and at Gough Island, with 23 individuals observed. We recorded a number of very small blue sharks, at a size reported for newly born individuals. We also recorded a new record for Tristan da Cunha (porbeagle shark, *Lamna nasus*), and the first video evidence of two Shepherd's beaked whales (*Tasmacetus shepherdi*) off Inaccessible Island. One Shepherd's beaked whale was measured at 2.5m. This is much less than their maximum reported size of 7.1m, suggesting that it may be a young individual.

OCEANOGRAPHY

We measured various properties of the water column at 11 sites using a profiling CTD attached to the deep-water drop cameras. As expected, temperature and salinity were different at Gough relative to the northern islands; Gough waters were colder and less saline. The depth at which both temperature and salinity equilibrated among islands was approximately 600–750m, the same depth range at which we saw a divergence in community structure from the deep-sea drop cameras. Patterns of dissolved oxygen were more difficult to interpret and less variable among islands.

PINNIPEDS

Research undertaken on this expedition is a continuation of a well-established and long-term research program on the Subantarctic fur seal (*Arctocephalus tropicalis*) conducted by Marthán Bester, Mia Wege and colleagues. Accomplishments on this expedition included: collection of scats and tissues for diet analyses, deployment of satellite tags for tracking movements of lactating females, and partial island counts. Counts of pups at Gough island were compared to a baseline survey done in January 1978. Over the approximately 40 year period, uncorrected pup numbers at the Tumbledown Beach area (n = ~24 vs. 591), Buttress Rock to Dell Rocks (n = a few at most vs. 724) and the north-east beaches north of Deep Glen (n = 0 vs. 5,199) have increased exponentially, while other small (Seal Beach, n = 1 vs. 2) and large open beaches (Reef Point to Deep Glen, n = 0 vs. 16; Capsize Sands, n = 0 vs. 8) have shown virtually no increases.

SHARKS

The status of sharks at Tristan da Cunha Islands is unknown. Several species have been observed in Tristan's offshore waters including blue sharks (*Prionace glauca*), shortfin mako sharks (*Isurus oxyrinchus*) and hammerheads (not observed on this expedition). For the first time in Tristan's waters, we tagged and will track movements of blue sharks and shortfin mako sharks in and around the archipelago and beyond. Using miniPAT pop-up archival transmitting tags we tagged three female blue sharks at Gough Island that ranged in size from 189 to 217cm TL. One shortfin mako shark was tagged following the expedition and an additional four tags were left with the Tristan Fisheries and Conservation Department heads and will be deployed near Tristan. Movement data will be available when the tags release from the animals around mid- to late-August, 2017.

INVASIVE MARINE SPECIES

The wreck of an oil platform in 2006 on the south coast of Tristan introduced a non-native fish, the South American silver porgy (*Diplodus argenteus argenteus*), that had spread throughout Tristan Island at the time of our surveys. On this expedition, we documented its spread to Inaccessible Island where it was present at four of the seven sites sampled and at all depths of SCUBA surveys (from 20 to 10m). While we did not record this species at Nightingale Island, the proximity to Tristan and Inaccessible make it likely that the porgy will spread to that island at some point in the future. We also performed targeted collections for diet studies using gut contents, and stable isotopes as well as age and growth measurement. Preliminary results of analysis of gut contents of the Porgy showed that the species is almost solely herbivorous at Tristan Island. The only evidence of invertebrates appeared to be epiphytic organisms, likely living on the ingested algae. We found a single larval lobster in the gut of one porgy. However, we also sampled guts of other fish species and did find recently ingested juvenile lobsters in the guts of both the false jacopever (*Sebastes capensis*) and the five finger (*Nemadactylus monodactylus*). Stable isotope and age and growth studies are ongoing.

FISHING VESSELS

Using data from Global Fishing Watch (GFW; 2014–2016), we examined the magnitude and spatial distribution of the behavior and fishing effort of industrial fishing vessels in and around Tristan's EEZ. The results reveal that up to 253 vessels from 12 different nationalities transited the northern region of the EEZ; spending a total of 1495 vessel days in the last three years within the EEZ. However, we see likely fishing activity on only 7% of those days, mostly by vessels from the UK and Japan operating in the eastern region of the EEZ. This suggest that there is little industrial fishing activity inside Tristan's EEZ, relative to the surrounding waters near seamounts to the southeast and southern boundary. In these adjacent waters, we observed 62 fishing vessels from eight different flag states that spent a cumulative 1,987 fishing days during the entire time period.

MICROPLASTICS

Given the increasing levels of plastic pollution of the oceans it is important to better understand the distribution and impact of microplastics in the ocean food web. We collected seawater samples at 19 locations, slightly offshore of SCUBA survey sites. Two samples were collected from the rock pools immediately adjacent to the Harbor at Edinburgh of the Seven Seas (i.e. the settlement) by the students of St. Mary's school. Microplastics were found in 15 of the 19 samples. The number of microplastic pieces per liter of water ranged from 0 to 2.3 (overall mean 0.83 pieces/L).

SUMMARY AND RECOMMENDATIONS

Tristan da Cunha Islands are a unique archipelago with healthy marine ecosystems — although with low species diversity, likely due to extreme isolation. This remote temperate archipelago provides one of the few places in the world to establish a baseline for unimpacted temperate systems. Quantitative data from the kelp forests was lacking while pelagic habitat and deep benthos were mostly unexplored prior to this expedition. We found that, despite an important commercial fishery for lobster and subsistence fishing for local islanders, marine habitats and biota appeared in very good condition. Biomass of fishes and lobster in particular were high. However, this unique marine ecosystem is not without potential threats: shipping traffic leading to wrecks and species introductions, pressure to increase fishing effort beyond sustainable levels and climate change all could potentially increase in the coming years. Currently, the low population density, difficult access to the marine environment and a proactive, well-managed lobster fishery provide a level of protection to nearshore habitats. However, offshore areas including seamounts would benefit from strong, enduring protection.

Specific recommendations include:

Due to Tristan's location in the South Atlantic Ocean, its EEZ includes a range of unique oceanographic features:

- To ensure a comprehensive protection regime, we recommend the creation of strongly protected marine reserves in both the northern and southern portions of the EEZ/MPA.

- We recommend the highest level of protection practicable for as many seamounts as possible throughout the EEZ, again making sure to include both northern and southern regions.

The nearshore waters of Tristan's Islands are healthy and sustain a well-managed lobster fishery with some take of finfish by local islanders for consumption and bait.

- Existing protection (from shore to 50nm) should be credited when designing additional protection schemes for Tristan and this area should remain permanently closed to foreign non-lobster vessels.
- We recommend that no additional spatial closures for lobster are needed as long as the fishery remains well managed (e.g. maintaining MSC certification). Monitoring and research on the fishery should continue.
- If nearshore spatial closures for finfish and other non-lobster resources are considered, due to low variation in habitat, they could be located in most locations around the islands and achieve similar protection of the habitat.
- Tristan Islands currently have a large amount of de facto protection, both temporally and spatially, but this could change with increased vessel capability or improved weather condition. We recommend a process be developed for regular assessment of access to fishing sites from Tristan islanders and non-locals.
- Future impacts from climate change and human pressure are increasing globally and might pose unanticipated challenges for Tristan's ecosystems and fishery.
- Future ocean warming and associated declines in productivity may affect nearshore ecosystems and fisheries. We recommend continued monitoring of temperature (*in situ* and remote sensing) combined with predictive modeling of climate change.
- Low levels of biological diversity in nearshore areas result in a simple, short-chain food web with very few key interacting species, making this system very sensitive to perturbations. We recommend future studies be conducted on predicting climate change effects for this region (from global models) at least for populations of the key species (giant kelp, lobsters, urchins), and major drivers (productivity, sea and land temperature, changes in storm frequency).
- Despite a documented expansion of the non-indigenous silver porgy (*Diplodus argenteus*) to Inaccessible Island from Tristan island, we found low density at most sites. Considering the density and the dietary habits (herbivory) of this species, we recommend that the populations continue to be tracked, but we would not prioritize active removal efforts at this time.
- Redistribution of species is one predicted effect of climate change. We recommend ongoing monitoring of species distributions and careful attention to the potential for development of new fisheries.

INTRODUCTION

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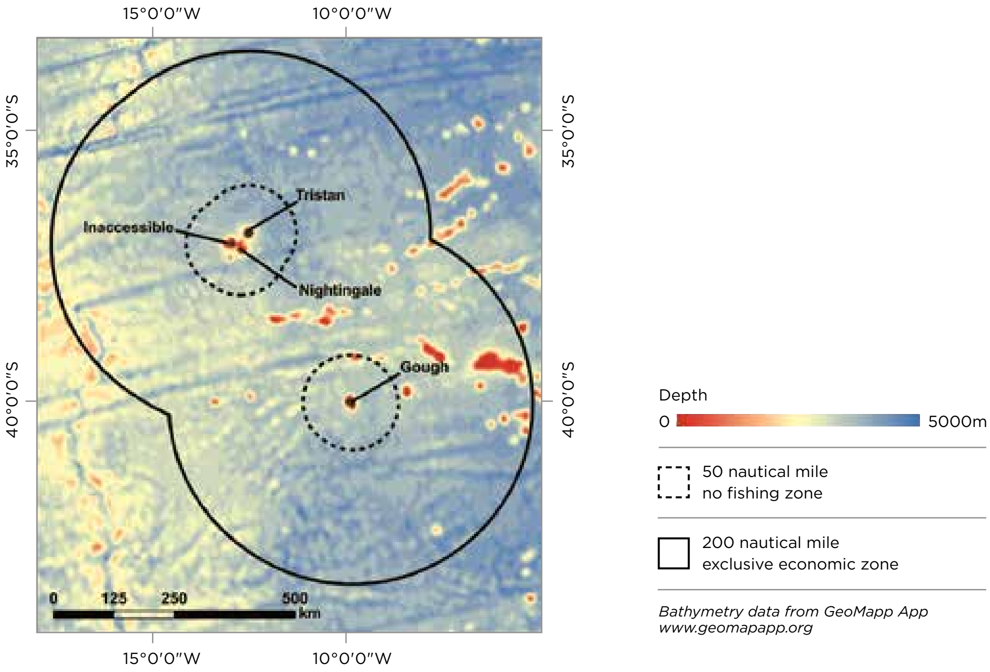
Tristan da Cunha is a group of islands in the South Atlantic, and also the name of the main island in that group. The archipelago lies over 2,700km from South Africa and 3,700km from the nearest shores of South America. The island of Saint Helena is the closest land, 2,400km to the north. This makes the Tristan archipelago one of the most geographically isolated island groups in the world. The northern group (37°04'S, 12°18'W) is composed of Tristan da Cunha, Inaccessible and Nightingale Islands. Inaccessible and Nightingale lie approximately 30km to the southwest and south of Tristan da Cunha respectively (Figure 1). The southernmost and most isolated island, Gough (40°19'S, 9°57'W), lies 380km to the south-southeast. Politically, the United Kingdom Overseas Territory of Tristan da Cunha is administered under the governorship of Saint Helena by a UK-appointed Administrator, and has an elected Island Council.

FIGURE 1.

Map showing location of Tristan da Cunha islands in the south Atlantic. EEZ is shown and seamounts within the EEZ.



Bathymetry data from GeoMapp App www.geomapapp.org

The Tristan group and Gough are volcanic islands formed from the outer slopes of the Mid-Atlantic Ridge (Heydorn and Lutjeharms 1980) and range in age from 200,000 years (Tristan) to around 18 million years (Nightingale) (Baker et al. 1962, Scott 2016). Tristan da Cunha Island is a near perfectly circular volcanic cone that rises abruptly from the Atlantic Ocean (with eruptions recorded as recently as 2004). The island is 12km in diameter at its widest point and over 2,060m at its highest peak, the summit of Queen Mary's Peak. A narrow coastal plain in the northwest of the island provides approximately 6km^2 of flat land. The population of 270 Tristanians live in a single settlement, 'Edinburgh of the Seven Seas', making Tristan da Cunha arguably the most remote inhabited island in the world.

The wide spaces between the islands in the group give Tristan da Cunha a very large Exclusive Economic Zone (EEZ) of approx. 754,000km^2, which includes a broad range of oceanographic features and offshore habitats, such as seamounts (Figure 1). The seabed around the islands is steeply sloping, reaching depths of more than 2,000m in less than a few kilometers from the coastline. Consequently, the majority of the seafloor within the EEZ is significantly deeper than 300m. The EEZ contains a number of seamounts including McNish, RSA, Crawford and the recently discovered Esk Guyot. Some of these rise to as shallow as 143m below sea surface. The Tristan Islands are located roughly at the boundary between the Southern ocean and the South Atlantic and sit at the confluence of two major ocean currents. The northern group (Tristan, Inaccessible and Nightingale) sits north of the Subtropical Front (STF) or convergence which is characterized by sharp differences in sea temperature and salinity (Doolittle et al. 2008). Gough sits on or below this front. Consequently, Gough Island sea surface temperature (SST) is on average 3–4°C colder than the northern islands and likely experiences enhanced nutrient availability. While the exact position and width of the

STF varies (Deacon 1982, Doolittle et al. 2008), these types of 'fronts' are locations of enhanced productivity (Lutjeharms et al. 1985, Laubscher et al. 1993) and often are hotspots for marine organisms such as sharks, pinnipeds, cetaceans, and seabirds due to aggregations of prey species (Etnoyer et al. 2004, Bakun 2006, Weng et al. 2008, Bost et al. 2009, Raymond et al. 2010, Block et al. 2011, Robinson et al. 2012, Queiroz et al. 2016). Fishery species also tend to aggregate at fronts (Polovina et al. 2001, Zainuddin et al. 2006), increasing the probability of direct (bycatch, entanglement) and indirect (competition for prey) negative interactions between fisheries and marine megafauna and seabirds (reviewed in Scales et al. 2014).

The Tristan archipelago is a hotspot of endemic biodiversity primarily in terrestrial systems and to a lesser extent in marine systems (although this may increase with further study and species identification (Scott and Holt *in prep*). The degree of endemism varies greatly among taxonomic groups, with marine algae showing the highest diversity (over 120 species present at the top islands) and the highest levels of endemism (Scott 2016). Other marine groups such as fishes, crustaceans and echinoderms are much less diverse, which is a common pattern on isolated islands. The Tristan rock lobster (*Jasus tristani*) is the most valuable commercial species, and is the target of an MSC-certified sustainable lobster fishery that provides over 80% of the island's GDP, employs over a quarter of the population at peak times, and provides an essential source of human and cargo transportation via fishing boats travelling to and from South Africa (see Fisheries below). The nearshore kelp forests (*Macrocystis pyrifera, Laminaria pallida*) that provide critical habitat for lobster and other species are of particular interest, especially as the northern group of islands experience summer temperatures that are near the limits of thermal tolerance for *Macrocystis* (Tegner et al. 1996, Steneck et al. 2002, Valdez et al. 2003). Offshore waters are also home to a number of pinnipeds, sharks and cetaceans (Bester 1990, Andrew et al. 1995, Best et al. 2009). Blue sharks (*Prionace glauca*) and shortfin mako sharks (*Isurus oxyrinchus*) are commonly observed in offshore waters while Broadnose sevengill sharks (*Notorynchus cepedianus*) inhabit the nearshore kelp forests (Andrew et al. 1995). Large and increasing numbers of Subantarctic fur seal *Arctocephalus tropicalis* breed at the islands and a very small southern elephant seal (*Mirounga leonina*) population exists at their only known breeding location at Gough Island (Bester et al. 2006 , Bester and Ryan 2007). Offshore waters also support populations of dusky dolphins (*Lagenorhynchus obscurus*), the rare Shepherd's beaked whale (*Tasmacetus shepherdi*), and are also known to be a nursery area for southern right whales (*Eubalaena australis*), (Best et al. 2009).

The terrestrial environment is home to seven endemic land bird species. It also provides key breeding sites for a large number of seabirds including three species of albatross (Andrew and Ryan 2007). The islands are also home to many species of endemic plants and invertebrates (Ryan 2007). Two of the endemic bird species, the Tristan albatross and the Gough bunting, are classified as Critically Endangered and small populations of several other species make them imperiled as well. Terrestrial biodiversity surveys (transects) were done by Mike Fay as part of the Pristine Seas expedition and are covered in a separate report (Fay 2017).

There are three fisheries at Tristan da Cunha: a MSC certified lobster fishery, a longline and trawl fishery for bottom fish such as bluefish, and a pelagic longline fishery for southern Bluefin tuna. The lobster fishery is the mainstay of the Tristan economy, while the other fisheries are limited in scope and conducted solely by foreign vessels via permits from the Tristan government. The lobster fishery is the only inshore fishery and provides 70–90% of the island's income, essential for economic independence for the islands. A variety of traps are used in the lobster fishery, ranging in size and materials from smaller hoop nets to large, steel framed "monster" traps (Glass 2014, Scott 2016). Total allowable catches (TAC) are set annually, are generally precautionary and at least in recent years have rarely been exceeded. Catch rate (CPUE) and size structure are measured annually but long term interpretation of data has been challenging because of many changes in gear types, vessel capabilities and seasonal changes over time. Recent catches average approximately 371 tons annually. The lobster fishery is licensed to a single company, Ovenstone Agencies, under a long-term concession. This unique situation gives a large amount of local control and oversight, in part resulting in MSC certification. Lobsters are processed onshore in a modern factory and shipped to buyers primarily in Japan and the US, although markets in Australia and Europe are emerging. Details on the regulations and current fishery can be found in Best et al. (2009), MacAlister Elliott and Partners Ltd. (2011), and Scott (2016). The pelagic longline fishery for southern bluefin tuna (SBT) is currently allocated to a single vessel in Tristan's EEZ. Southern bluefin tuna are managed as a single stock and while catches in Tristan's waters will count against the CCSBT limits of this critically endangered species (Collette et al. 2011), they are likely to be low and thus unlikely to have significant effects on the global conservation status of SBT. However, catches of SBT in Tristan's EEZ might affect local economics and may also have harmful bycatch effects (long-lines are known to catch sharks and seabirds — Brothers 1991, Francis et al. 2001, Anderson et al. 2011, Gallagher et al. 2014), Thus, fishing for SBT in Tristan's EEZ is unlikely to have significant effects on the global conservation status of SBT, but may have serious negative consequences for other Tristan species that rely on longline-free waters.

Both Gough and Inaccessible Islands have been given UNESCO World Heritage Site status for their near-pristine environments and vast wildlife populations. While all of the Tristan Islands ecosystems appear relatively intact and healthy, there are numerous threats, some urgent and some emerging. On both land and in the water, non-native and invasive species are of primary concern for these isolated islands. Mice and rats are a serious and urgent concern (Angel and Cooper 2006, Brown 2007). Plans are underway to conduct an eradication of mice on Gough Island (Tristan da Cunha Government and RSPB 2012) where the only substantial breeding population of the critically endangered Tristan albatross currently exists and is severely threatened by mouse predation on chicks. In the marine environment, the recent groundings of an oil rig in 2006 and a cargo ship in 2011 brought numerous non-native marine species as well as damage from spilled fuel. In particular, the oil rig stranding led to the introduction of a non-native fish, the South American silver porgy (*Diplodus argenteus argenteus*), that has become established around Tristan da Cunha. The cargo ship dumped

1,500 tons of heavy fuel into the water (along with 65,000 tons of soy cargo), killing over 3,000 rockhopper penguins (Scott 2016). The spill also closed the essential lobster fishery for the 2011/12 season and forced a reduction in the Total Allowable Catch (TAC) at Nightingale for that year. Increases in shipping traffic around the islands threaten the native ecosystems in the future. Legal and illegal longline fishing even outside of Tristan's EEZ threatens more mobile animals such as seabirds and sharks. Recent work has shown high overlap in space use of pelagic sharks and the longline fleet in the North Atlantic and it is likely that the South Atlantic shows similar patterns. There is also a high level of spatial overlap in foraging of the critically endangered Tristan albatross and longline fisheries, potentially causing substantial mortality (Cuthbert et al. 2005). Future threats include development of additional fisheries, both inshore and at the seamounts. Little is known about these habitats and the organisms occupying them, and any development of new fisheries must proceed with caution because seamounts are hotspots for pelagic species of conservation concern that can interact with fisheries (Worm et al. 2003, Morato et al. 2010), and seamount habitats and associated fishery species are highly vulnerable to degradation from fishing (Morato et al. 2006, Clark and Koslow 2007). Finally, climate change is the ultimate stressor on marine and terrestrial ecosystems with largely unpredictable effects in relatively unstudied systems such as Tristan da Cunha.

For a complete review of the physical setting, ecology, biology and conservation threats for the Tristan da Cunha archipelago, see Scott (2016).

OBJECTIVES

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The primary scientific goals of the National Geographic Pristine Seas project are to explore, document and protect the last wild places in the ocean. It is essential that we let the world know that these places exist, that they are threatened, and that they deserve to be protected. To this end, National Geographic Pristine Seas, in collaboration with the Royal Society for the Protection of Birds (RSPB) and the Tristan da Cunha Government (including the Fisheries and Conservation Departments) conducted a 21-day expedition to Tristan da Cunha, Nightingale, Inaccessible and Gough Islands in January–February 2017. Primary scientific goals of the expedition were to conduct comprehensive quantitative surveys of the health of the archipelago's largely unknown marine environment, and to assist the Government and people of Tristan da Cunha in planning for marine protection.

The core scientific research included:

- quantitative surveys of shallow flora and fauna using scuba diving surveys,
- documentation of pelagic (open ocean) communities using baited stereo-cameras,
- documentation of deep-sea habitats using specially built deep-water drop cameras,
- movement studies on sharks, birds and seals using satellite transmitters,
- basic ecological studies (e.g. food habits, growth) on key species in the marine ecosystems,
- documentation of the spatial spread and ecological function of a recently colonized, non-native fish species,
- documentation of the presence/absence of microplastics in the water,
- evaluation of the status of pinniped populations at the islands including quantitative surveys, and assessment of diet and condition.

The results of the expedition highlight the unique marine ecosystem of this archipelago, particularly the pelagic and deep-sea environments, which are virtually unstudied and documented scientifically, yet potentially impacted by shipping, fishing, climate change and other stressors. Strong partnership with the Tristan Island Conservation and Fisheries departments and Royal Society for the Protection of Birds (RSPB) will ensure ongoing communication and collaboration into the future. The work presented here is meant to complement ongoing research at Tristan da Cunha and provide a springboard to help inform the Tristan Government about potential protection schemes that protect both important fisheries and unique biodiversity of the archipelago. The intent of these results is to provide a scientific baseline to underpin further research that can help inform Tristan's future marine protection.

Expedition Partners:

- Tristan da Cunha Fisheries & Tristan da Cunha Conservation Departments
- Royal Society for the Protection of Birds (RSPB)
- The University of California, Santa Barbara (UCSB)
- Centre for Marine Futures, University of Western Australia (UWA)
- Moss Landing Marine Laboratories (MLML)
- Mammal Research Institute, University of Pretoria (UP)

RESULTS

RESULTS

Nearshore Kelp Forest Scuba Surveys

DIVERSITY AND HABITAT

In general, the Tristan da Cunha Islands have very low faunal biodiversity (species richness) in nearshore marine habitats, likely due to extreme isolation and lack of any island stepping-stones with appropriate habitat between the islands and any continental landmass. Taxonomic diversity and regional and global affiliations are currently being investigated through Darwin Initiative projects (Scott 2016). Consequently, we focused our quantitative surveys on the more abundant, conspicuous and strongly interacting members of the kelp forest communities, focusing particularly on lobsters and any organisms that might interact with lobsters.

These islands lack broad diversity in habitat types, with most nearshore areas comprised of high relief, boulder reefs covered in kelps or sandy areas between the boulder reefs. Prior to the expedition, we utilized LANDSAT imagery to create predictive maps of canopy forming giant kelp (see methods) around all four islands (Figure 2). These maps helped us to identify locations for sampling which were chosen both on the presence of kelp canopy and a roughly even distribution around each island (Figure 3).

Detailed methods for SCUBA surveys can be found at the end of this report. Briefly, we used standardized methods adapted from the Partnership for Interdisciplinary Studies of Coastal Oceans (PISCO; www.piscoweb.org). At each site four divers counted and estimated sizes for all fish species and the lobster (*J. tristani*) on four transects across two depth zones (10 and 20m). Conspicuous invertebrates and kelps were also counted and benthic substrate type and physical relief were measured.

FIGURE 2.

Predicted kelp distributions for (A) Tristan, (B) Inaccessible, (C) Nightingale, and (D) Gough islands. Green polygons represent kelp canopy observed from Landsat 7 or Landsat 8 images or predicted kelp occurrence based on other high resolution imagery or features seen in Landsat images. Data source: Tom Bell, UC Santa Barbara.

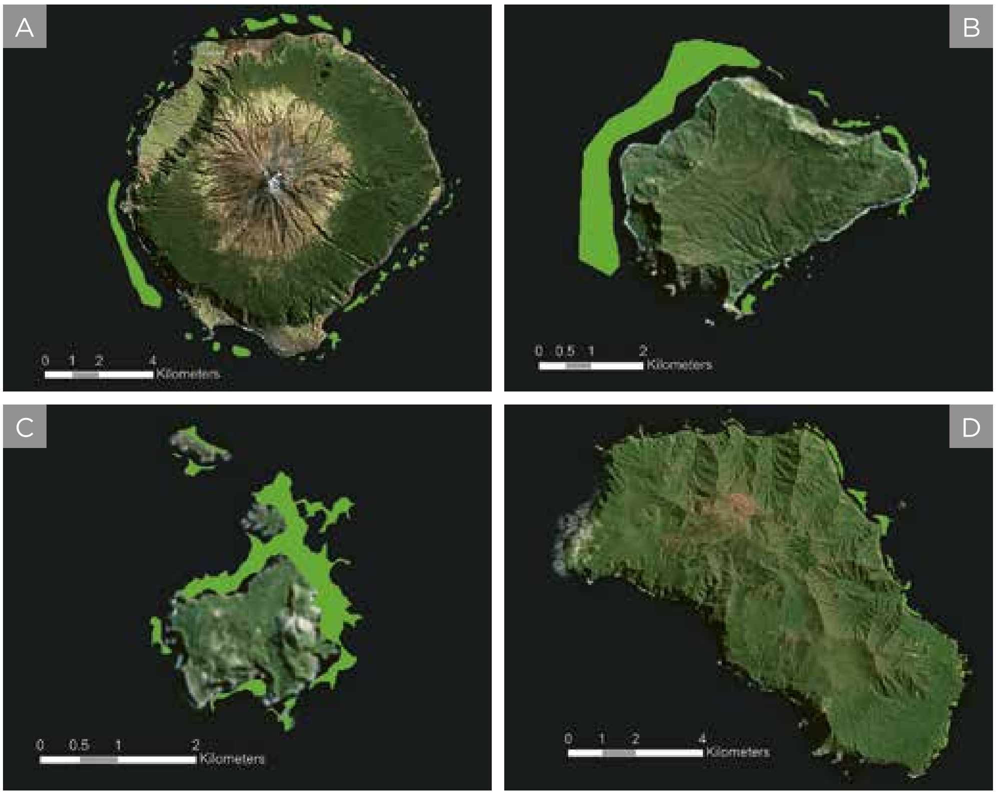
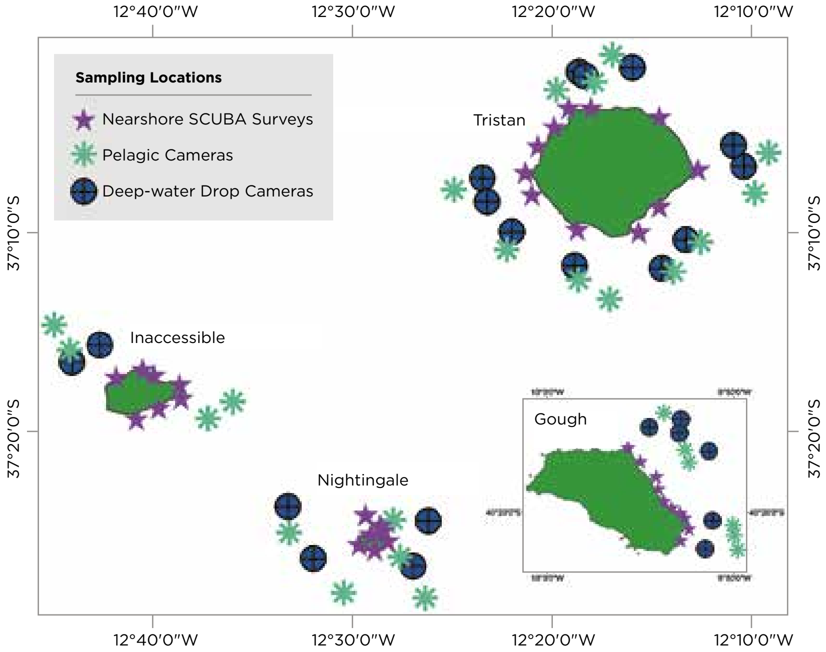


FIGURE 3.

Map of the study region depicting the locations of scuba surveys (n = 34), pelagic camera surveys (n = 26), and deep sea benthic camera surveys (n = 23) in the four islands of the Tristan da Cunha Islands group.

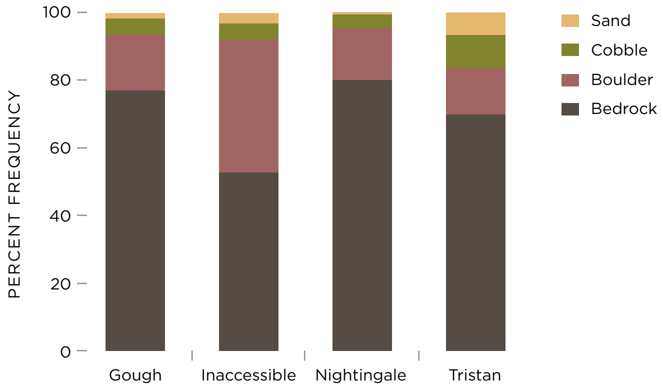


We targeted kelp-dominated boulder reefs for our surveys and even within this habitat type, there was very little variation in substrate type or physical relief (Figure 4) among islands. Inaccessible Island differed slightly from the other islands in the percent cover of small boulder (40% vs. 14%, 16% and 17% for Tristan, Nightingale and Gough) versus large boulder/bedrock (52% vs. 70%, 80% and 77% for Tristan, Nightingale and Gough). Only Tristan had appreciable cover of sand (6% vs. 4%, 3% and 1% for Gough, Inaccessible and Nightingale) but this was largely driven by a single site that had discontinuous reef. At all islands, the sand category of substrate was composed of relatively large grain, and volcanic sediment and patches of sand were small. Physical habitat relief was also similar among islands. Again, Inaccessible differed slightly from the other islands in having lower relief (74% Slight, 22% Mod and 4% High vs. 40–56% Slight, 22–31% Mod and 17–33% High for the other islands). Only Tristan had any flat relief, corresponding to the sand measured at the single sandy site surveyed.

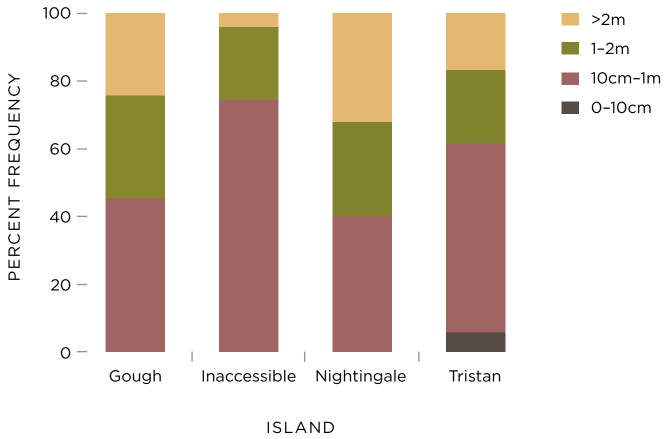
FIGURE 4.

Substrate composition and physical relief estimated using visual SCUBA surveys in nearshore kelp forests (10 and 20m depth) in the Tristan da Cunha Islands. Shown are the percent cover of (A) four different substrate categories and (B) vertical relief in four different relief categories.

A. Substrate



B. Physical Relief



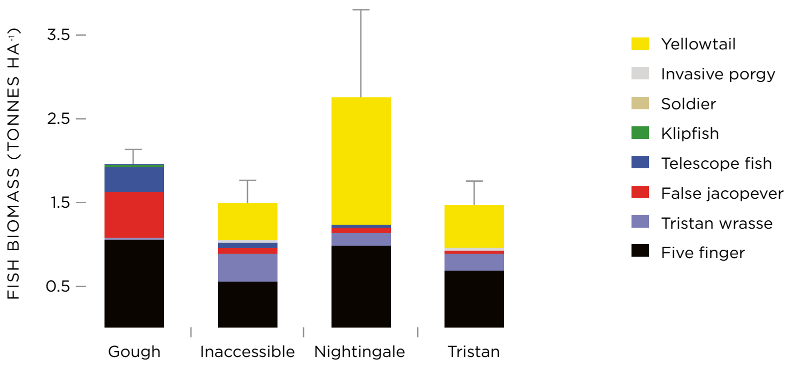
FISH DENSITY AND BIOMASS

TOTAL

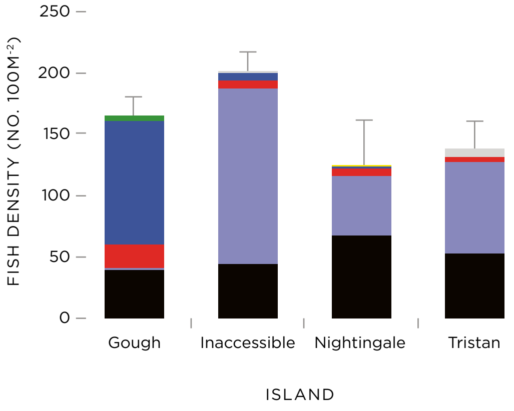
Mean fish biomass (tonnes/ha) and density (no./100m²) for all observed fish species at each island are listed in Table 1A, B and shown in Figure 5A, B. Average total fish biomass ranged between 1.5–2.75 tonnes per hectare. Five finger (*Nemadactylus monodactylus*) were dominant both numerically and in terms of biomass at all four islands. Interisland differences in biomass were driven by yellowtail (*S. lalandi*) which were abundant at the northern islands but absent at Gough. At Gough, where most fish were larger on average, false jacopever (*Sebastes capensis*) and telescope fish (*Mendosoma lineatum*) contributed to the differences in the biomass estimates between island groups. The most numerically abundant fish species at the northern islands was the endemic Tristan wrasse (*Suezichthys ornatus*), with higher densities than the five finger at most sites. At Gough, telescope fish were the numerically dominant species, often found in large schools of juvenile sized fishes.

FIGURE 5.

A. Fish Biomass



B. Fish Density



Stacked bar plots depicting fish biomass and density patterns in nearshore habitats of the Tristan da Cunha Islands from SCUBA surveys. Shown are the island-level mean (A) fish biomass (tonnes ha^{-1}) and (B) fish density (no. m^{-2}) for each species observed. Error bars are ± 1 standard error of the mean for the total fish biomass or total fish density respectively.

TABLE 1.

Biomass (tonnes ha^{-1}) and density (no. 100m^{2}) of fish, conspicuous benthic invertebrates, and kelps observed on SCUBA surveys at 10 and 20m depth in the Tristan da Cunha Islands. Values are means ± 1 standard error of the mean for each island.

Species				
A. Fish Biomass	**Gough**	**Inaccessible**	**Nightingale**	**Tristan**
Total fish biomass	1.95 ± 0.18	1.50 ± 0.27	2.75 ± 1.05	1.46 ± 0.29
Five finger (*Nemadactylus monodactylus*)	1.1 ± 0.10	0.56 ± 0.18	0.99 ± 0.32	0.69 ± 0.12
Tristan Wrasse (*Suezichthys ornatus*)	0.02 ± 0.008	0.33 ± 0.05	0.15 ± 0.07	0.21 ± 0.04
False Jacopever (*Sebastes capensis*)	0.55 ± 0.05	0.063 ± 0.02	0.071 ± 0.02	0.024 ± 0.005
Telescope fish (*Mendosoma lineatum*)	0.30 ± 0.07	0.06 ± 0.03	0.018 ± 0.01	0.005 ± 0.002
Klipfish (*Bovichtus diacanthus*)	0.018 ± 0.003	0.0001 ± 1e-4	0.0003 ± 2e-4	0 ± 0
Soldier (*Helicolenus mouchezi*)	0.002 ± 0.002	0.01 ± 0.01	0 ± 0	0 ± 0
Invasive porgy (*Diplodus argenteus*)	0 ± 0	0.009 ± 0.006	0 ± 0	0.036 ± 0.01
Yellowtail (*Seriola lalandi*)	0 ± 0	0.46 ± 0.24	1.52 ± 0.74	0.51 ± 0.28

B. Fish Density	**Gough**	**Inaccessible**	**Nightingale**	**Tristan**
Total fish density	164.5 ± 16.6	201. 1 ± 15.9	124. 4 ± 36.4	138.0 ± 21.0
Five finger (*Nemadactylus monodactylus*)	39.1 ± 4.0	42.0 ± 12.4	65.6 ± 18.2	52.0 ± 7.6
Tristan Wrasse (*Suezichthys ornatus*)	1.5 ± 0.4	146.1 ± 25.4	49.4 ± 19.7	75.4 ± 16.3
False Jacopever (*Sebastes capensis*)	18.9 ± 1.9	6.3 ± 1.3	7.0 ± 1.3	2.6 ± 0.5
Telescope fish (*Mendosoma lineatum*)	101.1 ± 16.0	4.7 ± 2.9	1.1 ± 0.7	0.4 ± 0.2
Klipfish (*Bovichtus diacanthus*)	3.9 ± 0.5	0.1 ± 0.1	0.1 ± 0.08	0 ± 0
Soldier (*Helicolenus mouchezi*)	0.04 ± 0.04	0.12 ± 0.12	0 ± 0	0 ± 0
Invasive porgy (*Diplodus argenteus*)	0 ± 0	1.3 ± 1.0	0 ± 0	7.1 ± 3.5
Yellowtail (*Seriola lalandi*)	0 ± 0	0.5 ± 0.2	1.2 ± 0.6	0.5 ± 0.2

C. Benthic species density and biomass	**Gough**	**Inaccessible**	**Nightingale**	**Tristan**
Giant kelp (*Macrocystis pyrifera*)	53.9 ± 10.1	53.8 ± 12.6	44.7 ± 15.6	49.1 ± 14.4
Giant kelp stipes	265.6 ± 29.8	279.5 ± 67.2	354.3 ± 117.5	262.9 ± 64.5
Pale kelp (*Laminaria pallida*)	235.9 ± 24.1	388.8 ± 56.1	511.8 ± 147.0	533.6 ± 109.4
Lobster biomass (*Jasus tristani*)	0.15 ± 0.022	0.11 ± 0.019	0.13 ± 0.014	0.085 ± 0.012
Lobster density	6.4 ± 0.8	8.6 ± 1.2	7.2 ± 1.3	8.8 ± 1.3
Sea urchin (*Arbacia dufresnii*)	389.4 ± 62.7	119.5 ± 25.4	60.1 ± 25.2	23.2 ± 7.2
Pink urchin (*Pseudechinus magellanicus*)	8.0 ± 4.1	0 ± 0	0 ± 0	0 ± 0
Sea star (*Henricia simplex*)	8.2 ± 1.3	6.1 ± 0.9	7.7 ± 4.5	12.6 ± 1.9
Sea star (*Odontaster penicillatus*)	0.54 ± 0.2	0 ± 0	0 ± 0	0 ± 0
Common octopus (*Octopus vulgaris*)	0.25 ± 0.09	0.47 ± 0.2	0.41 ± 0.2	0.13 ± 0.06
Barnacle (*Austromegabalanus* sp.)*	0.41 ± 0.42	0.12 ± 0.11	12.5 ± 12.4	0 ± 0

**Barnacles were recorded at high densities at the site of the Oliva cargo shipwreck, accounting for the high density average at Nightingale island. At natural reefs the density was low.*

ANOVA revealed significant variation among islands and sites nested within islands for total fish biomass (Table 2; Figure 6A). There was no effect of depth zone on total biomass (Table 2). Inaccessible showed the least variation in total fish biomass among sites (ranging from 0.7 to 2.3mt/ha). Nightingale had the greatest site to site variation in total fish biomass (ranging from 1.0 to 8.8mt/ha) due to very high biomass (primarily of Yellowtail) at a site located on Stoltenhoff Island, slightly offshore of the main island (Figure 6A).

Total fish density showed less variation overall compared to total fish biomass, but the effects of island, site (within island) and depth zone (nested within island and site) were all significant (Table 3; Figure 6B). Inaccessible showed the least variation among sites (ranging from 138.8 to 261.3 fish/100m^2) while Nightingale showed the most variation in density among sites (54.2 to 333.3) again, largely driven by a single site on Stoltenhoff Island (Figure 6B). Total fish density was greater on the deep zone transects (mean 190.1 ± 14.3 SE) than the shallow zone transects (121.1 ± 11.3 SE).

TABLE 2.

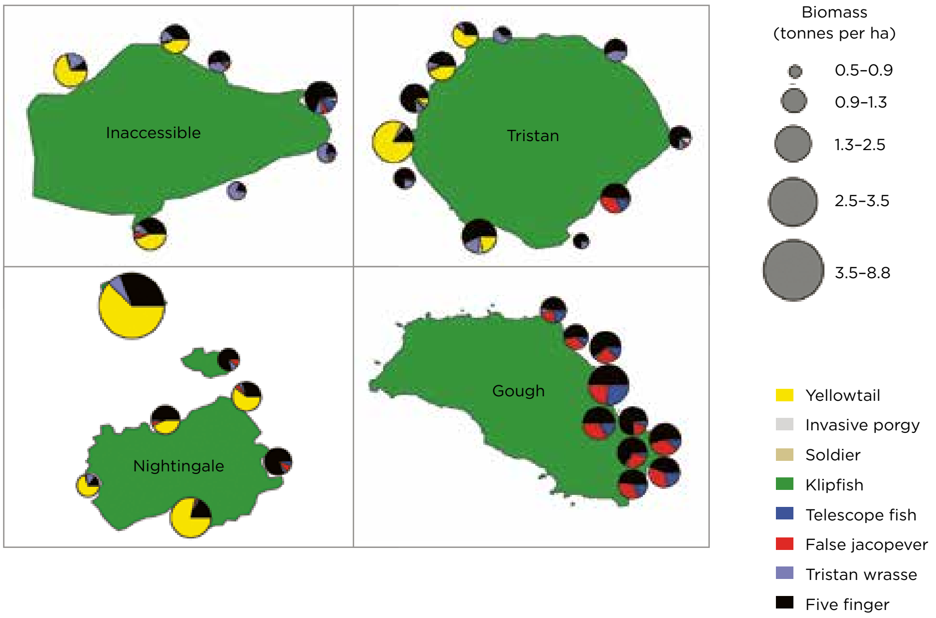
Nested analysis of variance (ANOVA) testing the effects of island, site, and depth zone on patterns of fish biomass among the Tristan da Cunha islands. Statistically significant p-values are in bold text.

	Model r^2	Factors	df	F-ratio	P-value
A. Total fish biomass	0.61	Island	3, 68	2.91	**0.041**
		Site [Island]	30, 68	2.18	**0.0041**
		Zone [Site, Island]	34, 68	0.97	0.53
B. Five finger biomass	0.84	Island	3, 68	12.35	**<0.0001**
		Site [Island]	30, 68	6.76	**<0.0001**
		Zone [Site, Island]	34, 68	3.98	**<0.0001**
C. Tristan wrasse biomass	0.84	Island	3, 68	58.35	**<0.0001**
		Site [Island]	30, 68	7.10	**<0.0001**
		Zone [Site, Island]	34, 68	3.43	**<0.0001**
D. False jacopever biomass	0.89	Island	3, 68	140.00	**<0.0001**
		Site [Island]	30, 68	2.25	**0.0029**
		Zone [Site, Island]	34, 68	1.08	0.38
E. Telescope fish biomass	0.61	Island	3, 68	10.41	**<0.0001**
		Site [Island]	30, 68	1.05	0.41
		Zone [Site, Island]	34, 68	1.34	0.15
F. Klipfish biomass	0.80	Island	3, 68	54.53	**<0.0001**
		Site [Island]	30, 68	2.30	**0.0023**
		Zone [Site, Island]	34, 68	1.30	0.17
G. Invasive porgy biomass	0.86	Island	3, 68	23.86	**<0.0001**
		Site [Island]	30, 68	5.52	**<0.0001**
		Zone [Site, Island]	34, 68	5.24	**<0.0001**
H. Yellowtail biomass	0.55	Island	3, 68	4.46	**0.0064**
		Site [Island]	30, 68	1.49	0.088
		Zone [Site, Island]	34, 68	0.73	0.83

FIGURE 6.

Bubble plots depicting site-level variation in fish biomass, density, and species composition of kelp forest fish communities in the four Tristan da Cunha Islands. Shown are plots of (A) fish biomass and (B) fish density. Bubble size scales with the biomass or density estimated using site-level means of SCUBA surveys.

A. Fish Biomass



B. Fish Density

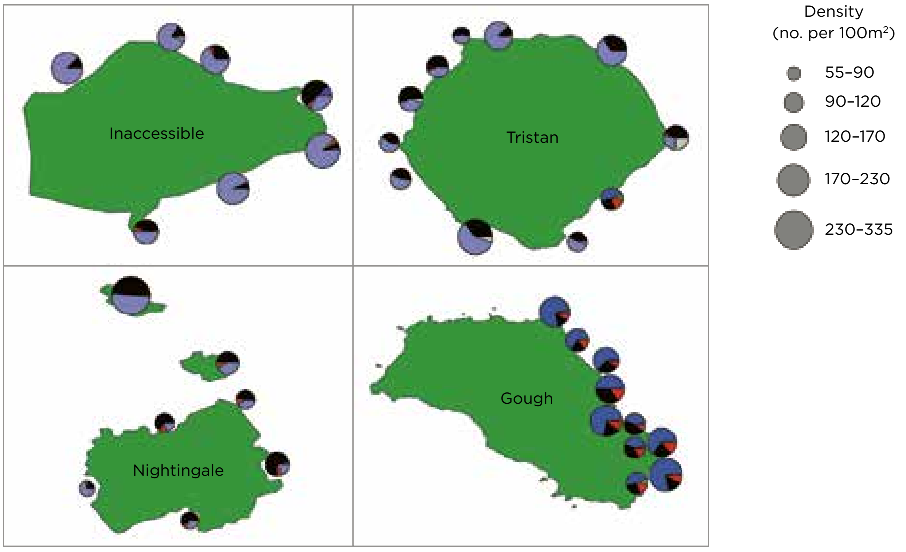


TABLE 3.

Nested analysis of variance (ANOVA) testing the effects of island, site, and depth zone on patterns of fish density among the Tristan da Cunha islands. Statistically significant p-values are in bold text.

	Model r^2	Factors	df	F-ratio	P-value
A. Total fish density	0.79	Island	3, 68	6.48	**0.0006**
		Site [Island]	30, 68	3.36	**<0.0001**
		Zone [Site, Island]	34, 68	3.99	**<0.0001**
B. Five finger density	0.83	Island	3, 68	8.42	**<0.0001**
		Site [Island]	30, 68	6.81	**<0.0001**
		Zone [Site, Island]	34, 68	3.71	**<0.0001**
C. Tristan wrasse density	0.87	Island	3, 68	68.09	**<0.0001**
		Site [Island]	30, 68	5.16	**<0.0001**
		Zone [Site, Island]	34, 68	2.34	**0.0014**
D. False jacopever density	0.88	Island	3, 68	109.52	**<0.0001**
		Site [Island]	30, 68	3.56	**<0.0001**
		Zone [Site, Island]	34, 68	2.43	**0.0009**
E. Telescope fish density	0.84	Island	3, 68	48.10	**<0.0001**
		Site [Island]	30, 68	1.62	0.051
		Zone [Site, Island]	34, 68	5.92	**<0.0001**
F. Klipfish density	0.81	Island	3, 68	54.53	**<0.0001**
		Site [Island]	30, 68	1.62	0.051
		Zone [Site, Island]	34, 68	1.99	**0.008**
G. Invasive porgy biomass	0.90	Island	3, 68	27.31	**<0.0001**
		Site [Island]	30, 68	9.42	**<0.0001**
		Zone [Site, Island]	34, 68	8.01	**<0.0001**
H. Yellowtail density	0.53	Island	3, 68	3.59	**0.018**
		Site [Island]	30, 68	1.42	0.12
		Zone [Site, Island]	34, 68	0.74	0.84

INDIVIDUAL SPECIES

There were significant island to island differences in the biomass and the density of every individual species tested (Table 2, 3). Most of this variation was driven by differences between the northern islands and Gough as described above. All species except for telescope fish, yellowtail and the rarely seen klipfish differed significantly in biomass and density (density only for klipfish) among sites within islands. Yellowtail are known to be highly mobile (at the scale of these islands) and telescope fish mobility is unknown, but may play a role in the homogeneity across sites. Depth zone had a lesser effect on biomass than on density. Depth zone explained significant variance in biomass of five finger, Tristan wrasse (higher biomass on deep transects) and the invasive silver porgy (higher biomass on shallow transects) and explained significant levels of variance in density for all species except yellowtail (Table 3). In general, telescope fish, Tristan wrasse, five finger, false jacopever, and yellowtail were all present in greater density in the deep zone while klipfish and silver porgy were more abundant in the shallow zone.

INVERTEBRATE AND MACROALGAE BIOMASS AND DENSITY

LOBSTERS

Lobster (*Jasus tristani*) biomass and density both varied significantly among islands, sites within island, and biomass also varied by depth zone (Table 1, 4; Figure 8A). Interestingly, the density and biomass patterns were inversely related across islands (Figure 7A, B). For example, lobsters were most numerous at Tristan and Inaccessible Islands but biomass was largest at Gough and Nightingale. This was due to size structure differences (see below section), with larger lobsters at Gough and Nightingale and smaller lobsters at Tristan and Inaccessible. The variation in biomass among sites within islands was smallest at Nightingale (mean site biomass ranged from 0.10 to 0.21mt/ha) and was largest at Gough (mean site biomass 0.07 to 0.31mt/ha). There were no strong patterns with depth (despite a marginally significant effect of zone on biomass, Table 4).

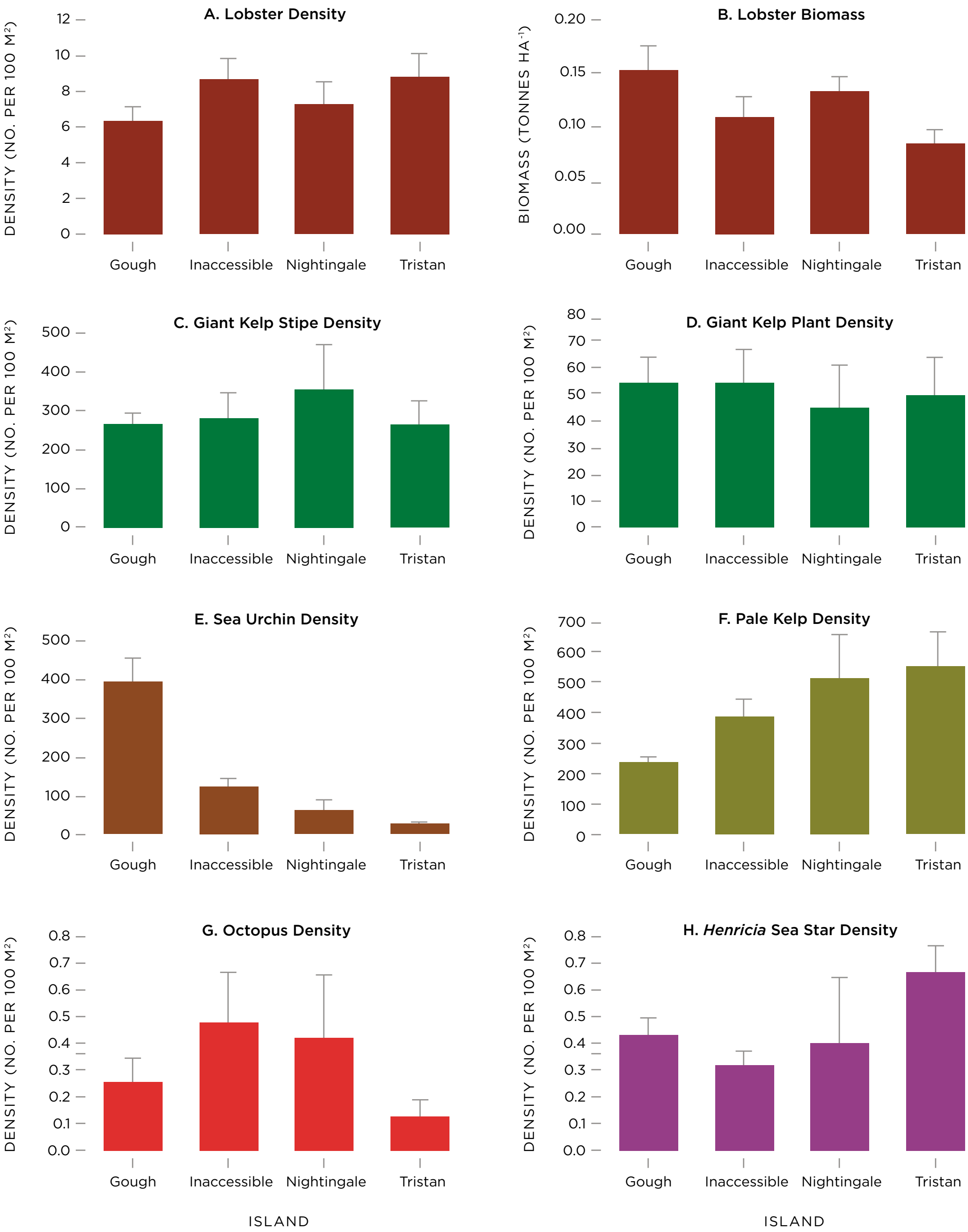
TABLE 4.

Nested analysis of variance (ANOVA) testing the effects of island, site, and depth zone on patterns of invertebrate and macroalgal density and lobster biomass among the Tristan da Cunha islands. Statistically significant p-values are in bold text.

	Model r^2	Factors	df	F-ratio	P-value
A. Lobster biomass	0.67	Island	3, 68	5.93	**0.0012**
		Site [Island]	30, 68	2.01	**0.0094**
		Zone [Site, Island]	34, 68	1.65	**0.043**
B. Lobster density	0.70	Island	3, 68	3.23	**0.028**
		Site [Island]	30, 68	3.19	**<0.0001**
		Zone [Site, Island]	34, 68	1.56	0.06
C. Sea urchin density	0.75	Island	3, 68	39.75	**<0.0001**
		Site [Island]	30, 68	2.06	**0.0071**
		Zone [Site, Island]	34, 68	0.54	0.97
D. *Henricia* sea star density	0.83	Island	3, 68	8.52	**<0.0001**
		Site [Island]	30, 68	5.65	**<0.0001**
		Zone [Site, Island]	34, 68	4.24	**<0.0001**
E. Octopus density	0.72	Island	3, 68	2.99	**0.037**
		Site [Island]	30, 68	2.38	**0.0017**
		Zone [Site, Island]	34, 68	3.00	**<0.0001**
F. Giant kelp plant density	0.78	Island	3, 68	54.53	0.76
		Site [Island]	30, 68	1.62	**<0.0001**
		Zone [Site, Island]	34, 68	1.99	**<0.0001**
G. Giant kelp stipe density	0.90	Island	3, 68	1.58	0.20
		Site [Island]	30, 68	4.57	**<0.0001**
		Zone [Site, Island]	34, 68	3.19	**<0.0001**
H. Pale kelp density	0.70	Island	3, 68	7.21	**0.0003**
		Site [Island]	30, 68	2.68	**0.0004**
		Zone [Site, Island]	34, 68	1.62	**0.047**

FIGURE 7.

Bar plots depicting island-level variation in lobster biomass and the density of common invertebrates and macroalgae in the Tristan da Cunha Islands from SCUBA surveys of nearshore habitats. Shown are mean values for each species ± 1 standard error of the mean. Urchins are all *Arabacia dufresnii*.



OTHER BENTHIC INVERTEBRATES

Densities of all surveyed invertebrates (lobsters, urchins, seastars, octopus, and barnacles) are given in Table 1C. Many of these species were not abundant so we performed ANOVA on a subset of them (Sea urchin [*Henricia* spp.], seastars and Octopus; Table 4). For *Henricia* spp. and octopus densities, there was significant variation among islands, sites within island and depth zones (Table 4C, D, E). For these two species, we did not observe consistent differences between the northern island group and Gough. Sea urchins also varied significantly among islands and sites but there was no effect of depth (Table 4; Figure 7E). Urchin density was highest at Gough and lower at Inaccessible, Nightingale and Tristan, respectively (Figure 7E, 9).

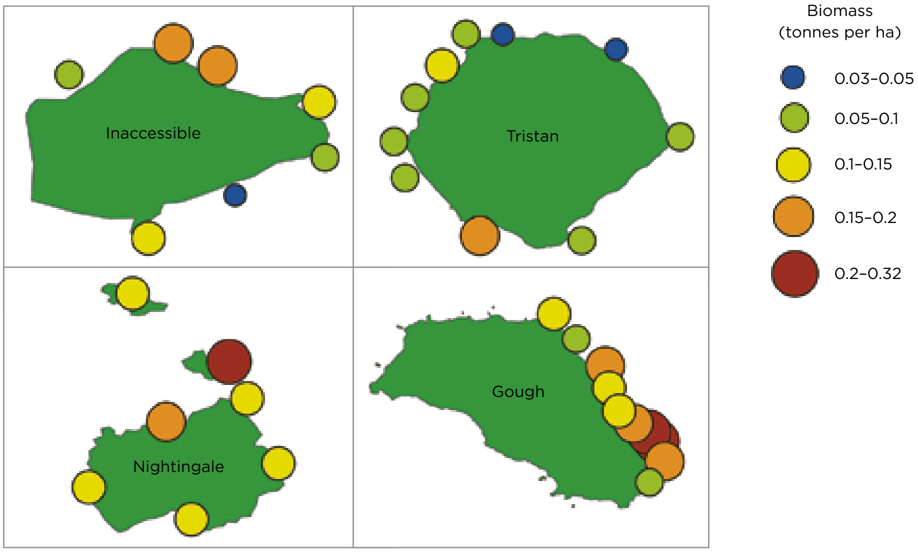
KELPS

Interestingly, giant kelp (*M. pyrifera*) plant and stipe density was the least variable among islands with no consistent differences (Table 1, 4; Figure 7C, D), even between the northern islands and Gough. Mean island-level densities of kelp plants ranged from 44.7 to 53.9 per 100m^2 and stipe counts ranged from 262.9 to 353.4 stipes per 100m^2 (Table 1C). The relationship between plant counts and stipe counts was inverse, with the highest stipes/plant and lowest plant density at Nightingale and lower stipe counts but higher plant densities at the other three islands. There was, however, significant site to site variation and depth variation for both kelp plant density (Table 4F) and stipe density (Table 4G; Figure 8B). Despite some site to site variation in giant kelp plant and stipe densities, within each island, the range in density across sites for kelp was less than

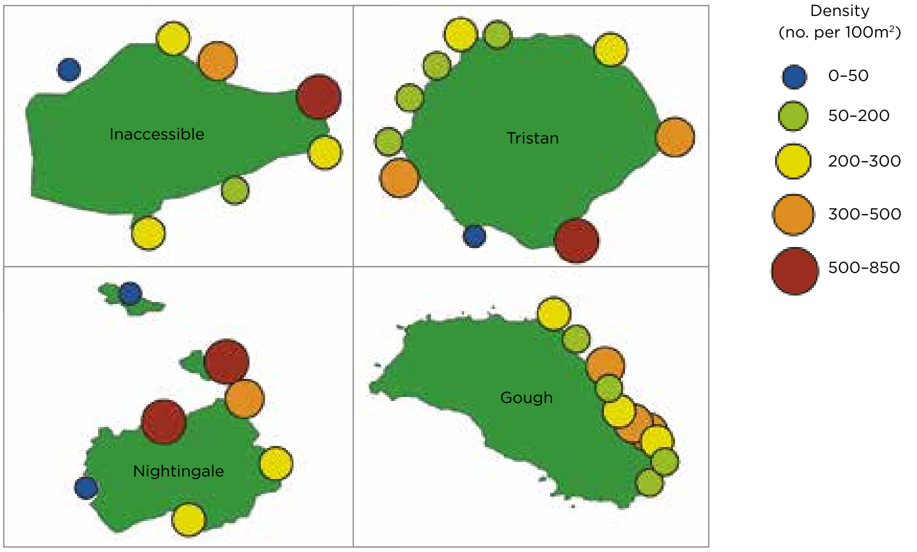
FIGURE 8.

Bubble plots depicting site-level variation in (A) lobster (*Jasus tristani*) biomass and (B) giant kelp (*Macrocystis pyrifera*) stipe density across the four Tristan da Cunha Islands. Bubble size scales with the biomass or density estimated using site-level means of SCUBA surveys in nearshore habitats.

A. Lobster Biomass



B. Giant Kelp Stipe Density



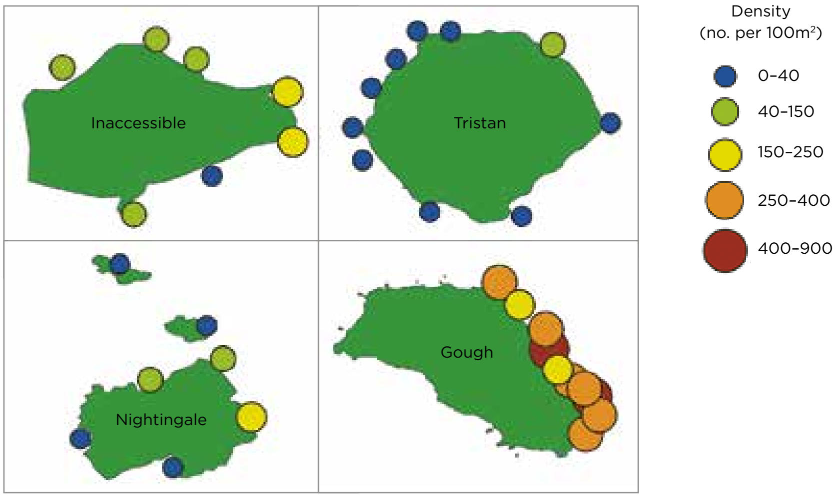
that for fishes and invertebrates (Figure 8B, compare to Figure 6A, B – fishes and Figure 8A – lobsters). The significant effect of depth on giant kelp was due to much greater density of both plants and stipes at the shallow zone at Gough Island relative to the deep zone. At the northern islands, this depth stratification was not apparent.

Pale kelp (*Laminaria pallida*) was the other major component of the kelp forests at Tristan Islands. Unlike giant kelp, pale kelp densities varied significantly across islands, with this warm water tolerant species showing higher densities at the northern islands, relative to colder Gough Island (Table 1, 4; Figure 7F). The range of variation among sites within islands in pale kelp density was similar at the three northern islands (Figure 9B). With lower densities overall, the range in variation in giant kelp density among sites was less at Gough (Figure 8B). Depth structure for pale kelp was inconsistent: at Gough, Inaccessible and Nightingale, pale kelp was more abundant in the deep zone, while at Tristan, the opposite was true.

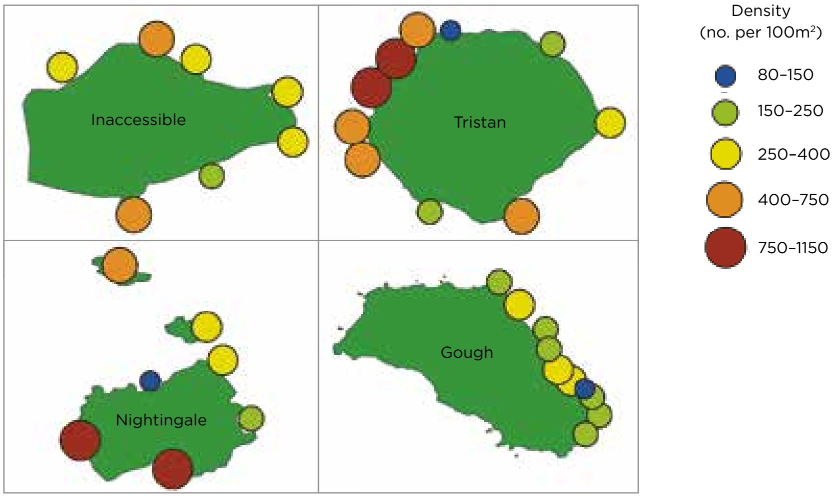
FIGURE 9.

Bubble plots depicting site-level variation in (A) sea urchin (*Arbacia dufresnii*) and (B) pale kelp (*Laminaria pallida*) density across the four Tristan da Cunha Islands. Bubble size scales with the biomass or density estimated using site-level means of SCUBA surveys in nearshore habitats.

A. Sea Urchin Density



B. Pale Kelp Density

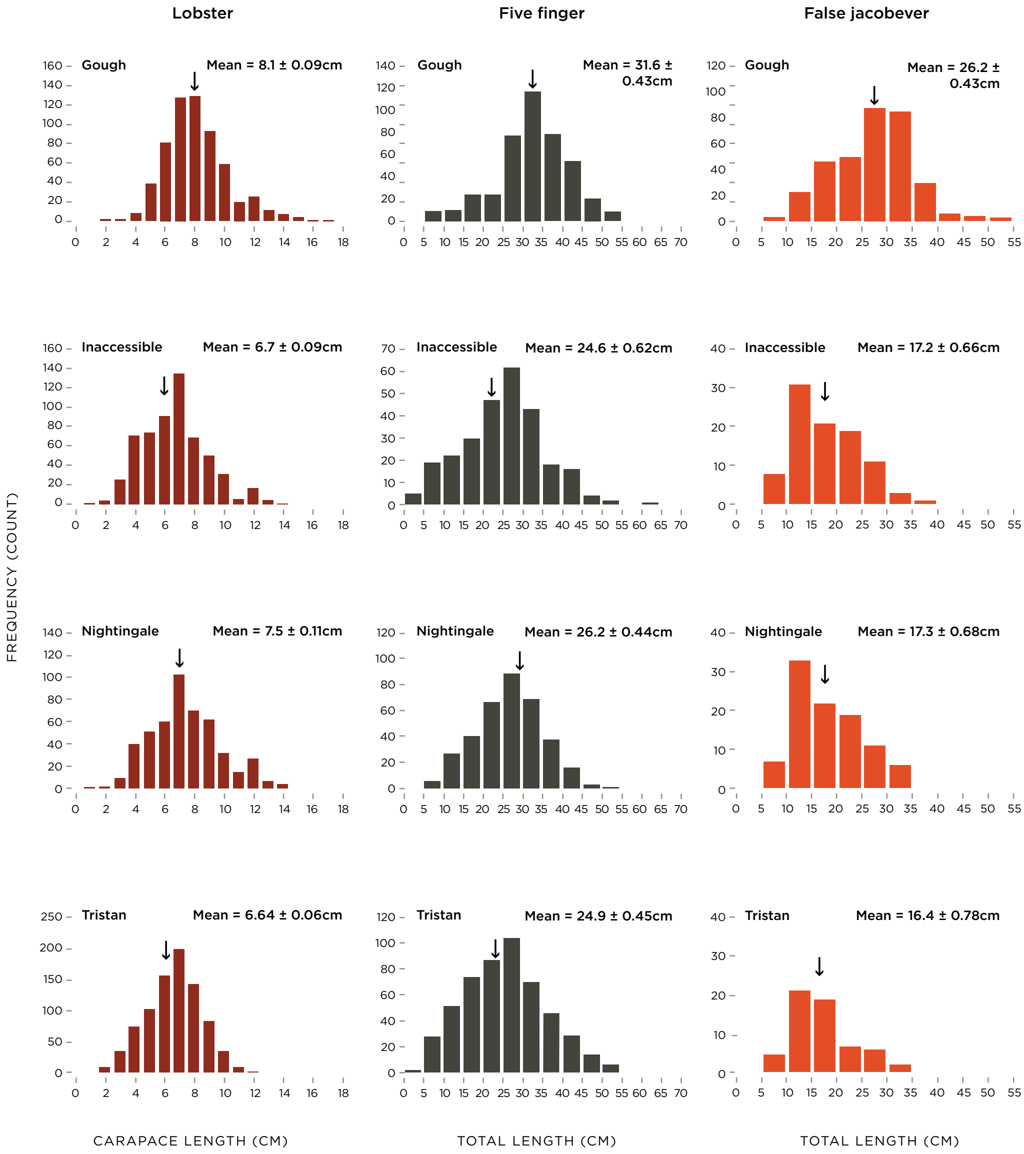


SIZE STRUCTURE

We observed notable differences in the sizes of several species in our surveys across the islands, with the biggest differences being between the northern islands and Gough (Figure 10). These size differences drive the biomass patterns discussed above but are worth highlighting here. Notably, lobsters, false jacopever, and five finger were all much larger at Gough than the other islands. Lobster carapace length averaged 8.1cm at Gough compared to 6.7, 7.5 and 6.6cm at Inaccessible, Nightingale and Tristan, respectively. The two fish species showed very similar patterns across islands.

FIGURE 10.

Spatial variation in size structure of lobsters, and two common nearshore fish species estimated using visual SCUBA surveys. Shown are size frequency histograms for each species pooled at the island-level. Arrows above the histogram depict the estimated mean size at each island. Values on plots are the mean size ± 1 standard error of the mean.



COMMUNITY STRUCTURE AND SPECIES INTERACTIONS

We used PERMANOVA to test for (Table 5, 6) and NMDS to visualize (Figure 11A, B) differences in the structure of fish and benthic communities (invertebrates/kelps) separately. For fish communities, we found significant differences in the community structure between islands, with each pair of sites significantly different (Table 5). Percent similarity was highest between the northern island pairs (ranging from 68.3 to 72.8%) while the similarity of Gough to each of the northern islands was much less (ranging from 41.1 to 48.5%). Within islands, site to site variation in fish community structure was quite high and ranged from 75.4 to 86.1%.

TABLE 5.

Results of a PERMANOVA testing differences in fish community structure among the Tristan da Cunha Islands. Comparisons between the same locations indicate similarity among sites within that island.

A. PERMANOVA Results

Source	df	SS	MS	Pseudo-F	P-value
Island	3	21869.0	7289.5	28.61	0.001
Error	30	7642.5	254.75		
Total	33	29511.0			

B. Post-hoc pairwise comparisons

Groups	t	P-value
Gough vs. Inaccessible	7.73	0.001
Gough vs. Nightingale	6.72	0.002
Gough vs. Tristan	9.11	0.001
Inaccessible vs. Nightingale	2.19	0.018
Inaccessible vs. Tristan	2.05	0.012
Nightingale vs. Tristan	1.78	0.021

C. Average percent similarity between/within Islands

	Gough	Inaccessible	Nightingale	Tristan
Gough	86.1	–	–	–
Inaccessible	43.8	75.8	–	–
Nightingale	48.5	68.3	75.4	–
Tristan	41.1	72.5	72.8	77.8

TABLE 6

Results of a PERMANOVA testing differences in benthic invertebrate and kelp community structure among the Tristan da Cunha Islands.

A. PERMANOVA Results

Source	df	SS	MS	Pseudo-F	P-value
Island	3	4165.3	1388.4	4.57	0.001
Error	30	9122.7	304.1		
Total	33	13288.0			

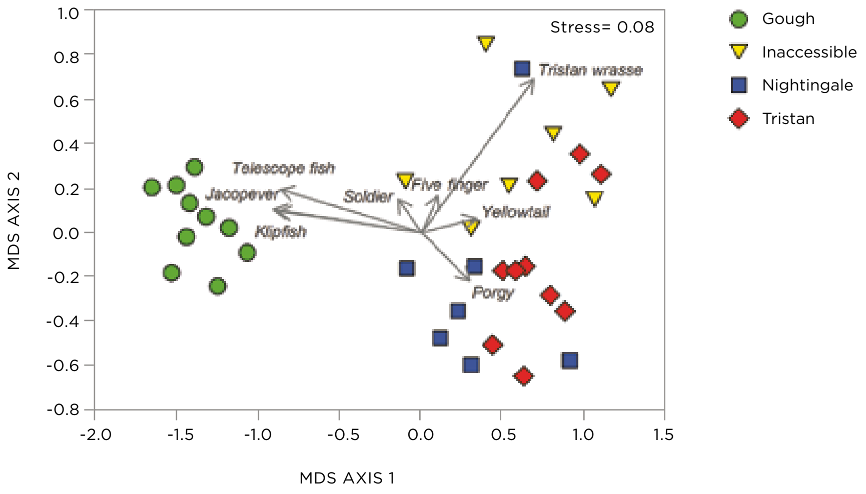
B. Post-hoc pairwise comparisons

Groups	t	P-value
Gough vs. Inaccessible	2.86	0.001
Gough vs. Nightingale	2.64	0.001
Gough vs. Tristan	3.84	0.001
Inaccessible vs. Nightingale	0.68	0.72
Inaccessible vs. Tristan	1.61	0.046
Nightingale vs. Tristan	0.56	0.85

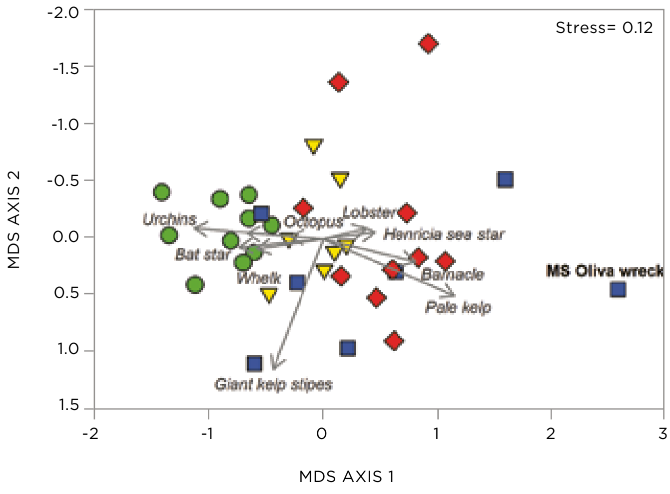
C. Average percent similarity between/within Islands

	Gough	Inaccessible	Nightingale	Tristan
Gough	85.8	–	–	–
Inaccessible	78.9	83.3	–	–
Nightingale	67.3	74.6	65.8	
Tristan	68.5	76.9	71.6	76.3

For benthic communities (invertebrates and algae) there were also island effects but these were largely driven by differences between Gough and all the northern islands. The three northern islands were not different from one another, as was found for fishes. Percent similarity was high between the northern island pairs (ranging from 71.6 to 76.9%), as observed with the similarity patterns for fishes, while the similarity of Gough to each of the northern islands was less but still higher than similarity patterns for fishes (ranging from 67.3 to 78.9%) (Table 6). Within islands, site to site variation in benthic community structure ranged from 65.8 to 85.8%.



A. Fish Community Structure



B. Benthic Community Structure

FIGURE 11.

Multivariate description of fish and benthic communities in the Tristan da Cunha Islands from nearshore SCUBA surveys. Plots depict non-metric multidimensional scaling (nMDS) analyses of (A) fish assemblages and (B) benthic assemblages using site-level densities of species observed. Vectors overlaying the plot depict the species that are driving separation among sites and islands in species composition in fish and benthic communities. Data were square root transformed prior to analysis.

Kelp forest communities at the Tristan Islands are not species rich and the trophic structure is simple compared to more species rich kelp forests in other parts of the world. Surprisingly, we found no relationship between giant kelp (primary producer) and urchins (grazer) but a strong inverse relationship between pale kelp and urchins. During survey and non-survey SCUBA dives, we never observed urchins grazing or resting on giant kelp but we did observe urchins both on the holdfasts and blades of the pale kelp (Figure 12). Lobster density and biomass also showed no clear relationship with urchins (their putative prey) or octopus (their putative predators) but did show similarly low levels of variation among islands as giant kelp.

FIGURE 12.

Urchins (*Arbacia dufresnii*) on the holdfast and blades of pale kelp (*Laminaria pallida*).

OLIVA WRECK SITE

We surveyed the wreck site of the MV Oliva at Nightingale. Surveys took place on natural habitat just inshore of the wreck and deeper habitat directly over much of the structure of the wreck. In general, we saw no dramatic differences relative to other sites at Nightingale (apart from the structure of the wreck itself). Most of the remaining structure is covered completely with benthic turfing, foliose algae and pale kelp (*L. pallida*) (Figure 13). There was very little giant kelp (*M. pyrifera*) at that site. In community structure analysis of the benthic community, this particular location was separated in space from all of the other sites, and was more closely related to Tristan communities. This result was due to high abundance of barnacles (common on the structure of the wreck) and pale kelp and very low abundance of giant kelp and urchins. For fish communities, no strong differences were observed, likely due to the mobility of this taxonomic group.

FIGURE 13.

Benthic community at the site shipwreck of the Oliva.

Deep Sea and Pelagic Camera Surveys

Pristine Seas utilizes remote cameras extensively in order to characterize fish and marine mammal communities without the presence of divers and in habitats inaccessible to divers (e.g. to deep, too far offshore). Open ocean and very deep habitats are all within camera survey reach but are rarely surveyed despite their importance to humans (e.g. fisheries) and ocean biodiversity. Specially built National Geographic deep-sea drop cameras and stereo video mid-water camera rigs were used to enumerate open ocean wildlife. Detailed methods and description of camera systems are provided at the end of this report (Detailed Methods).

DEEP-SEA DROP CAMERA RESULTS

Deep-sea drop cameras were deployed at sites on each side of every island (Figure 3), when conditions allowed and in a variety of depths on each island (Table 7). In all, we completed 23 deep-water drops to depths ranging from 164m to 1414m. Cameras are timed to record for two hours but several drops suffered premature releases, sometimes due to sharks severing the lines (Drops T9, T17 and T21; Table 7).

TABLE 7.

Details of the deep-sea drop camera deployments for all stations surveyed in the Tristan da Cunha Islands group.

Drop No.	Date	Island	Latitude	Longitude	Depth (m)	Time of day	Duration (min)
T1	16-Jan-17	Tristan	-37.03262	-12.31115	893	14:55	120
T2	17-Jan-17	Tristan	-37.02890	-12.26669	1173	07:21	120
T3	17-Jan-17	Tristan	-37.03608	-12.30570	656	10:18	120
T4	18-Jan-17	Tristan	-37.12096	-12.39176	1075	09:55	120
T5	18-Jan-17	Tristan	-37.14068	-12.38775	953	11:53	120
T6	21-Jan-17	Gough	-40.25043	-9.88091	1414	08:24	120
T7	21-Jan-17	Gough	-40.27888	-9.85603	1404	12:55	120
T8	22-Jan-17	Gough	-40.34076	-9.85285	714	08:49	120
T9	22-Jan-17	Gough	-40.36598	-9.85951	467	10:56	60
T10	23-Jan-17	Gough	-40.25772	-9.90926	190	13:15	120
T11	23-Jan-17	Gough	-40.26271	-9.88240	1027	13:29	120
T12	26-Jan-17	Tristan	-37.19624	-12.24193	1027	11:04	120
T13	26-Jan-17	Tristan	-37.17244	-12.22194	1122	14:13	120
T14	28-Jan-17	Inaccessible	-37.27441	-12.73408	225	08:39	120
T15	28-Jan-17	Inaccessible	-37.26014	-12.71080	164	10:42	120
T16	29-Jan-17	Nightingale	-37.39484	-12.55375	1227	07:26	120
T17	29-Jan-17	Nightingale	-37.43821	-12.53270	608	10:14	60
T18	30-Jan-17	Nightingale	-37.44452	-12.44944	418	07:10	120
T19	30-Jan-17	Nightingale	-37.40688	-12.43674	708	10:24	120
T20	31-Jan-17	Tristan	-37.19418	-12.31482	994	08:00	120
T21	31-Jan-17	Tristan	-37.16568	-12.36713	712	10:50	45
T22	01-Feb-17	Tristan	-37.11137	-12.17371	1200	07:21	120
T23	01-Feb-17	Tristan	-37.09369	-12.18232	1203	10:48	120

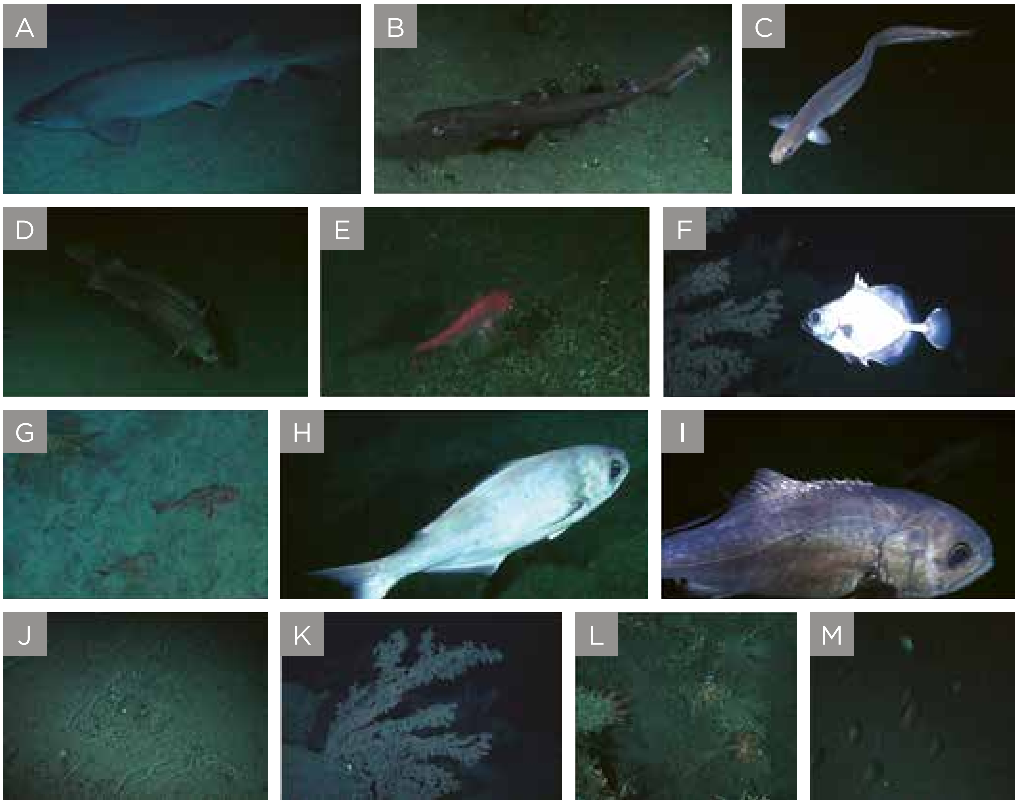
We observed a total of 21 species or species groups and frequency of occurrence (occurrence on a drop/total number drops [n = 23]) ranged from 0.04 to 0.61 (Table 8). Several species or groups were observed on greater than 30% of deep-water drops. These included: bluenose (*Hyperoglyphe antarctica*); lantern shark (*Etmopyerus* sp.); bluntnose sixgill shark (*Hexanchus griseus*); grenadiers (family Macrouridae); deepwater cod (*Physiculus karrerae*); cutthroat eels (*Synaphobranchus* sp.). Shark observations varied with lantern sharks seen at 56% of drops from 714–1,404m, bluntnose sixgill sharks on 30% of drops from 190–1,027m and a single sevengill shark recorded on a drop at 164m. Not all recorded organisms were fish, a southern elephant seal (*Mirounga leonina*) was observed at a single camera drop at 190m depth.

TABLE 8.

Fish taxa observed on deep-sea drop cameras in the Tristan da Cunha Islands group. Freq. = frequency of occurrence out of all drops (n = 23).

Family	Taxa	Common name	Freq.	Depth range (m)
Berycidae	*Beryx decadactylus*	Alfonsino	0.18	656–1203
Carangidae	*Seriola lalandi*	Yellowtail amberjack	0.04	164
Centrolophidae	*Hyperoglyphe antarctica*	Bluenose	0.30	190–708
Centrolophidae	*Schedophilus velaini*	Oval driftfish	0.04	418
Emmelichthyidae	*Emmelichthys nitidus*	Southern rover	0.04	714
Etmopteridae	*Etmopterus* sp. (likely *granulosus*)	Lantern shark	0.56	714–1404
Hexanchidae	*Hexanchus griseus*	Bluntnose sixgill shark	0.30	190–1027
Hexanchidae	*Notorynchus cepedianus*	Sevengill shark	0.04	164
Macrouridae	*Coelorinchus* sp.	Grenadier	0.61	656–1414
Moridae	*Physiculus karrerae*	Deepwater cod	0.39	467–1414
Moridae	*Antimora* sp.	Antimora cod	0.17	994–1414
Ophidiidae	*TB IDed*	Cusk-eel	0.17	1027–1414
Oreosomatidae	*Neocyttus* sp.	Oreo dory	0.22	994–1200
Phocidae	*Mirounga leonina*	Southern elephant seal	0.04	190
Polyprionidae	*Polyprion oxygeneios*	Wreckfish	0.09	164–418
Sebastidae	*Helicolenus mouchezi*	Soldier	0.22	190–608
Sebastidae	*Sebastes capensis*	False jacopever	0.04	190
Serranidae	*Lepidoperca coatsii*	Seabass	0.04	190
Somniosidae	*Somniosus antarcticus*	Southern sleeper shark	0.04	1027
Synaphobranchidae	*Synaphobranchus* sp. (likely *brevidorsalis*)	Cutthroat eel	0.39	656–1414
TB IDed	*TB IDed*	Eel sp. 1	0.17	953–1414

FIGURE 14.



Photos depicting representative fish and habitat-forming benthic species observed on deep-sea camera drops at the Tristan da Cunha Islands. (A) Bluntnose sixgill shark (*Hexanchus griseus*), (B) Lantern shark (*Etmopterus* sp., likely *granulosus*), (C) Cutthroat eel (*Synaphobranchus* sp., likely *brevidorsalis*), (D) Deepwater cod (*Physiculus karrerae*), (E) Roughy (*Beryx decadactylus*), (F) Oreo dory (*Neocyttus* sp.), (G) Soldier (*Helicolenus mouchezi*) and Octopus (*Octopus vulgaris*), (H) Bluenose (*Hyperoglyphe antarctica*), (I) Oval driftfish (*Schedophilus velaini*), (J) Whip coral, (K) Deepwater gorgonian, (L) Field of gorgonians and crinoids, (M) Sea pens.

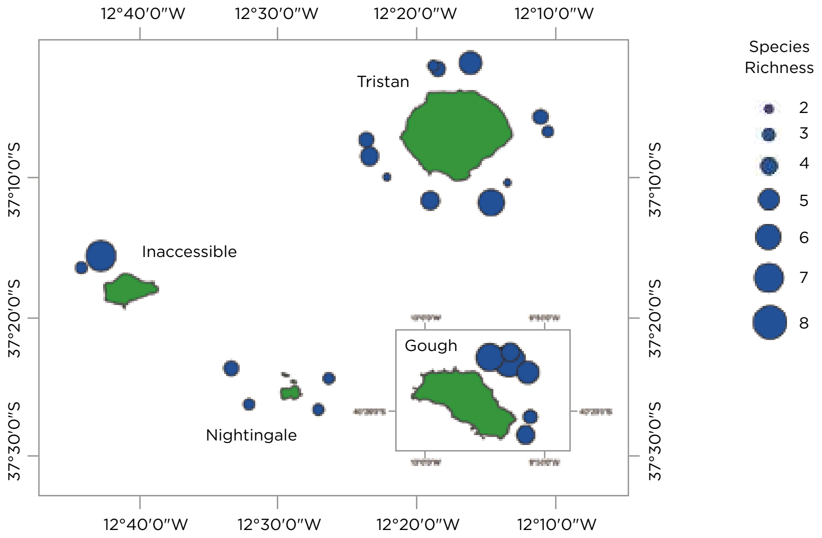
Habitat varied on the deep-water drops. Observed biotic habitat forming organisms included whip corals, gorgonians, sea pens and sea fans (Figure 14). The benthos also varied with some drops on sand/mud habitat and others on what appeared to be moderate to high rock relief. Representative species and habitats observed on deep-water drop cameras are shown in Figure 14.

Species richness ranged from 2 to 8 species per drop and did not show consistent spatial patterns except that the four drops at Nightingale were all relatively low in species richness (Figure 15A). Total N_{max} (sum of all N_{max} across all species per drop) ranged from 2 to 28 individuals per drop and differed spatially among drop camera stations (Figure 15B), with the highest fish abundances on Gough and Tristan compared to the other two islands. Drop camera stations were categorized as having primarily soft or rock substrate. We observed a non-significant trend for higher species richness in drops on rocky habitat (t_{21} = 1.64, P = 0.11) and a significant effect of substrate on total N_{max} (t_{21} = 2.47, P = 0.022), with more fish individuals observed in drops on rocky habitat.

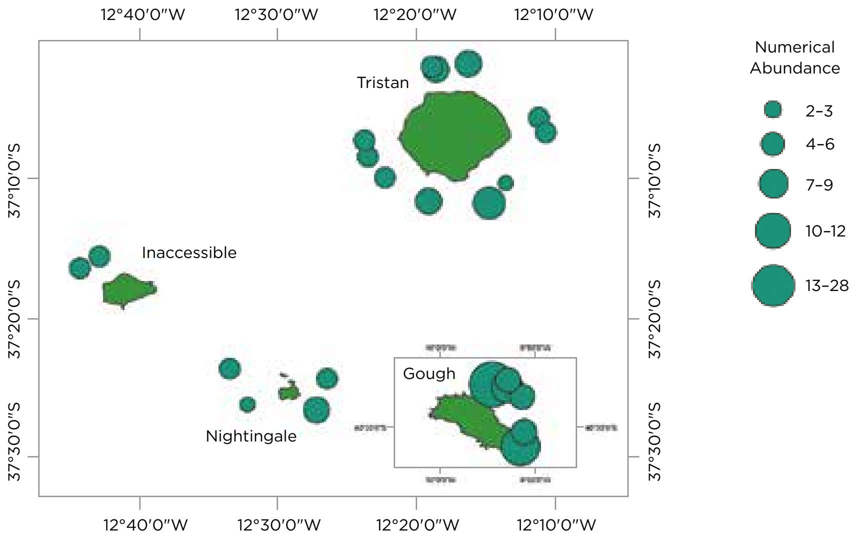
FIGURE 15.

Bubble plots depicting site-level variation in (A) species richness and (B) numerical abundance from deep-sea drop camera stations sampled across the four Tristan da Cunha Islands. Bubble size scales with the richness or total abundance of individual fishes observed from each drop camera deployment.

A. Species Richness



B. Numerical Abundance



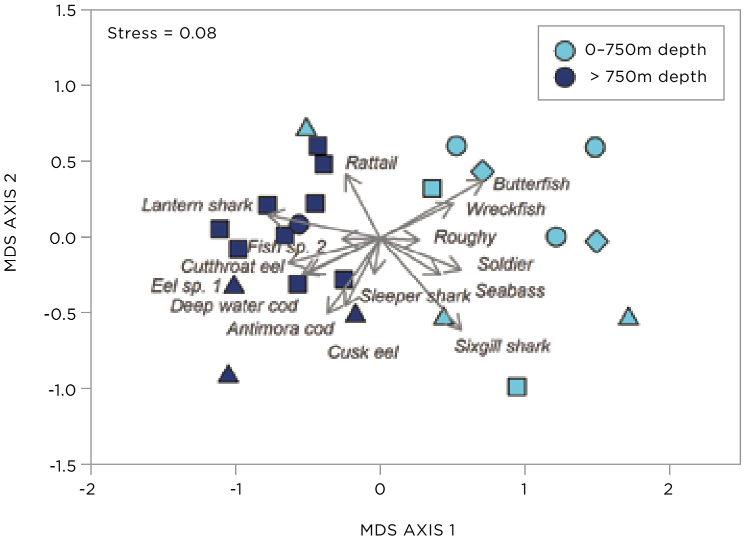
Drop camera stations were also categorized into two different depth zones (0–750m and > 750m depth) for further analysis of diversity and community patterns. We did not observe significant differences in species richness (t_{21} = 0.93, P = 0.36) or total N_{max} (t_{21} = 1.04, P = 0.31) between the drops conducted in the two depth zones. However, we found that the deep-water fish assemblages differed significantly between the shallower and deeper strata, but did not differ among islands (Table 9; Figure 16). Communities shallower than 750m were characterized by the presence of soldiers, bluefish, wreckfish, small seabass and roughy, while communities deeper than 750m were composed of deepwater and antimora cods, cusk eels, rattails, cutthroat eels, and lantern sharks.

TABLE 9.

Results of a PERMANOVA testing differences in fish community structure between depth categories (0–750m vs. > 750m depth) from the deep-sea drop camera surveys at the Tristan da Cunha Islands.

A. PERMANOVA Results – Presence/absence data					
Source	**df**	**SS**	**MS**	**Pseudo-F**	**P-value**
Depth Category	1	15.21	15.2	8.7	0.001
Error	21	36.7	1.7		
Total	22	51.9			

FIGURE 16.



Multivariate description of the fish community in the Tristan da Cunha Islands from deep-sea camera surveys categorized into two depth strata (0–750m and > 750m depth). Shown is a non-metric multidimensional scaling (nMDS) analysis of the fish assemblage using presence/absence data at each station and a Euclidean distance matrix. Vectors overlaying the plot depict the species that are driving separation among sites and islands (Gough = triangles, Inaccessible = diamonds, Nightingale = circles, Tristan = squares).

PELAGIC CAMERA SURVEY RESULTS

Mid-water baited remote underwater video systems (BRUVS) in sampling sets of 3 rigs in "longline" formation were deployed at 27 sites at a distance of approximately 5km from shore. Sampling was distributed across the islands with 12 sites at Tristan da Cunha, five sites at Nightingale Island, four sites at Inaccessible Island and 6 sites at Gough Island (Figure 3). We recorded 402 individual pelagic fishes, marine mammals, birds and turtles, representing 14 species from 11 families.

Across all sample sites, mean total abundance per sample set was 11.8 ± 4.21 (SE) individuals. Mean species richness per sample set was 1.7 ± 0.14 (SE) species. The most abundant species was a horse mackel (*Trachurus* sp. likely *T. longimanus*) with 198 individuals observed across 44.5% of sites (Table 10). The second most abundant species was yellowtail amberjack (*Seriola lalandi*) with 69 individuals observed, although this species was observed primarily in large schools and thus was only present across 7.4% of sites, all of which were in the northern islands. Blue sharks (*Prionace glauca*) by contrast were observed at 55.6% of sites both in the northern group and at Gough Island, with 23 individuals observed. Yellowfin tuna (*Thunnus albacares*) were observed at 11.1% of sites with three individuals noted, all around the northern islands. Observations of all remaining species consisted of one or two individuals occurring at single sites only, this included one sub-Antarctic fur seal (*Arctocephalus tropicalis*) at Gough Island, one loggerhead turtle (*Caretta caretta*), one porbeagle shark (*Lamna nasus*), one pilot fish (*Naucrates ductor*), one crested bellowfish (*Notopogon lilliei*), one oval driftfish (*Schedophilus velaini*), one striped marlin (*Kajikia audax*) and two albacore (*Thunnus alalunga*) at Tristan da Cunha, and two Shepherd's beaked whales (*Tasmacetus shepherdi*) off Inaccessible Island (Figure 17).

TABLE 10.

Species observed in mid-water BRUVS surveys at Tristan da Cunha group. Total number of individuals recorded using highest MaxN per site and MaxN by drop, Mean MaxN per site, associated standard error (SE) and the percentage of sites that species were present.

Common Name	Scientific Names	Total/ Site	Total/ drop	Mn MaxN/ site	SE	% Sites
Subantarctic fur seal	*Arctocephalus tropicalis*	1	1	0.038	0.038	3.7%
Loggerhead Turtle	*Caretta caretta*	1	1	0.038	0.038	3.7%
Porbeagle shark	*Lamna nasus*	1	1	0.038	0.038	3.7%
Pilot fish	*Naucrates ductor*	1	1	0.038	0.038	3.7%
Crested bellowfish	*Notopogon lilliei*	1	1	0.038	0.038	3.7%
Blue shark	*Prionace glauca*	23	40	0.885	0.199	55.6%
Oval driftfish	*Schedophilus velaini*	1	1	0.038	0.038	3.7%
Yellowtail amberjack	*Seriola lalandi*	69	69	2.654	2.415	7.4%
Shepherd's beaked whale	*Tasmacetus shepherdi*	2	2	0.077	0.075	3.7%
Striped marlin	*Kajikia audax*	1	1	0.038	0.038	3.7%
Albacore	*Thunnus alalunga*	2	2	0.077	0.075	3.7%
Yellowfin tuna	*Thunnus albacares*	3	3	0.115	0.063	11.1%
Horse mackerel	*Trachurus* sp.	198	276	7.615	3.687	44.4%

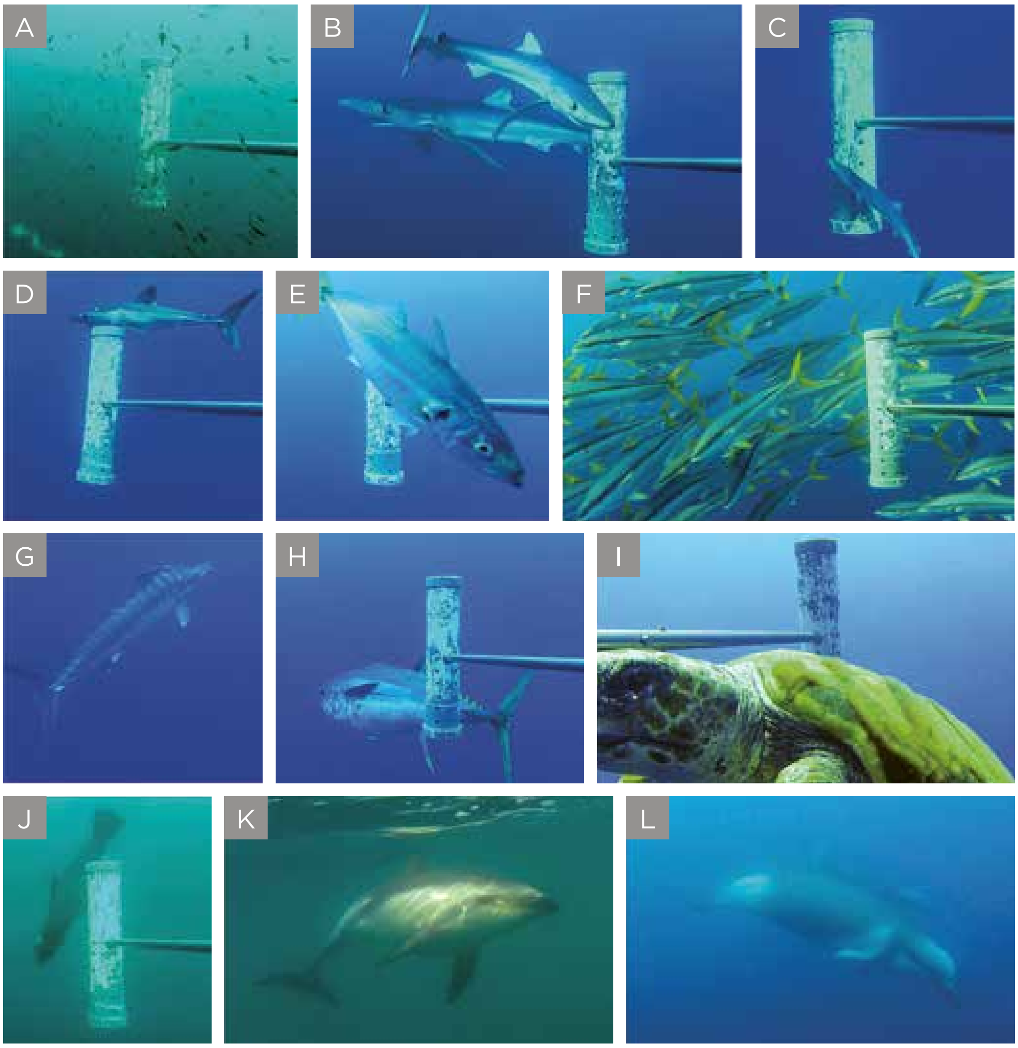


FIGURE 17.

Photos depicting representative species observed on pelagic camera drops at the Tristan da Cunha Islands. (A) Krill school (*Euphausiidae* spp.), Gough; (B) Blue sharks (*Prionace glauca*), Nightingale; (C) Recently born blue shark (*Prionace glauca*), Tristan da Cunha; (D) Porbeagle shark (*Lamna nasus*), Tristan da Cunha; (E) Southern horse mackerel (*Trachurus longimanus*), Inaccessible; (F) Yellowtail amberjack school (*Seriola lalandi*), Tristan da Cunha; (G) Striped marlin (*Kajikia audax*), Tristan da Cunha; (H) Yellowfin tuna (*Thunnus albacares*), Tristan da Cunha; (I) Loggerhead turtle (*Caretta caretta*), Tristan da Cunha; (J) Subantarctic fur seal (*Arctocephalus tropicalis*), Gough; (K) Dusky dolphin (*Lagenorhynchus obscurus*), Gough; (L) Shepherd's beaked whale (*Tasmacetus shepherdi*); Inaccessible.

Mean total abundance per site was much higher in the northern islands than at Gough with 14.75 ± 5.43 SE and 2.17 ± 0.37 SE individuals respectively. Similarly mean species richness was higher in the northern islands at 1.85 ± 0.17 SE species compared to 1.17 ± 0.15 SE. Within the standard two hours of video, Gough had only two species recorded, blue sharks and fur seals, additionally penguins and dolphins were observed after three hours. A different and more diverse assemblage was noted in the northern islands, with tuna, marlin, yellowtail and beaked whales observed among others. However, it should be noted that more sites were sampled in the northern islands than at Gough so this could account for some of the differences.

Blue sharks were more abundant at Gough Island than the northern islands, they were observed at 100% of sites with an average MaxN per site of 2 ± 0.41 SE individuals, compared to 0.55 ± 0.17 SE individuals at 45% of sites in the northern Islands. None of the other species observed in the northern Islands were observed at Gough.

Mean length of horse mackerels was 7.8cm (Table 11), with a single large individual recorded at 56.5cm. Mean length of blue sharks was 140cm, with both small juveniles and larger individuals observed (36.6 to 217.2cm range). The reported size at birth is 30 to 42cm, indicating that the smallest individuals were born recently. Yellowtail amberjack had a mean length of 87.3cm with a range of 64.8 to 126.9cm. Mean length of yellowfin tuna at 157.6cm was similar to the common length reported of 150cm. The single porbeagle shark recorded measured 95cm which is well below the reported common length of the species of 244cm but larger than its size at birth (61 to 69cm; www.iucnredlist.org/details/11200). At 3.4cm the single crested bellowfish measured was likely a juvenile. One Shepherd's beaked whale was measured at 2.5m, this is much less than their maximum reported size of 7.1m, suggesting that it may be a young individual.

In some cases, the rigs were left to drift after the standardised two hour period. This additional footage was scanned to check for additional species. This extension yielded 106 additional individuals, new species included a single bluenose (*Hyperoglyphe antarctica*) of 84.8cm at Nightingale Island, a northern rockhopper penguin (*Eudyptes moseleyi*) of 36.7cm, and five dusky dolphins (*Lagenorhynchus obscurus*) (Figure 17) at Gough Island. There is a large population of northern rockhopper penguins on Gough Island and large pods of dusky dolphins were seen at the surface on multiple occasions while sampling on the northern side of the island. All other observations were of species already observed.

TABLE 11.

Measurements of species observed in mid-water BRUVS surveys at Tristan da Cunha, total number of individuals measured, mean length for each species and associated standard error (SE). All measurements are presented as fork lengths.

Common name	Species	Number of Measurements	Mean length (cm)	SE
Subantarctic fur seal	*Arctocephalus tropicalis*	1	103.51	-
Loggerhead turtle	*Caretta caretta*	1	55.93	-
Porbeagle shark	*Lamna nasus*	1	95.03	-
Pilot fish	*Naucrates ductor*	1	21.14	-
Crested bellowfish	*Notopogon lillei*	1	3.41	-
Blue shark	*Prionace glauca*	35	139.98	7.22
Oval driftfish	*Schedophilus velaini*	1	30.21	-
Yellowtail amberjack	*Seriola lalandi*	25	87.29	2.67
Shepherd's beaked whale	*Tasmacetus shepherdi*	1	249.68	-
Yellowfin tuna	*Thunnus albacares*	2	157.60	12.78
Horse mackerel	*Trachurus* sp.	81	7.80	0.64

Shark Research - Movement Studies

The status of sharks at Tristan da Cunha and outlying islands is unknown. The migratory pathways and large-scale movements of these animals are of important ecological and management significance. Several species have been observed in Tristan's offshore waters including blue sharks (*Prionace glauca*), shortfin mako sharks (*Isurus oxyrinchus*) and hammerheads (Andrew et al. 1995). Pelagic sharks are frequently caught as by catch in long line fisheries and recent work in the North Atlantic has documented high spatial overlap in pelagic sharks and longline fleets (Queiroz et al. 2016). Movement patterns of many pelagic shark species have been extensively studied in the North Atlantic (Carey et al. 1990, Kohler et al. 2002, Queiroz et al. 2005, Campana et al. 2011, Vandeperre et al. 2014, Vaudo et al. 2016) but much less is known about movement and habitat use in the South Atlantic and what is known about shark distributions and migration patterns across the South Atlantic is largely reliant on fishery-dependent catch data (Hazin et al. 2000, Montealegre-Quijano and Vooren 2010, Joung et al. 2017). Movement information from tagging of pelagic sharks in the South Atlantic is virtually non-existent (but see da Silva et al. 2010). For the first time, we tagged and will track movements of Blue sharks and shortfin mako sharks in and around the remote Tristan archipelago. We used miniPAT pop-up archival transmitting tags (Wildlife Computers). MiniPAT tags are archival tags that collect sensor data and transmit the entire data package upon release from the animal. Detailed methods are provided at the end of this report (See Methods).

We tagged three female blue sharks at Gough Island that ranged in size from 189 to 217cm TL. We caught four additional blue sharks that were all juvenile males and deemed too small to tag (Table 12). These ranged in size from 98 to 146cm TL. During fishing activities and on the pelagic BRUVs we observed large numbers of small blue sharks (with the smallest individual measuring 36.6cm on a pelagic BRUV set). Following the expedition, the Tristan fisheries and conservation departments tagged an additional shortfin mako shark and four tags remain to be deployed. Movement data should be available when the tags pop off around mid to late August, 2017.

TABLE 12.

Number and species of sharks caught at Gough and Tristan islands during the expedition. Some sharks were too small to tag with satellite tags.
TL = Total length;
FL = Fork length;
PCL = Precaudal length.

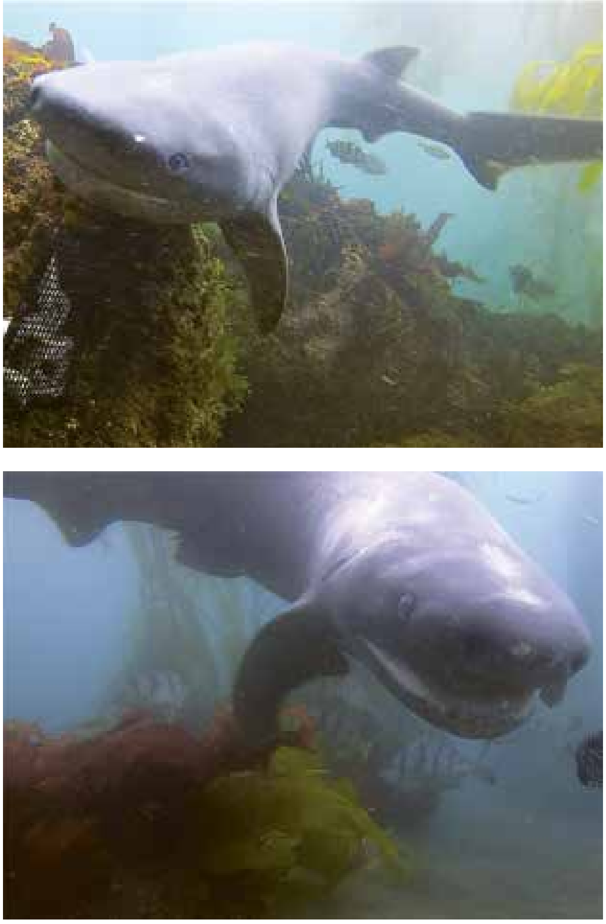
Species	Date	TL/FL/PCL	Sex	Lat	Long	Tag Serial
Blue shark, *Prionace glauca*	23-Jan-17	142/117/108	M	-40.32523	-9.82439	n/a
Blue shark, *P. glauca*	23-Jan-17	107/95/88	M	-40.32841	-9.82032	n/a
Blue shark, *P. glauca*	24-Jan-17	211/175/160	F	-40.28879	-9.86909	16P1720
Blue shark, *P. glauca*	23-Jan-17	146/114/105	M	-40.29512	-9.86619	n/a
Blue shark, *P. glauca*	23-Jan-17	189/150/138	F	-40.30183	-9.85909	16P1794
Blue shark, *P. glauca*	23-Jan-17	217/179/159	F	-40.31792	-9.83323	16P1721
Blue shark, *P. glauca*	23-Jan-17	98/78/70	M	-40.31673	-9.83498	n/a
Shortfin mako shark, *Isurus oxyrinchus*	31-Mar-17	158/140/125	F	-37.04678	-12.3282	16P1139

NEARSHORE SHARKS

Sevengill sharks (*Notorynchus cepedianus*) are the most common sharks in the nearshore environment at the Tristan group and Gough (Andrew et al. 1995). Despite reports that divers are regularly approached by this species at the islands, the survey divers observed only one shark at a distance at Nightingale Island (in over 70 dives made by the dive team). However when baited remote underwater video (BRUV) cameras were deployed at Gough Island multiple individual sevengill sharks were observed (Figure 18). The cameras are baited with just two small mackerel so the area of attraction of the bait is not large, and sharks arrived at the BRUVs after a short time, indicating that the sharks were in the vicinity of the location of the BRUVs when they were dropped. The underwater media team also encountered a number of sharks when they filmed using a bait attractor. These same areas had been recently (within two days) surveyed by divers without shark encounters. Divers can affect shark behavior in many ways which can impede the ability for census (Bradley et al. 2017b), so often methods that don't utilize divers are better for estimating shark abundance (Bradley et al. 2017a).

FIGURE 18.

Sevengill sharks (*Notorynchus cepedianus*) recorded on baited remote underwater video cameras at Gough.



Invasive Species Research – Surveys, Diet and Life History Studies

Following the wreck of an oil platform in 2006 on the south coast of Tristan, several marine introduced species have occupied Tristan's waters. In particular, a non-native fish that arrived with the rig, the South American silver porgy (*Diplodus argenteus argenteus*) has bred rapidly in Tristan waters and is now well-established around Tristan (Scott 2016). The extent of dispersal around the islands was reported in 2016, and at that time the porgy was said to have established all the way around Tristan (Scott 2016). They were reportedly confined to the shallowest 6m of water at that time. Aspects of the ecology such as diet and growth rates were previously unknown although Tristan islanders feared that porgy could be preying on juvenile lobsters. Here we provide the first systematic and quantitative results on presence/absence, density and biomass of the invasive porgy at all four islands as well as a preliminary study of feeding and growth.

SCUBA SURVEY RESULTS

Porgy abundance was quantified on diver transects and presence/absence was recorded at each survey site in case densities were low and porgy were not recorded on any transects. We found that the majority of sites on Tristan (9 out of 11) had porgy present (note that we documented presence of porgy when observed at a site even when not counted on the quantitative transect). We also documented porgy at Inaccessible, where, to our knowledge they had not been seen previously (4 out of 7 sites). Porgy were found at all depths of surveys (from 20 to 10m) and did not appear to be restricted to shallow depths as previously noted. However, we did not conduct quantitative surveys shallower than 10m, so densities could be underestimated in our surveys. Island means were 7.1 ± 3.5 fish/100m^2 at Tristan and 1.3 ± 1.0 fish/100m^2 at Inaccessible (Table 13).

TABLE 13.

Microplastics number per liter water sampled and frequency in samples by island.

Island	Mean Number Pieces per liter	Percent samples with microplastics	N samples
Tristan Offshore	0.59	75	4
Tristan Rockpools	1.59	100	2
Inaccessible	0.79	80	5
Nightingale	0.39	50	4
Gough	1.15	100	4

DIET, AND AGE AND GROWTH RESULTS

We also performed targeted collections (spear) and conducted gut contents, age and growth measurement and stable isotope analyses to investigate the diet with particular focus on the potential for newly settled lobster in the porgy diet. Fish were measured, weighed, and sexed onboard the ship and guts, otoliths and muscle and fin tissue were preserved for analysis in the lab. Detailed methods are below (Methods). We used otoliths to investigate the age and growth of collected samples and compared these results to information from the native range of this species (i.e. South America). These studies are ongoing and results will be reported when concluded. Preliminary results of the gut contents of the porgy showed that the species is almost solely herbivorous at Tristan Island. The only evidence of invertebrates appeared to be epiphytic organisms, likely growing on the ingested algae. We found a single larval lobster in the gut of one porgy. However, we also sampled guts of other fish species and did find a recently ingested juvenile lobster in the guts of both the false jacopever (*Sebastes capensis*) and the five finger (*Neadactylus monodactylus*) (Figure 19). These two native fish species were both previously known to consume juvenile lobsters in the Tristan Islands (Pollock 1991, Andrew 1992).

FIGURE 19.

Juvenile lobster (*Jasus tristani*) collected from the gut of an adult five finger (*Nemadactylus monodactylus*).

Microplastics - Methods and Results

Given the increasing levels of plastic pollution of the oceans it is important to better understand the impact of microplastics in the ocean food web (Ryan 2016). We partnered with National Geographic Emerging Explorer Gregg Treinish from Adventurers and Scientists for Conservation (ASC; www.adventurescience.org) to analyze water samples for the presence of microplastics at the Tristan Islands. We collected samples of seawater in 1L bottles at 19 locations, slightly offshore of our SCUBA survey sites. Two samples were collected from the Rockpools immediately adjacent to the Harbor at Edinburgh of the Seven Seas (i.e. the settlement) by the students of St. Mary's school.

Samples were sent to ASC in Maine, USA for processing. Once received, the water is vacuum pumped over a gridded 0.45 micron filter and dried from a minimum of 24 hours. Using a microscope at 40x magnification, pieces of microplastic (< 5mm) on the filter are systematically counted along the grid lines. Each plastic piece is categorized based on shape (round, filament/microfiber, other) and color (blue, red, green, black, transparent/white, other). The volume of water is recorded and the final count for the sample divided by the quantity of water to obtain a density estimate for each piece.

Microplastics were found in 15 of the 19 samples. The number of microplastic pieces per liter of water ranged from 0 to 2.3 (overall mean 0.83 pieces/L). Microplastic type included clear, blue and black filaments of various sizes. There were no strong patterns between islands but caution must be taken when interpreting these results because sample volumes and sample numbers are very small (Table 12).

Pinniped Research

Predator-prey interactions are of great interest in ecosystem studies, however, little is known about the marine predator-prey interactions at the Tristan da Cunha Islands. Fur seals inhabit top trophic levels within the ecosystem and therefore can act as good indicators of ecosystem shifts and threatened environments (Boyd et al. 2002). Fur seals travel between breeding and feeding grounds and when these areas are disturbed and resources are depleted, their populations are subsequently affected (Beauplet et al. 2004). Population fluctuations of these predators at the various colonies therefore largely reflect the relative availability of prey (Boyd et al. 1994, Wege et al. 2015, Wege et al. 2016a) within its foraging range.

METHODS

(a) Collection of scats deposited by seals on land allow for a non-invasive measurable sample which contains prey hard parts for identification (Klages and Bester 1998, Casper et al. 2006), ultimately representing the availability of prey in specific foraging areas (Makhado et al. 2008, Bester et al. 2011, Makhado et al. 2013). **(b)** Molecular techniques, such as stable isotope analysis, provide an integrated view of seal diets over different temporal and spatial scales (Cherel et al. 2009, Cherel et al. 2010, Bowen and Iverson 2012, Lübcker 2015). Serial sampling of archival tissues (such as whiskers) can be used as a means of acquiring high resolution information on the foraging strategies of individuals, as well as the prevalence of dietary specialization within and among populations (Newsome et al. 2010). The biogeochemical data (stable isotope signatures) captured along the length of vibrissae can be utilized to study the diet (Cherel et al. 2009, Kernaléguen et al. 2012, Walters 2014). **(c)** Poor foraging conditions often result in longer foraging trips by fur seal mothers, and slower pup growth (Boyd et al. 1997, McCafferty et al. 1998, Kirkman et al. 2003, Lea et al. 2006) affecting pup body mass (Georges and Guinet 2000a, Oosthuizen et al. 2016) which conceivably affects their foraging ability, and ultimately influences future survival (Georges and Guinet 2000b, Guinet and Georges 2000, Arnould et al. 2003). Satellite tracking of the at-sea movements of lactating female Subantarctic fur seals, both from the incipient population at TdC, juxtaposed with the much larger population at 380km distant GI, will inform on their foraging ranges, pup attendance, and the measurable result thereof, pup growth. **(d)** Total island counts of fur seals have never been completed in any year on the Tristan da Cunha Islands, and in the present study pups were counted directly (on foot, or from RIB close inshore) on accessible beaches so as to either compare counts amongst years (previously counted), or to set a baseline value (not previously counted).

RESULTS

(a) Scats were primarily collected from Reef Point and Seal Beach (Gough), the two unnamed beaches on either side of Cave Rock (Inaccessible), at West Landing (Nightingale) and at Cave Point, Seal Bay (Tristan). The approximate total of 200 scats will be processed at the MRI, UP to isolate hard prey remains for subsequent identification and measurement (see Methods). **(b)** Whiskers and hair samples were collected from 10 adult females (seven with attending pups), and whole blood samples from three of these females. The samples await biogeochemical analyses for stable isotope content. **(c)** The ten adult females above were instrumented with satellite tracking devices, three each at Tumbledown Beach and north of Deep Glen (Gough), and four at Cave Point (Tristan). Tracking data (via CLS Argos) started coming in from end-February, and final analyses of the full set of tracking information will commence from May 2017 when the instrument batteries are expected to have run flat. Pups, at approximate mean age of 50 days, were weighed (n = 60) at Cave Point (Tristan) only, while those at Tumbledown Beach (n = 100) were weighed the week before by RSPB personnel from Gough Base (both localities are included in a long-term study of pup body growth). This early after the breeding season, the expected divergence of pup growth from Gough (high colony density) and Tristan (low colony density) is unlikely to have taken place (see below). The data has been added to the long-term database. **(d) Gough -** compared to end-January 1978, uncorrected pup numbers at the Tumbledown Beach area (n = ~24 vs. 591), Buttress Rock to Dell Rocks (n = a few at most vs. 724) and the north-east beaches north of Deep Glen (n = 0 vs. 5,199) have increased exponentially, while other small (Seal Beach, n = 1 vs. 2) and large open beaches (Reef Point to Deep Glen, n = 0 vs. 16; Capsize Sands, n = 0 vs. 8) have shown virtually no increases over 40 years. These largely open boulder beaches on the leeward northeastern sector of the island, which are of similar physiognomy to those beaches with large concentrations of fur seal pups counted in the present survey, have not developed into breeding colony beaches in the intervening period. The factors responsible for such demographic peculiarities in Gough's fur seal population are unknown. **Inaccessible -** Although no baseline counts at the remaining Tristan da Cunha Islands are available for the 1970s, there is still a preference in the selection of beaches for pupping, such as at Inaccessible (Blendan Hall to Warren's Cliff, n = 446; Blendan Hall to West Point, n = 1). On the south-eastern aspect of Inaccessible only some (n = 35) pups were clustered at East Point, and then again on the approach to Cave Rock (n = 166). The entire section in between these points, such as Tom's Beach, was devoid of pups. South-east of Cave Rock there was a concentration of 345 pups towards South Hill. **Nightingale –** Except for the area directly in front of the 'Huts', the un-named eastern beaches from West Landing (n = 17) towards Pequena Point, all had pups ashore (estimated from RIB close inshore). These counts (n = 1126) are augmented by those counts at informally named 'Bester's Beach' (n = 1509), counted directly (on foot), further along the eastern section, short of reaching Pequena Point. **Tristan da Cunha** – Pups occur only at Cave Point, and current pup population is estimated at ~75, perhaps up from earlier counts (but in the month of September, by when mortalities would have taken effect) of some 40 pups.

The above estimation of fur seal pup numbers ashore in January 2017 is a rough approximation of the true situation. Adjustments to 'boat-based' counts as opposed to 'beach-based counts' are likely to be 40–50%, and undercount corrections to augment the 'beach-based' estimates must also be effected (neither of which is presented here).

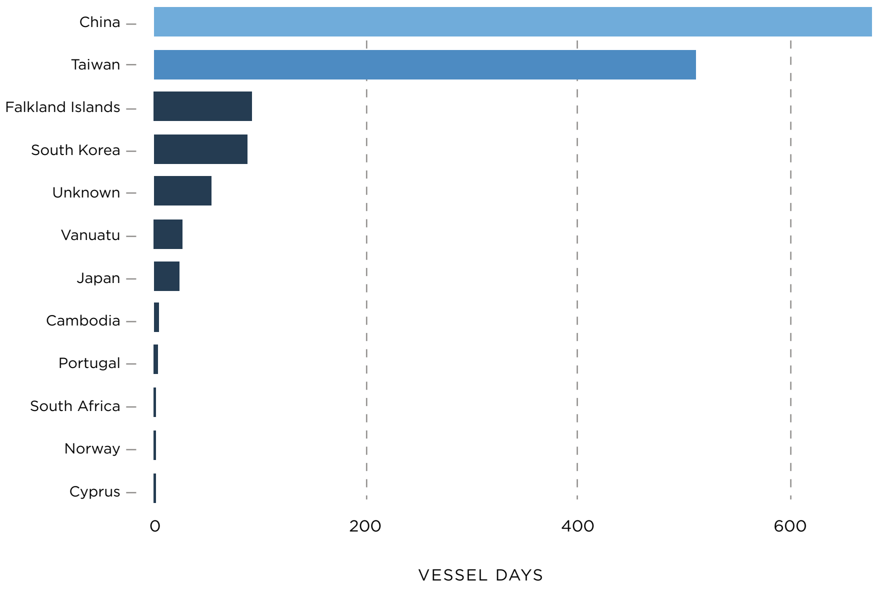
Fishing Effort and Vessel Traffic

We examined the magnitude and spatial distribution of vessel activity and fishing effort in and around Tristan's EEZ using data from Global Fishing Watch (GFW; 2014–2016). We created a dataset of 'likely fishing vessels' containing boats whose identity has been matched to fishing registries, who identify themselves as fishing vessels, or whom GFW's machine learning algorithms classify as fishing vessels.

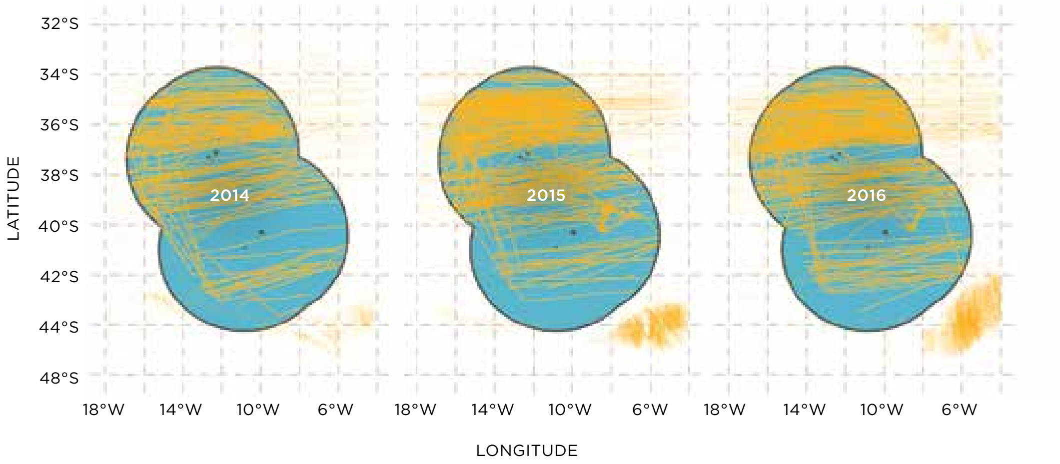
FIGURE 20.

A) Number of days spent in the Tristan da Cunha EEZ for vessels flying flags of different countries in the period 2014–2016. B) Tracks of all fishing vessels activity in the Tristan da Cunha EEZ from 2014–2016, by year.

A. Total Vessel Days



B. FIshing Vessel Tracks



During this time period, we are able to observe 253 vessels, from 12 different flag states, spent a total of 1495 vessel-days inside Tristan's EEZ (Table 14). The majority of these vessels fly flags of China (142), Taiwan (64), South Korea (16) and the UK (Falkland Islands, 3) (Figure 20A) and most of their time inside the EEZ is spent transiting; predominantly in the northern region (92% of days) (Figure 20B). A small subset of these vessels (11) appears to be actively fishing inside the EEZ for a total of 107 days (7.1%); most of this fishing activity was by UK Trawlers and Japanese long liners (Figure 21) in the eastern portion of the EEZ (Figure 22).

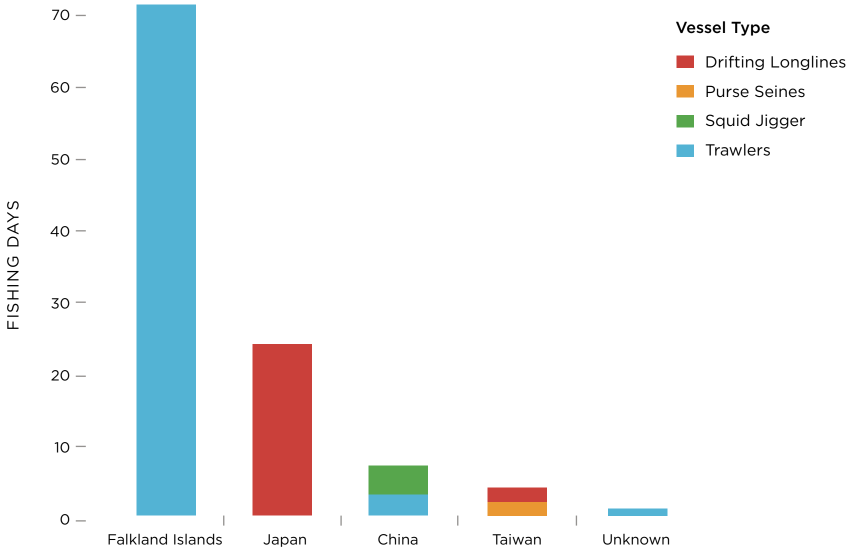
TABLE 14.

Total vessels, total number of days, fishing vessels and fishing effort (days) inside Tristan's EEZ for the period 2013–2016. Fishing days are for vessels that were likely fishing at least once during the annual time period.

Year	Total Vessels	Days	Fishing Vessels	Fishing Days
2014	111	353	0	0
2015	149	623	6	76
2016	142	519	7	31

FIGURE 21.

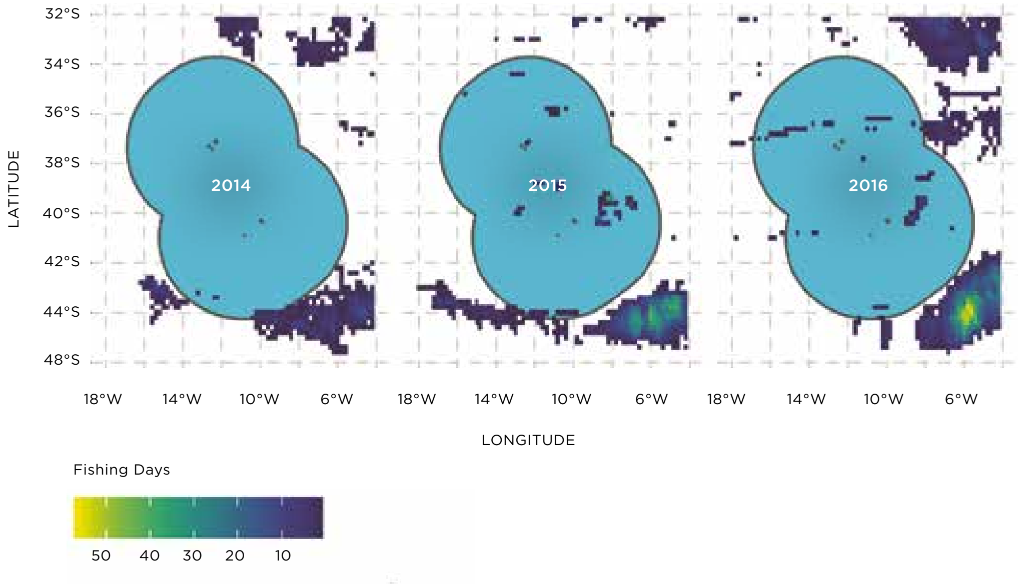
Fishing days inside Tristan's EEZ by flag state and gear type (2014–2016). Included here are vessels that were likely fishing at least once during the entire time period.



In stark contrast, on the waters surrounding Tristan's EEZ, we observe a total of 62 fishing vessels, from 8 different flag states, spend a total of 1987 fishing days. Most of this fishing activity takes place on seamounts to the southeast and southern boundary of EEZ and is done mostly by long liners from Japan, South Korea, and Spain (Figure 22).

FIGURE 22.

Fishing intensity (days) in and around Tristan's EEZ from 2014–2016, by year. Included here are vessels that were likely fishing at least once during the entire time period.

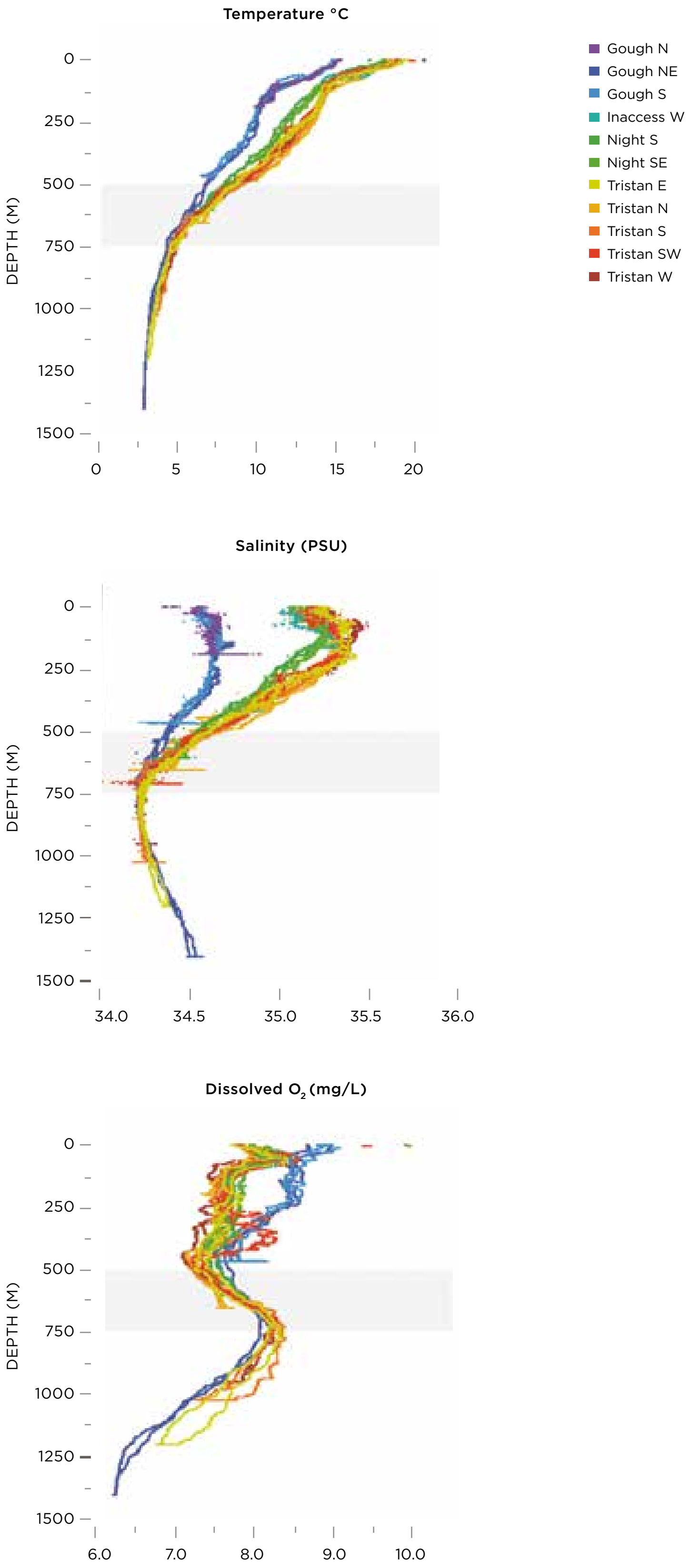


Oceanography

We measured various properties of the water column with a CTD attached to the deep-water drop cameras. We conducted 11 profiles across all islands (Gough N = 3; Inaccessible n = 1; Nightingale n = 2; Tristan da Cunha n = 5). As expected, temperature and salinity were different at Gough relative to the northern islands; Gough waters were colder and less saline (Figure 23). Surface (3–10m) mean water temperatures were 14.9°C at Gough, 17.9°C at both Inaccessible and Nightingale, and 18.6°C at Tristan da Cunha. The depth at which both temperature and salinity equilibrated among islands was approximately 600–750m, also the depth range at which we saw a divergence in community structure from the deep-sea cameras (Figure 16). Nightingale Island, which had similar surface temperature to Tristan da Cunha, diverged and became colder and less saline at approximately 100m depth and remained so to about 500m. Inaccessible may have shown the same pattern as Nightingale but our only drop at that island extended to 164m. Patterns of dissolved oxygen were more difficult to interpret and less variable among islands (Figure 23).

FIGURE 23.

Oceanographic properties of the water column from a CTD attached to the deep-sea cameras. Temperature (C), salinity (PSU) and dissolved oxygen (mg/L) from each island by shore orientation are shown. The depth at which both temperature and salinity equilibrated among islands (approximately 600–750m) is shown in the grey band.



DISCUSSION

DISCUSSION

The Tristan da Cunha Islands are a unique archipelago with healthy marine ecosystems. As a result of their extreme isolation, they are species poor (Pollock 1991). This remote temperate archipelago provides one of few places in the world to establish a baseline for unimpacted temperate systems. Quantitative data from the kelp forests was lacking and pelagic habitat and deep benthos were mostly unexplored prior to this expedition. We found that, despite an important commercial fishery for lobster and subsistence fishing for local islanders, marine habitats and biota appeared in very good condition. Biomass of fishes and lobster in particular were high. However, this unique marine ecosystem is not without potential threats: shipping traffic leading to wrecks and species introductions, pressure to increase fishing effort beyond sustainable levels and climate change all could potentially increase in the coming years. Currently, the low population density, difficult access to the marine environment and a pro-active, well-managed lobster fishery provide a level of protection to nearshore habitats. However, offshore areas including seamounts, would benefit from strong, enduring protection.

Marine Ecosystems

NEARSHORE REEFS

Although there are few temperate island systems to compare to, and none as isolated as Tristan, we found that levels of fish biomass were comparable to island sites in southern California's Channel Islands. The northern Channel Islands (NCI) are broadly similar in habitat (e.g. *M. pyrifera* is the dominant biotic habitat former) and sea temperatures, and the region has a large amount of available, long-term monitoring data (Hamilton et al. 2010, Caselle et al. 2015). Despite very large differences in fish species richness (2 to 4 species per site in TdC and 11 to 22 per site in NCI), total fish biomass in the Tristan Islands is comparable to Channel Island sites. Thus, for the few individual species observed at Tristan, abundance and biomass is extremely high. The total fish biomass across Tristan da Cunha Islands ranged from 1.5 to 2.75mt/ha while the average biomass across the NCI locations was 1.2mt/ha. The highest recently recorded biomass for the Channel Islands was 2.7mt/ha and this was inside a Marine Protected Area (MPA; Caselle et al. 2015). Lobster density was also extremely high at TdC with 6.4 to 8.8 lobsters per 100m^2 across the islands compared to an average density of lobsters in the NCI (a different species to Tristan, *Panularis interruptus*) of 0.66 per 100m^2. In California, the highest density of lobster ever observed on an annual survey in 15 years of monitoring was 9.1 lobsters per 100m^2 and this was in a very old MPA (Caselle unpub. data). At Tristan, lobsters were present on every single transect surveyed. Lobsters were also often found on the holdfasts of giant kelp (Figure 24), indicating a potentially important species-habitat association.

The expedition divers noted that in Tristan's islands, adult lobsters were often found roaming during the daytime and not exclusively located in cracks or crevices, something rarely if ever observed in California or other locations with intense lobster fishing. During the expedition, several small lobster pots were set at Tristan for personal consumption and we observed more than 30 lobsters caught with soak times as little at three hours. Our experience in California is that even larger traps set for several days would not catch that number of lobsters. However, it should also be noted that the same duration trap sets at Gough during the expedition all came up empty, indicating some variance in CPUE across islands. CPUE in the commercial fishery is measured and data are analyzed by the fisheries department to help set annual TACs (James Glass, personal communication).

Giant kelp, *Macrocystis pyrifera* was the major habitat-forming species across the Tristan Islands, with the pale kelp, *Laminaria pallida* forming very dense sub canopy forests at some sites. Both species are reported to be prominent components of the diet of Tristan lobsters (Pollock 1991). Interestingly, giant kelp plants across the Tristan Islands were composed of single or very few stipes, compared to other regions of the world where each plant can be massive, often containing hundreds of individual stipes. Stipe number is a measure of growth in Macrocystis and is sensitive to temperature,

FIGURE 24.

Lobsters (*Jasus tristani*) on the holdfasts of giant kelp (*Macrocystis pyrifera*).

depth, latitude, degree of wave exposure, upwelling, and other factors (Jackson 1987, Tegner et al. 1997, Rodriguez et al. 2013). Divers also observed that the giant kelp at Tristan appeared to be extremely tough, with blades that resisted tearing, very thick stipes and numerous floats, which could indicate local adaptation to sea conditions. Sea conditions at Tristan da Cunha are notoriously rough; storms, large waves and very high winds are common and it is likely that *M. pyrifera* suffer frequent high mortality events. It has been suggested that plants with low stipe counts might have higher survivorship due to a lowered rate of entanglement and dislodgement (Utter and Denny 1996). However, single stipe plants are likely young individuals, and the turnover of giant kelp in this region may be very high. The consequences of this growth form for resilience to climate change induced effects (which may include increased sea temperatures or increases in waves and sea state – but this remains to be investigated further) are currently unknown.

Within each island, we found a remarkable lack of variation in measured habitat features such as benthic substrate and physical relief. Although we did focus on rocky reefs that had emergent kelp canopy and avoided large expanses of sand (which do exist between reefs), almost all sites surveyed were composed of moderate or very large sized boulders and very low cover of sand or cobbles. Even within our broad categories of physical relief, the average relief across islands was similar (Figure 4). There are virtually no flat areas on these reefs. These boulder reefs provided ample crevices and large overhangs that were utilized extensively by lobsters (Figure 25). We also found very similar island-wide averages of kelp plant and stipe density, despite large differences in temperature between the northern islands and Gough. We hypothesized that Gough would have thicker, healthier kelp beds with colder temperatures but this was not found. It is likely that the primary driver of kelp abundance in this system is physical stress due to chronic, year-round wave and storm impacts (Cavanaugh et al. 2011, Reed et al. 2011, Bell et al. 2015).

FIGURE 25.

Lobsters (*Jasus tristani*) utilizing cracks and overhangs.

The extreme lack of biodiversity in the kelp forests of Tristan results in a very simple, short-chain food web with limited ecological interactions. While detailed feeding habits and information for some of the key species in Tristan's kelp forests is unknown (we collected guts and muscle tissue for many of these species and detailed predator-prey relationships will be investigated in the future), we can hypothesize the linkages based on the few existing studies and on known prey preferences for these taxa in other locations (Baxter 1960, Heydorn 1969, Ebert 1991, Pollock 1991, Andrew 1992, Andrew et al. 1995, Smith 2003, Barrientos et al. 2006, Dubiaski-Silva and Masunari 2006). Our surveys quantified the likely key components of this simple food web, with kelps and benthic algae at the base providing both food and shelter; urchins (and possibly the invasive Porgy) as the primary grazers; five finger, false jacopever, wrasses, lobsters and octopus as the main carnivores; sevengill sharks (we counted none on transects) and possibly yellowtail as the primary piscivores; and lobsters as omnivores, consuming both algal material and benthic invertebrates. Potentially important transient species include seals, sealions and rockhopper penguins which may also feed in the kelp forests. Simple food webs such as Tristan's are far less resilient in the face of perturbations that might reduce or remove one of the links than systems with greater functional redundancy (Borrvall et al. 2000, EklÖF and Ebenman 2006). For example, though many specific trophic linkages in Tristan's kelp forests are currently unknown, the sevengill shark is known in other parts of its range to prey on benthic associated teleosts, especially at smaller size classes (Ebert 2002). As one of very few shallow reef associated, piscivorous species, it may be particularly important to controlling populations of the reef fishes that prey on lobsters. The lack of functional redundancy in the piscivore group may place disproportionate importance on particular species in this simple food web.

We identified two key members of this food web that might be at risk. Some areas of the world have experienced massive declines in giant kelp (*M. pyrifera*) (Johnson et al. 2011, Wernberg et al. 2011, but see Krumhansl et al. 2016, Reed et al. 2016) and in

Tristan's northern islands this species may currently be at or near its thermal limits. Future increases in temperature could place this species at risk with unknown but likely dramatic effects on the entire ecosystem. Pale kelp (*L. pallida*), while abundant and more resilient to higher sea temperature, might increase in abundance but its relationship to other members in the community and its ecological importance is less well known. We observed urchins grazing only on pale kelp, not on giant kelp and never observed lobsters on the pale kelp. Lobsters are the other key species that may be at risk from both fishery effects and climate change. Currently the lobster fishery appears extremely well managed and with conservative allowable take. However, many important life-history attributes such as recruitment and growth rates are not known and could be affected by climate-related changes. Lobster are the purported predators on urchins in this system (although some fish species might also prey on urchins) and it has been demonstrated in many temperate systems that the loss of this key species can transition systems to urchin barrens (Shears and Babcock 2002, Lafferty 2004, Ling et al. 2009b). For simple food webs with little functional redundancy and hence resilience, it is critical to ensure that the key players remain at healthy levels of abundance (Dunne et al. 2002, Jordan et al. 2002).

Redistribution of species is one predicted effect of climate change. For example, yellowtail (*Seriola lalandi*) have apparently increased recently in the northern islands (Scott 2016), and islanders report concurrent declines in snoek (*Thyrsites atun*). Divers on this expedition also documented the presence of a group of almaco jack (*Seriola riviolana*) which hadn't before been reported for the Tristan Islands (Figure 26; Andrew et al. 1995, Scott 2016). Increases in abundance and establishment of these two subtropical species could be indicators of changing ocean conditions. Species range shifts can have impacts on fisheries (reductions in existing fisheries but increases in new fisheries). In some systems, climate-driven species range shifts have had significant ecological impacts as well (Ling et al. 2009a, Wernberg et al. 2013, Vergés et al. 2014)

FIGURE 26.

Almaco jacks (*Seriola rivoliana*) at Tristan Island.

OFFSHORE OCEANOGRAPHY

The Tristan da Cunha Exclusive Economic Zone (EEZ) spans just over 3° latitude, encompassing approximately 754,000km^2. Due to the unique placement of the islands in the Southern ocean the EEZ also includes a range of oceanographic features and offshore habitats such as seamounts. A major feature of the offshore waters is the subtropical front (STF), which normally is located between the northern islands and Gough, with Gough sitting somewhere near or below the frontal region. The STF delimits the northern limit of the Antarctic Circumpolar Current (Orsi et al. 1995) and corresponds to the boundary of the Southern Ocean (following Deacon 1982). Although the STF may be weaker and more ephemeral in the Tristan EEZ relative to the south of Africa (Smythe-Wright et al. 1998, Llido 2005) it is still characterized by high productivity and can be considered a biological habitat on its own (Bakun 2006, Bost et al. 2009).

Recent work has demonstrated the importance of oceanographic features such as fronts to apex predators via enhanced primary and secondary productivity (Worm et al. 2003, Block et al. 2011). The Tristan Islands are a globally recognized region for seabirds and also home to pelagic sharks, tunas, cetaceans, seals, and sea lions. During our expedition we documented many of these offshore roaming marine species on our mid-water cameras including blue and porbeagle sharks; tunas, yellowtail and marlin; dusky dolphins, fur seals, turtles and the rare Shepard's beaked whale. The extent to which these species and the seabirds utilize the STF as a critical feeding area is unknown at this time but in other regions, these areas are important for population persistence for a number of species (Bradshaw et al. 2004, Bakun 2006, Bost et al. 2009). Unfortunately, fronts and convergences are also areas with high fishing pressure, primarily from longline vessels. Recent work in the north Atlantic showed strong preferences of pelagic sharks (tiger, blue, mako and hammerhead) for areas of high thermal gradients and high productivity (Queiroz et al. 2016). This study also showed that these shark hotpots overlapped significantly with longline vessel fishing activities and that the overlap persisted across seasons and years. Longline fishing is also responsible for the deaths of an astonishing number of seabirds annually (Anderson et al. 2011). For example, Cuthbert et al. (2005) estimates an annual mortality rate of 471 to 554 for the critically endangered Tristan albatross from interactions with longline fisheries in the South Atlantic, due to strong spatial overlap between fishing effort and albatross foraging grounds. While there is currently little to no longline fishing in Tristan's waters, protection of these offshore habitats would likely add significant future insurance for a large number of species that might be utilizing the frontal zone.

The STF is generally characterized by a surface temperature discontinuity of 4 or 5°C and a salinity difference of 0.5×10^{-3} (Deacon 1982) with warmer, more saline waters north of the front, partly derived from strong southward currents. Our oceanographic sampling confirmed a strong temperature and salinity difference between the northern islands and Gough (Figure 23). Surface water temperature ranged from 14.9°C at Gough to 18.6°C at Tristan da Cunha while surface salinity ranged from 34.57 PSU at Gough to 35.23 PSU at Tristan da Cunha. These differences likely lead to productivity differences among the island groups, with higher productivity at Gough, located in the front or convergence zone. Colder temperature and higher productivity can lead to faster organismal growth rates, likely explaining the larger fish and lobsters at Gough relative to the other islands. For example, marked variation in growth rates have been documented between the top islands and Gough for lobster and false jacopever (Pollock 1991, Andrew and Hecht 1996).

In addition to increased productivity, the subtropical front is potentially an effective barrier to dispersal of propagules between the northern and southern islands much of the year. Thus, there is potentially limited population connectivity between the northern islands and Gough. Populations of lobsters and fishes may be relatively isolated and may not serve as reliable sources for replenishment across the frontal zone.

Regrettably we were not able to sample any seamounts during the expedition. However, isolated seamounts are well known for high levels of biodiversity (Morato et al. 2010). Globally, there is increasing attention calling for protecting seamounts and their associated offshore systems. The Tristan EEZ contains many seamounts which should be considered in any protection scheme. The Tristan Fisheries department has plans to conduct fish biomass surveys on the seamounts, during the next phase of fishing which will inform a fisheries management plan for the islands.

PELAGIC AND DEEP-SEA HABITATS

The STF zone around the Tristan da Cunha Islands is a hotspot for pelagic diversity in the Atlantic Ocean, likely due to the convergence and high productivity of these oceanic fronts. While shipboard observations of seabirds, marine mammals and pelagic sharks have been conducted in the region, for the first time we surveyed the upper water column for pelagic predators using baited cameras. Our mid-water cameras recorded an average species richness of 1.7 and an average abundance (measured as N_{max}) of 11.8 individuals per sample set. Notably, blue sharks (*Prionace glauca*) were commonly observed in the pelagic zone, occurring on just over half of all camera deployments. Interestingly, we recorded a relatively high abundance of juvenile blue sharks, many of which were close to the size reported at birth, indicating that the smallest individuals we observed were born recently and that the islands may serve as a pelagic nursery for this species. Observations of rare Shepherd's beaked whales are the first underwater documentation of this species in the islands and the survey also yielded the first sighting of a porbeagle shark in Tristan waters. Similar to the nearshore reef surveys, the pelagic surveys showed differences in the assemblage structure between the northern islands and Gough. Krill, dusky dolphins, seals and penguins were present in samples at Gough Island while warmer water species such as tuna, marlin, turtle, and yellowtail were observed in the northern islands.

This expedition provided one of the first surveys of very deep habitats around Tristan (but see (Scott 2016) for list of all prior work, including deep-water scientific trawls). The deep-sea drop cameras indicated that species richness and abundance is positively related to the presence of hard rocky substrate (present on 35% of camera drops). Biogenic habitats were present at the deep-sea camera sites on 40% of the deployments, including sea pens, crinoids, whip corals, and small to very large gorgonians. Many of the taxa observed in the deep sea off Tristan da Cunha also occur in other locations around the globe where these camera systems have been deployed (e.g. (Friedlander et al. 2013, Friedlander et al. 2014), indicating some level of geographic uniformity in deep-sea fish assemblages. Species composition changed with depth, likely responding to changes in pressure, light levels, temperature, and oxygen saturation. We observed distinct differences in the fish assemblage structure above and below 750m depth. Our oceanographic sampling also showed a discontinuity or 'breakpoint' in temperature and salinity at approximately 500–750m depth. At this depth range, we found that both variation among islands and total variation became minimal with further increases in depth (Figure 23). Although levels of dissolved oxygen with depth showed more complex patterns, it was at this same depth range where among island variation was reduced. While our CTD measurements were a snapshot in time, this dataset is the first to our knowledge in the nearshore areas of Tristan Islands. Further work will be needed to understand the effects of ocean conditions on deep-water assemblages, however here we have identified depth zones between which both assemblage structure and oceanography showed marked differences.

Potential Threats: Vessel Traffic, Shipwrecks and Invasive Species

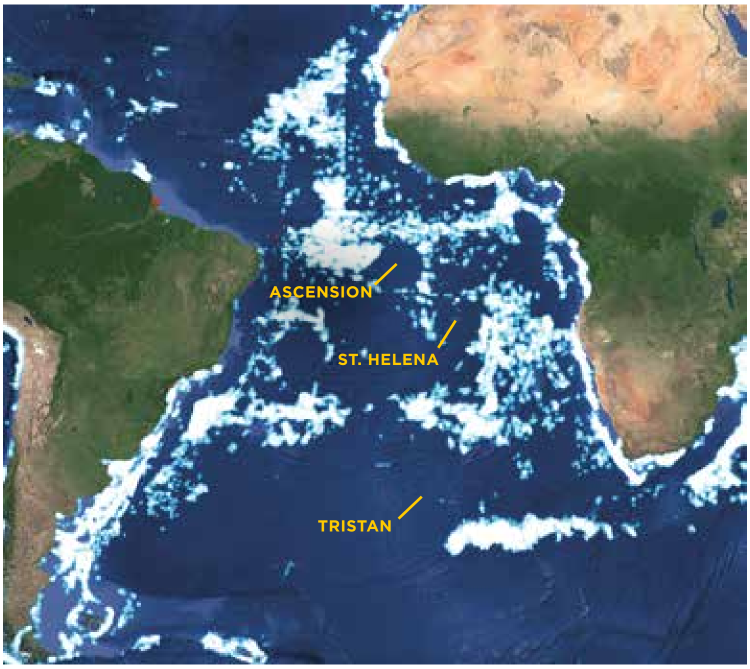
FISHING VESSELS

Results from our AIS analysis indicate relatively low levels of fishing throughout the EEZ of Tristan; with only three industrial vessels responsible for 88% of all fishing days (on average 19 days per vessel per year). In contrast, hotspots of fishing activity are observed near the borders of the EEZ, especially in the Southeast region.

This spatial pattern is similar to what has been observed in other island EEZs in the South Atlantic (e.g., Ascension, St. Helena; Figure 27) and suggests that Tristan is overall exerting the sovereignty of its waters, while at the same time controlling the development of its own industrial fishing fleet.

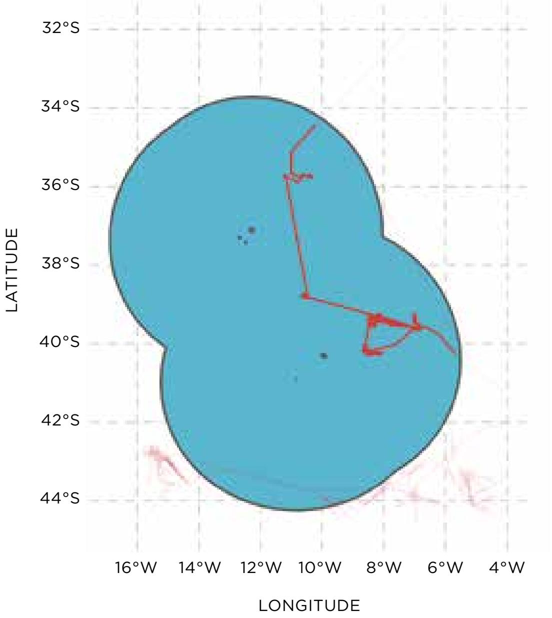
The only considerable evidence of foreign fishing activity happened in 2015 when we observed 24 fishing days by one Japanese longliner who was authorized to operate inside the EEZ (Figure 28). Nevertheless, foreign vessels straddle the southern borders of the EEZ and make occasional brief crossings into the EEZ. Thus, given the destructive potential that even a single longliner can have for pelagic fish populations — particularly sharks — it's important to continue strengthening the monitoring and enforcement of foreign fishing activity in and around Tristan's EEZ.

FIGURE 27.



Fishing intensity in the Atlantic, showing intensity (lighter colors indicate higher fishing intensity) around the EEZ's of several islands.

FIGURE 28.



Track of a Japanese longliner permitted to fish in Tristan's EEZ in 2015.

SHIPPING, WRECKS AND INVASIVE SPECIES

Despite its isolation, the Tristan da Cunha archipelago lies on an increasingly busy international shipping lane and has been heavily impacted by invasive species. Invasive and alien species are considered the single biggest threat to the terrestrial and avian biota of these and other islands throughout the world (Wanless et al. 2007, Jones et al. 2008, Doherty et al. 2016). However, much less is known about the extent and impacts of marine invasions, although several have been documented (Wanless et al. 2009) on the Tristan Islands. We surveyed several sites that had been impacted by vessel (or oil rig) groundings during the expedition and focused analyses on large-scale community impacts at those sites and the distribution and life history of a notable invader, the South American silver porgy (*Diplodus argenteus*).

In March 2011, the bulk carrier MV Oliva ran aground and ultimately broke up at Nightingale Island. The breakup of the vessel resulted in the discharge of 1,500 tonnes of heavy fuel and 65,000 tonnes of soya bean cargo (Scott and Franklin 2011). Both Nightingale and Inaccessible were affected; the lobster fishery was closed for the 2011/12 season at Nightingale and the TAC was reduced at Inaccessible that same season. We surveyed the kelp forest and reef areas around Nightingale and at the wreck site itself. Island-wide fish and lobster biomass at Nightingale were both well within the range found on the other islands. Lobster biomass at Nightingale was second highest, only to Gough. Giant kelp was also very abundant around the island, and the divers observed some of the densest kelp beds at Nightingale. However, the survey dives on the wreck showed some differences in the benthic community, to be expected in that we surveyed directly on the metal and wood structure of the wreck itself. The wreck site was isolated statistically from the other sites in terms of the benthic community composition (Figure 11B) and the differences were driven by high density of barnacles (on the metal structure of the wreck), very high density of pale kelp and the lack of giant kelp. However, as expected, the fish community did not differ greatly from other sites and lobster density at that site was average for the island. Although non-native mussels (*Mytilus galloprovincialis*) were found on the propeller shaft of the MV Oliva shortly after the wreck (Scott 2016), we did not observe any individuals of this species near the wreck or at any of our dive sites (although we did not survey the intertidal areas, only shallow subtidal reefs). The consequences of the wreck were not limited to the reef environment, but had major impacts on the northern rockhopper penguins (Scott 2016).

In 2006, a decommissioned oil platform that was being towed from Brazil to Singapore, broke free of its tow and ran aground on the Southeast coast of Tristan Island at Trypot Bay. Initial surveys showed that the rig was heavily fouled with organisms from the subtropical waters of Brazil (Scott unpublished data). Follow up surveys of the marine environment provided a comprehensive list of marine fauna associated with the rig and nearby areas (Wanless et al. 2009). We focused on one species of particular concern to the Tristan islanders, the South American Porgy. This fish species was known to have spread all around Tristan Island at the time of our surveys. We documented its spread to Inaccessible Island where it was present at four of the seven sites sampled. Porgy were found at all depths of surveys (from 20 to 10m) and did not appear to be restricted to shallow depths as previously noted. While we did not record this species at Nightingale Island, the proximity to Tristan and Inaccessible make it likely that the Porgy will spread to that island at some point in the future. We note that it is highly unlikely that the Porgy will spread to Gough Island due to the very large distance from the northern islands and the much colder water temperature.

The islanders expressed concern that the invasive Porgy could have negative interactions with lobster, potentially affecting the lobster fishery. The porgy favors shallow waters in its native range (David et al. 2005) and this is also the habitat for newly settled lobsters (Booth et al. 1991, Pollock 1991). Diet studies of this species in its native range showed that they are omnivorous, eating both algae and invertebrates and showed ontogenetic variation in diet (Dubiaski-Silva and Masunari 2004). In Brazil, diets of the smallest individuals were composed of copepods and small mussels, while the largest individuals at mainly algae and to a lesser extent, decapods. We heard reports of Tristan islander(s) finding larval or juvenile lobster in the stomach of a Porgy. Our observations of Porgy were primarily of small individuals (mean total length was 13.6cm, range 10 to 27cm). Our collections were also of smaller individuals although we did collect a single larger individual (total length 29.8cm). We saw no settlement sized or young of year individuals on our dives. While fish in the size range of 10 to 15cm are unlikely to have a gape that would allow predation on juvenile lobsters, certainly fish greater than 20 or 25cm would be capable predators. Our preliminary analyses on stomach contents of the porgy did not find any lobsters or lobster parts, the porgy appears to be solely herbivorous on Tristan. However, we also sampled guts of other fish species and did find recently ingested juvenile lobsters in the guts of both the false jacopever (Sebastes capensis) and the five finger (*Nemadactylus monodactylus*). *Sebastes* are predatory, carnivorous fish and are known to consume crustaceans elsewhere in the world (Murie 1995) and both species are reported predators of lobsters at Tristan (Pollock 1991, Andrew 1992).

Fisheries

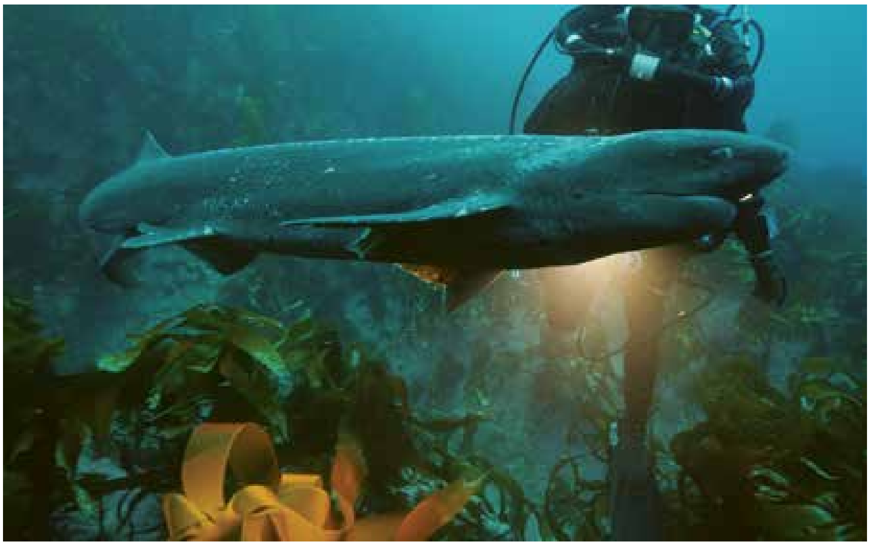
Fisheries are a mainstay of the Tristan economy. In particular, the MSC certified lobster fishery provides over 80% of the island's GDP, employing over 25% of the island population at peak times. The island community relies heavily on permitted vessels fishing around the island as a mean of revenue and transport to and from the island, unlike other overseas territories that have more frequent transport and receive more aid from the UK. The Tristan fishery department collects a large amount of data using experimental fishing before each season and from catch monitoring during the fishing season. Data on CPUE and size structure are then used to set annual catch limits (James Glass pers comm.). Our fisheries independent, visual survey data on lobster size structure corroborates the information on size structure from the fishery (Glass 2014) as well as a previous dive survey conducted in 1989 (Pollock 1991). Lobsters were largest at Gough, moderately sized at Nightingale and smallest at Inaccessible and Tristan Islands. These size differences likely reflect the differences in productivity among the islands. However, lower fishing pressure at Gough may also contribute to the presence of larger individuals at that island. SCUBA surveys provide an independent source of data to contribute to ongoing management of the fishery.

RECOMMENDATIONS

RECOMMENDATIONS

Due to Tristan's location in the South Atlantic Ocean, its EEZ includes a range of unique oceanographic features:

1. The subtropical front (STF) is an area characterized by high productivity that is likely an important habitat for apex predators and birds. The location of the STF is seasonally variable, shifting north and south during the summer and winter, respectively. At times, the STF's location may limit larval dispersal and reduce connectivity between the northern islands and Gough, thus creating two similar but isolated regions. To ensure a comprehensive protection regime, **we recommend the creation of strongly protected marine reserves in both the northern and southern portions of the EEZ/MPA.**
2. Seamounts are globally recognized as biodiversity hotspots, concentrating marine predators, marine mammals and birds and there are a number of seamounts in Tristan's EEZ (Figure 1). **We recommend the highest level of protection practicable for as many seamounts as possible throughout the EEZ, again making sure to include both northern and southern regions.** Presently there is one permit for trawling for the bluefish on the seamounts in the Southeastern portion of the EEZ.



The nearshore waters of Tristan's Islands are healthy and sustain a well-managed lobster fishery with some take of finfish by local islanders for consumption and bait.

3. The nearshore waters are already protected from foreign fishing activities extending from shore to 50nm. **This protection should be credited when designing additional protection schemes for Tristan and should remain permanently closed to foreign non-lobster vessels.**

4. The lobster fishery is well managed and has maintained MSC certification. It is also a huge component of the livelihoods and culture of the community. **We recommend that no additional spatial closures for lobster are needed as long as the fishery remains well managed. Monitoring and research on the fishery should continue.**

5. Our findings show that nearshore habitat structure and biological communities are remarkably similar around the islands (in the depths we surveyed). **If nearshore spatial closures for finfish and other non-lobster resources are considered, they could be located in most locations around the islands and achieve similar protection of the habitat.** However, population connectivity and source/sink dynamics are currently unknown and should be taken into consideration; other considerations might include adjacent seabird nesting and pinniped breeding sites or ease of human access.

6. Due to difficulties with weather and access, Tristan Islands currently have a large amount of de facto protection, both temporally and spatially. If conditions were to change (i.e. vessel capabilities or weather improvements due to climate effects) then additional protection measures should be considered. **We recommend a process be developed for regular assessment of access to fishing sites from Tristan islanders and non-locals.**

Future impacts from climate change and human pressure are increasing globally and might pose unanticipated challenges for Tristan's ecosystems and fishery.

7. The giant kelp, *M. pyrifera* foundation species is the key habitat-forming species and provides energetic resources to support the food web and lobster fisheries at Tristan da Cunha. This species is near its northern range limit and thermal tolerance at the northern islands in summertime. Future ocean warming and associated declines in productivity may affect nearshore ecosystems and fisheries. **We recommend continued monitoring of temperature (in situ and remote sensing) combined with predictive modeling of climate change.**

8. Our surveys found low levels of biological diversity in nearshore areas, resulting in a simple, short-chain food web with very few key interacting species. This makes this system very sensitive to perturbations (both acute, e.g. oil spills and chronic, e.g. climate change). Loss of one species could have very large impacts on the nearshore ecosystem because of lack of functional redundancy. **We recommend future studies be conducted on predicting climate change effects for this region (from global models) at least for populations of the key species (giant kelp, lobsters, urchins), and major drivers (productivity, sea and land temperature, changes in storm frequency).**

9. Invasive species have already been documented in Tristan's waters. Our research found that the non-indigenous silver porgy has expanded its presence beyond Tristan Island to Inaccessible Island but was in low densities at most locations. **Given the life history and reproductive success of marine fishes, and the low density and dietary habits (herbivory) of this species, we recommend that the populations continue to be tracked but we would not prioritize active removal efforts at this time.**

10. Redistribution of species is one predicted effect of climate change. This can have impacts on fisheries (reductions in existing fisheries but increases in new fisheries). For example, yellowtail (*Seriola lalandi*) have apparently increased recently in the northern islands, and islanders report concurrent declines in snoek (*Thyrsites atun*). **We recommend ongoing monitoring of species distributions and careful attention to the potential for development of new fisheries.**

DETAILED METHODS

DETAILED **METHODS**

Landsat Kelp Imagery

Landsat satellite imagery was used to create maps of predicted kelp (*M. pyrifera*) canopy presence (Cavanaugh et al. 2011, Bell et al. 2015). Kelp canopy was observed from Landsat 7 or Landsat 8 images (approximately 35 images used for Tristan da Cunha Islands). Those layers were augmented with predicted kelp occurrence based on other high resolution imagery or features seen in Landsat images (Bell, personal communication).

Nearshore Kelp Forest SCUBA Survey Techniques

To characterize the nearshore marine environment, we employed visual SCUBA surveys based on a modification of sampling methods developed by the Partnership for Interdisciplinary Studies of Coastal Oceans (PISCO; www.piscoweb.org). PISCO survey methods have been widely used in rocky reef and kelp forest monitoring throughout the world (e.g., [Hamilton et al. 2010b, Caselle et al. 2015a]), allowing for direct comparisons across locations. We modified the PISCO protocol to focus on fishes, lobsters, macroinvertebrates, kelps, and habitat characteristics of nearshore reefs. Sampling sites on each island were selected with the help of Landsat kelp canopy data to identify where kelp forests (and thus rocky habitat) have historically existed at each island (Figure 2). Sites were allocated around each island in a stratified uniform design, with the spacing between sites dictated by island size (sites were spread further apart on bigger islands) the presence of appropriate rocky habitat, and expedition logistics. On some islands, inclement wind and swell conditions limited the spatial survey effort to leeward shores.

At each survey site, divers quantified macroalgae, macroinvertebrates, and fishes in two depth strata, positioned at 10 and 20m depths. Each depth strata per site consisted of n = 2 belt transects that were 30m in length (n = 4 transects per site). While laying out the tape, one diver counted and estimated the sizes of all fishes to the nearest 1cm along a 30 x 2m transect. The transect extended to the surface or as far as visibility allowed, to include species associated with the kelp canopy and water column. At the end of each fish transect, the diver returned along the transect line, counting the number of giant kelp (*Macrocystis pyrifera*) plants and the number of stipes per plant, in addition to the number of pale kelp (*Laminaria pallida*) individuals within the 30 x 2m transect area. Size thresholds were used to exclude juveniles of both species (*M. pyrifera* was counted if the stipe was > 1m in length, while *L. pallida* when the stipe length was > 10cm). Finally, the fish diver characterized the substrate type and vertical relief at each meter along the transect line using uniform point contact methodology at 30 points. The substrate was classified as bedrock, boulder (10cm–1m in the longest diameter), cobble (< 10cm), or sand, while vertical relief was classified as the largest vertical distance between the highest and lowest points in a 0.5 x 1m box surrounding the point. Relief categories were defined as: 0–10cm, 10–100cm, 1–2m, and > 2m. A second diver proceeded to count and estimate the size of all Tristan rock lobster (*Jasus tristani*) to the nearest 1cm (carapace length) within a 30 x 8m transect along the same line. At the end of the lobster transect, that diver swam back along the tape counting other conspicuous macroinvertebrates (sea urchins, sea stars, etc.) in a 30 x 2m swath.

Pelagic and Deep Drop Camera Survey Techniques

National Geographic's Remote Imaging Team has developed Deep Ocean Dropcams, which are high definition cameras (Sony Handycam HDR-XR520V 12 megapixel) encased in a borosilicate glass sphere that are rated to 10,000m depth. Viewing area per frame is between 2–6m^2, depending on the steepness of the slope where the Dropcam lands. Cameras were baited with frozen fish and deployed for 2–4 hours. Lighting at depth is achieved through a high intensity LED array directed using external reflectors. Depth gauging is accomplished using an external pressure sensor. The Dropcams are weighted with a 22kg external weight with a descent rate of 1.5m s^{-1}. The primary release mechanism is a burn wire that is activated using onboard battery voltage. The Dropcams are positively buoyant resulting in an ascent rate of 0.5m s^{-1}. Dropcams have an onboard VHF transmitter that allows for recovery using locating antennae with backup location achieved via communication with the ARGOS satellite system.

Deep-water drop cameras were deployed at sites on each side of every island (Figure 3), when conditions allowed and in a variety of depths on each island (Table 7). The cameras were set to record with lights in the following sequence: 40 minutes on, 30 minutes off and 30 minutes back on (10 minutes to drop) Similar to the nearshore drop camera, relative abundance of each species was estimated as the maximum number of individuals observed in any given video frame (N_{max}). This was summed to total numerical abundance as a minimum estimate of number of individuals.

Mid-water baited remote underwater video stations (BRUVS) are designed to quantify pelagic fish assemblage characteristics. These pelagic BRUVS consist of a cross bar with two GoPro cameras fixed 0.8m apart on an inward convergent angle of 8°. In longline formation, three units are deployed concurrently and separated by 200m. Rigs were baited with 1kg of crushed fish and were deployed 2–4km offshore of each island, with sampling sites stratified by island group and shore (Figure 3). Mid-water BRUVS were deployed for a minimum of two hours at each sampling site and video analysis was standardised to two hours. The stereo configuration allows for measurement of individual fish sizes. Relative abundance of each species per deployment was estimated as the maximum number of individuals observed in any given video frame (N_{max}). Sizes of each individual were calculated using EventMeasure software.

Data Analysis for SCUBA, Deep Sea and Pelagic Camera Surveys

To estimate fish and lobster biomass, counts were tallied by length class and individual-specific lengths converted to body weights using the allometric length-weight conversion: $W = a L^b$, where parameters a and b are species-specific constants, L is total length for fishes and carapace length for lobsters in cm, and W is weight in grams. Length-weight fitting parameters were obtained from FishBase (Froese and Pauly 2017) for fishes and from the Tristan Fisheries Department for lobsters (Glass 2014) with the product of individual weights and numerical densities used to estimate biomass by species. Numerical density (abundance) was expressed as number of individuals per $100m^2$ and biomass was expressed as tonnes (t) per hectare (ha). We used analysis of variance (ANOVA) to test for spatial differences in abundance and biomass of fishes, macroinvertebrates, and kelps, and percent cover of substrate characteristics at the island scale.

To examine multivariate differences in the composition of fish and benthic assemblages among islands, we used the site-level mean densities for each species. Nonmetric multidimensional scaling (nMDS) analyses were used to visualize spatial differences in fish and benthic communities among islands. Species abundances were square root transformed prior to analysis and a Bray-Curtis similarity matrix was used. PERMANOVA tests with the single factor of Island were used to statistically test for differences in fish and benthic communities among islands. Multivariate analyses were conducted in PRIMER v.6.

Similar analyses were used to test for island-level differences in relative species abundance for the response variables extracted from drop camera surveys.

Shark Tagging Methods

We focused on Blue sharks as they were more frequently encountered on the pelagic cameras and caught in our shark fishing trips. However, shortfin mako sharks were also tagged following the expedition. Sharks were caught with hook and line, tail roped and restrained alongside the vessel where they were inverted and induced into tonic immobility. Once cataleptic, sharks were measured, sexed and tagged externally with satellite tags. We used miniPAT tags (Wildlife Computers) affixed externally in the dorsal musculature. The MiniPAT is a pop-up archival transmitting tag (PAT tag, also known as a PSAT). It is a sophisticated combination of archival and Argos satellite technology. PAT tags are designed to track the large-scale movements and behavior of fish and other animals that do not spend enough time at the surface to allow the use of real-time Argos satellite tags. Sensor data (vertical depth movement and environmental temperature) are collected during deployment and archived in onboard memory. Our tags were set to release 180 days post deployment.

Pinniped Methods

PUP COUNTS

Towards the end of January, i.e. after the last pups are born and before they start playing in the shallows, particular stretches of coastlines (variously on all four islands — see Results) were searched on foot, clambering among the rocks and boulders to find the pups. All live pups were counted, and the exact location(s) of each breeding colony (or not) recorded. Given the time at our disposal, prevailing weather, primarily wind strength, direction and sea state, and physiognomy of the beaches, not all beaches could be accessed from land (steep cliffs backing the beaches) or from the sea (poor landing conditions), and counts from close inshore from a RIB were done on occasion. Only live pups were recorded, but constraints of time, high density of pups at breeding colony beaches, and manpower availability, precluded repeat counts to calculate errors of the pup number estimates for each beach. Naturally, an unknown number of pups escaped being counted (hidden under rocks and in large concentrations), and a percentage undercount, using known undercounts for the different beach types, such as open boulder beach versus jumbled rocky beach, will be used to adjust counts (Condy 1978, Bester 1980, Kerley 1983, Wilkinson and Bester 1990, Hofmeyr et al. 1997, Wege et al. 2016a). An estimate of the number of dead pups (which together with the undercount corrected number of live pups, provides estimated number of births) will follow application of published mortality rates, over the first six weeks of life, for the species (Bester 1978). Comparison of pup counts in the present study, will be compared primarily with those recorded previously (Bester 1990, Bester et al. 2006) to calculate rates of population trend, using pup numbers as a proxy of population size.

SCAT SAMPLING

Scats were collected at each beach landing opportunity, taking care to label date and site of collections correctly, keeping each stool separate. Cleaning and sorting of samples will be done back in the laboratory in South Africa. Essentially, scat samples will be individually washed through a 0.5mm sieve under running water to collect the prey hard parts. The fish otoliths and cephalopod beaks from each scat are then stored separately in individual plastic packets and 70% ethanol in vials respectively for further identification following Klages and Bester (1998) and Makhado et al. (2008). Fish species will be identified from sagittal otoliths using a Zeiss microscope with the magnification setting Axioskop 10 x 1.6. Otolith lengths will be measured to the nearest 0.01mm using Axiovision V 4.8.3.0 (2006–2012 Carl Zeiss Microscopy GmBH). A visual database of photographed otoliths from Axiovision V 4.8.3.0, including its measurements, already created in a previous study, will allow prey species identification down to the lowest possible taxonomic rank. Fish are identified from the prey hard parts using a reference manual (Smale et al. 1995), as well as the reference collection of the Port Elizabeth Museum at Bayworld, Port Elizabeth, South Africa. Using regression equations, the standard length and mass of prey will be estimated from fish sagittal otolith length measurements (Clarke 1986, Williams et al. 1990, Makhado et al. 2008). Squid lower beaks will be used for identification and quantification of cephalopod prey using Clarke (1986) and Smale et al. (1993). Examining scats will also inform on the ingestion of microplastics by the prey of the fur seals (Ryan 2016).

PUP WEIGHING

Subantarctic fur seal pups (50 male and 50 female) were weighed, broadly following Convention for the Conservation of Antarctic Marine Living Resources procedure at particular sites (Tumbledown Beach at Gough, and The Caves at Tristan da Cunha) in support of a long-term study (Oosthuizen et al. 2016). The mean age of weighed pups was estimated from the calculated median birthdate (10/11 December – Bester 1987). Pups were weighed with a 25kg Rapala Electronic Scale. Pups were temporarily marked with spray paint to prevent repeat weighing of individuals, and had to appear healthy, whether thin or fat. Dead and obviously dying and emaciated pups were excluded from weighing. Every third pup encountered, or all the pups in herded groups, were weighed to remove possible bias in selecting pups (Kirkman et al. 2002).

LACTATING FUR SEAL FORAGING AREAS

At-sea movements were recorded through satellite tracking of foraging females (TdC = 4, Gough = 6). Presumed healthy individuals were randomly selected, captured with a fur seal hoop net, weighed using a 100kg Clover scale, and a platform transmitter terminal (PTT), SPOT-311A (Wildlife Computers, 38g, 51 x 27 x 19mm) linked to the ARGOS Collection and Location System (Argos 1996) was attached to the fur on the dorsal midline of each seal, immediately posterior to the scapulae, using a double component Araldite® epoxy resin. At each of the deployments, the restrained seals were released after 30 minutes. Once the batteries fail (in May 2017, after about four months), data analysis can commence. Spatial analysis will follow Arthur et al. (2016), Wege et al. (2016b), and Wege (2017). Location information relayed through the global ARGOS satellite system contains inherent errors and some erroneous data. To account for this observation error we will fit a two-state, behaviourally switching, state-space model to Argos tracks (Jonsen et al. 2005, Jonsen 2016). State-space models will filter out erroneous location estimates and provided interpolated tracks with locations at set time intervals. State-space models also estimate the behavioural mode of each location point, which identifies areas of restricted search (ARS), likely to be foraging locations, and distinguishes these from travelling areas (Bestley et al. 2016). In other words, a location point is identified where searching/foraging behaviour (hereafter referred to as ARS) or travelling behaviour is engaged in. These location estimates will then be linked to remotely sensed environmental variables such as: bathymetry (GEBCO Digital Atlas, IOC 2003), monthly sea-surface temperature (MODIS, (Feldman and McClain 2008), seasonal chlorophyll-a (SeaWiFS, (Feldman and McClain 2009) and daily sea-surface height anomaly (SSHA)/altimetry data. We will test for intra-seasonal differences in mean values of environmental variable values interpolated to the location data based on transit type (ARS vs. travelling) and direction travelled. Inter-seasonal variation (summer vs. autumn) in environmental features targeted by females will also be compared using appropriate statistical analyses.

INDIRECT PREY IDENTIFICATION

Whisker and hair samples, and on occasion, blood samples from superficial hindflipper veins, were collected (syringe & hypodermic needle) from the restrained lactating females when receiving a tracking device. The vibrissae of fur seals grow continuously at a constant rate, are retained for multiple years, and the biogeochemical data (stable isotope signatures) captured along the length of vibrissae will be utilized to study the diet (Kernaléguen et al. 2012). Hair, on the other hand, are retained for shorter periods (replaced during the moult), while blood has a much higher metabolic turnover rate. As each sampled tissue has a tissue-specific isotopic turnover rate, or unique period of biomolecule deposition (Tieszen et al. 1983), it enables the reconstruction of the trophic ecology of the fur seals integrated over days (e.g. plasma) to weeks (blood), months (hair), or over a much longer period of time (whiskers). Samples will be processed for stable isotope content at the Stable Isotope Laboratory of the MRI, Department of Zoology & Entomology, University of Pretoria, using standard procedures (Lübcker 2015).

Invasive Species Research – Methods

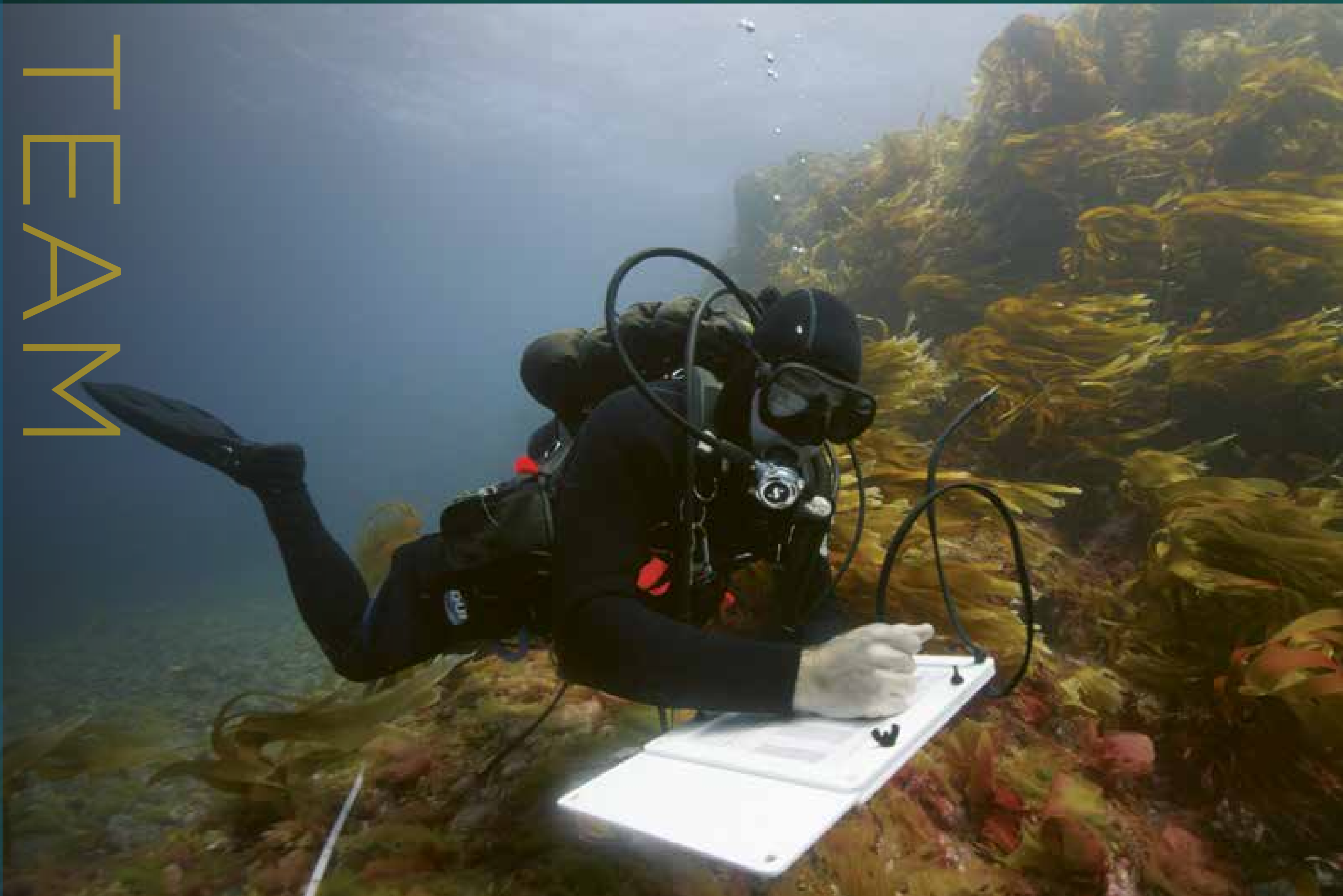
We collected n = 27 *Diplodus argenteus* by spear being careful to spear each individual as soon as it was observed. We weighed each fish (to the 0.1 gram) on a portable table top balance, and measured SL and TL. We also weighed the intestinal fat and visually inspected the gonads to identify the sex. Tissue samples (for stable isotope analysis) were taken and kept frozen. Fin clips (for genetics) were taken and stored in ethanol. Otoliths were stored dry. A sample of gonads were also preserved in ethanol for later inspection in the lab. Guts were stored initially in formalin and transferred to ethanol.

Gut contents were sorted and identified to the lowest taxonomic ranking. We did not estimate percent volume by weight as the field dissection methods were imprecise and many samples had lost parts of stomach and intestine due to spearing.

Oceanographic Research

A RBR concerto CTD coupled with JFE Advantek O2 sensor was attached to the deep-water drop cameras to measure ocean characteristics with a particular interest in spatial differences between the northern group of island and Gough. A total of 11 drops were made and all islands were sampled (Gough n = 3; Inaccessible n = 1; Nightingale n = 2; Tristan da Cunha n = 5). Data were downloaded directly from the instrument and processed using the RBR software.

EXPEDITION TEAM



EXPEDITION TEAM

List of Expedition Participants

Name	Role	Institution
Paul Rose	Expedition Leader	National Geographic & the Royal Geographical Society
Dr. Jennifer Caselle	Chief Scientist	Marine Science Institute, University of California, Santa Barbara
Katie Davis	Science diving, nearshore cameras	Marine Science Institute, University of California, Santa Barbara
Dr. Scott Hamilton	Science diving, invasive species	Moss Landing Marine Laboratories, California
Dr. Ryan Jenkinson	Science diving, lobsters	Humboldt State University
Doug Simpson	Science diving, outreach	University of Montana
Dr. Marthán Bester	Marine Mammals	Mammal Research Institute University of Pretoria
Dr. Mia Wege	Marine Mammals	Mammal Research Institute University of Pretoria
Dr. Mike Fay	Terrestrial science	National Geographic
Alan Turchik	Deep-water drop cameras	National Geographic
Chris Thompson	Pelagic cameras	Centre for Marine Futures, University of Western Australia
Dave McAloney	Diving / Medical / Logistics	National Geographic
Sam Dews	Boat driver/logistics & science support	National Geographic
Alexandra Verville	Producer	National Geographic
Brian Canavan	Topside filming	National Geographic
Roger Horrocks	Underwater filming	National Geographic
Nathan Lefevre	Underwater camera assistant	National Geographic
Dan Myers	International policy	National Geographic
Jonathan Hall	Political liaison and birds	Head of UK Overseas Territories; Royal Society for the Protection of Birds
Andy Schofield	Birds, terrestrial, local guidance and liaison	UK Overseas Territories Officer; Royal Society for the Protection of Birds.
James Glass	Fisheries, local guidance	Director of Fisheries, Tristan da Cunha
Trevor Glass	Birds and terrestrial conservation, local guidance	Director of Conservation, Tristan da Cunha
Rodney Green	Coxswain/diver/SFO	Assistant Director of Fisheries, Tristan da Cunha
Julian Repetto	Coxswain/guide	Assistant Conservation Department, Tristan da Cunha
George Swain	Coxswain/guide	Assistant Conservation Department, Tristan da Cunha
Katrine Herian	Terrestrial guide	Conservation policy, Tristan da Cunha
Ian Lavarello	Chief Islander	Public Works Department

REFERENCES

Anderson, O. R. J., C. J. Small, J. P. Croxall, E. K. Dunn, B. J. Sullivan, O. Yates, and A. Black. 2011. Global seabird bycatch in longline fisheries. Endangered Species Research 14:91–106.

Andrew, T. 1992. Feeding biology of Acantholatris monodactylus (Pisces: Cheilodactylidae) at Tristan da Cunha and Gough Island, South Atlantic. S Afr J Antarct Res 22:41–49.

Andrew, T., T. Hecht, P. Heemstra, and J. Lutjeharms. 1995. Fishes of the Tristan da Cunha group and Gough Island, South Atlantic Ocean. Ichthyological Bulletin of the JLB Smith Institute of Ichthyology 63:1–43.

Andrew, T., and P. Ryan. 2007. Field guide to the animals and plants of Tristan da Cunha and Gough Island. Pisces Publications.

Andrew, T. G., and T. Hecht. 1996. Age and growth of the soldier, Sebastes capensis (Pisces: Scorpaenidae) at Tristan da Cunha and Gough Island, South Atlantic Ocean. Journal of Zoology 238:125–135.

Angel, A., and J. Cooper. 2006. A Review of the Impacts of Introduced Rodents on the Islands of Tristan da Cunha and Gough. Royal Society for the Protection of Birds Sandy.

Arnould, J. P. Y., S. P. Luque, C. Guinet, D. P. Costa, J. Kingston, and S. A. Shaffer. 2003. The comparative energetics and growth strategies of sympatric Antarctic and subantarctic fur seal pups at Îles Crozet. Journal of Experimental Biology 206:4497–4506.

Arthur, B., M. Hindell, M. N. Bester, P. J. N. de Bruyn, P. Trathan, M. Goebel, and M. A. Lea. 2016. Winter habitat predictions of a key Southern Ocean predator, the Antarctic fur seal (*Arctocephalus gazella*). Deep Sea Research Part II: Topical Studies in Oceanography.

Baker, P. E., I. G. Gass, P. G. Harris, and R. W. le Maitre. 1962. The volcanological report of the Royal Society expedition to Tristan da Cunha, 1962. . Philosophical Transactions of the Royal Society of London. Series A, Mathematical and Physical Sciences. 256:447–576.

Bakun, A. 2006. Fronts and eddies as key structures in the habitat of marine fish larvae: opportunity, adaptive response and competitive advantage. Scientia Marina 70:105–122.

Barrientos, C. A., M. T. Gonzalez, and C. A. Moreno. 2006. Geographical differences in the feeding patterns of red rockfish (*Sebastes capensis*) along South American coasts. Fishery Bulletin 104:489–497.

Baxter, J. L. 1960. A study of the yellowtail Seriola dorsalis (Gill). State of California Department of Fish and Game Marine Resources Operations Fish Bulletin No. 110:1–91.

Beauplet, G., L. Dubroca, C. Guinet, Y. Cherel, W. Dabin, C. Gagne, and M. A. Hindell. 2004. Foraging ecology of subantarctic fur seals *Arctocephalus tropicalis* breeding on Amsterdam Island: seasonal changes in relation to maternal characteristics and pup growth. Marine Ecology Progress Series 273:211–225.

Bell, T. W., K. C. Cavanaugh, D. C. Reed, and D. A. Siegel. 2015. Geographical variability in the controls of giant kelp biomass dynamics. Journal of Biogeography 42:2010–2021.

Best, P., J. Glass, P. Ryan, and M. Dalebout. 2009. Cetacean records from Tristan da Cunha, South Atlantic. Journal of the Marine Biological Association of the United Kingdom 89:1023–1032.

Bester, M. N. 1978. Gougheiland Pinnipedia Navorsing. Vorderings-verslag Julie 1978–Desember 1978. Research Report.

Bester, M. N. 1980. Population increase in the Amsterdam fur seal *Arctocephalus tropicalis* at Gough Island. South African Journal of Zoology 15:229–234.

Bester, M. N. 1987. The Subantarctic fur seal *Arctocephalus tropicalis* at Gough Island (Tristan da Cunha Group). Proceedings of the Fur Seal Workshop April 1984. NOAA Technical Reports NMFS 51:57–60.

Bester, M. N. 1990. Population trends of Subantarctic fur seals and southern elephant seals at Gough Island. South African Journal of Antarctic Research 20:9–12.

Bester, M. N., P. J. N. de Bruyn, W. C. Oosthuizen, C. A. Tosh, T. McIntyre, R. R. Reisinger, M. Postma, D. S. Van Der Merwe, and M. Wege. 2011. The Marine Mammal Programme at the Prince Edward Islands: 38 years of research. African Journal of Marine Science 33:511–521.

Bester, M. N., and P. G. Ryan. 2007. Mammals. Pages 99–108 in P. G. Ryan, editor. Field guide to the animals and plants of Tristan da Cunha and Gough Island. Pisces Publications, Newbury.

Bester, M. N., J. W. Wilson, M. H. Burle, and G. J. G. Hofmeyr. 2006 Population trend of Subantarctic fur seals at Gough Island. South African Journal of Wildlife Research 36:191–194.

Bestley, S., I. D. Jonsen, R. G. Harcourt, M. A. Hindell, and N. J. Gales. 2016. Putting the behavior into animal movement modeling: Improved activity budgets from use of ancillary tag information. Ecology and Evolution:1–13.

Block, B. A., I. D. Jonsen, S. J. Jorgensen, A. J. Winship, S. A. Shaffer, S. J. Bograd, E. L. Hazen, D. G. Foley, G. A. Breed, A. L. Harrison, J. E. Ganong, A. Swithenbank, M. Castleton, H. Dewar, B. R. Mate, G. L. Shillinger, K. M. Schaefer, S. R. Benson, M. J. Weise, R. W. Henry, and D. P. Costa. 2011. Tracking apex marine predator movements in a dynamic ocean. Nature 475:86–90.

Booth, J. D., A. D. Carruthers, C. D. Bolt, and R. A. Stewart. 1991. Measuring depth of settlement in the red rock lobster, *Jasus edwardsii*. New Zealand Journal of Marine and Freshwater Research 25:123–132.

Borrvall, C., B. Ebenman, and T. J. Tomas Jonsson. 2000. Biodiversity lessens the risk of cascading extinction in model food webs. Ecology Letters 3:131–136.

Bost, C. A., C. Cotté, F. Bailleul, Y. Cherel, J. B. Charrassin, C. Guinet, D. G. Ainley, and H. Weimerskirch. 2009. The importance of oceanographic fronts to marine birds and mammals of the southern oceans. Journal of Marine Systems 78:363–376.

Bowen, W. D., and S. J. Iverson. 2012. Methods of estimating marine mammal diets: A review of validation experiments and sources of bias and uncertainty. Marine Mammal Science 29:719–754.

Boyd, I. L., J. P. Y. Arnould, T. Barton, and J. P. Croxall. 1994. Foraging behaviour of Antarctic fur seals during periods of contrasting prey abundance. Journal of Animal Ecology 63:703–713.

Boyd, I. L., D. J. McCafferty, and T. R. Walker. 1997. Variation in foraging effort by lactating Antarctic fur seals: Response to simulated increased foraging costs. Behavioral Ecology and Sociobiology 40:135–144.

Boyd, I. L., I. J. Staniland, and A. R. Martin. 2002. Distribution of foraging by female Antarctic fur seals. Marine Ecology Progress Series 242:285–294.

Bradley, D., E. Conklin, Y. P. Papastamatiou, D. J. McCauley, K. Pollock, A. Pollock, B. E. Kendall, S. D. Gaines, and J. E. Caselle. 2017a. Resetting predator baselines in coral reef ecosystems. Scientific Reports 7:43131.

Bradley, D., Y. P. Papastamatiou, and J. E. Caselle. 2017b. No persistent behavioural effects of SCUBA diving on reef sharks. Marine Ecology Progress Series 567:173–184.

Bradshaw, C. J. A., M. A. Hindell, M. D. Sumner, and K. J. Michael. 2004. Loyalty pays: potential life history consequences of fidelity to marine foraging regions by southern elephant seals. Animal Behaviour 68:1349–1360.

Brothers, N. 1991. Albatross mortality and associated bait loss in the Japanese longline fishery in the Southern Ocean. Biological Conservation 55:255–268.

Brown, D. 2007. A feasibility study for the eradication of rodents from Tristan da Cunha. Unpublished Report, Royal Society for the Protection of Birds, Sandy, United Kingdom.

Campana, S. E., A. Dorey, M. Fowler, W. Joyce, Z. Wang, D. Wright, and I. Yashayaev. 2011. Migration pathways, behavioural thermoregulation and overwintering grounds of blue sharks in the Northwest Atlantic. PLoS One 6:e16854.

Carey, F. G., J. V. Scharold, and A. J. Kalmijn. 1990. Movements of blue sharks (*Prionace glauca*) in depth and course. Marine Biology 106:329–342.

Caselle, J. E., A. Rassweiler, S. L. Hamilton, and R. R. Warner. 2015. Recovery trajectories of kelp forest animals are rapid yet spatially variable across a network of temperate marine protected areas. Scientific Reports 5.

Casper, R. M., N. J. Gales, M. A. Hindell, and S. M. Robinson. 2006. Diet estimation based on an integrated mixed prey feeding experiment using *Arctocephalus* seals. Journal of Experimental Marine Biology and Ecology 328:228–239.

Cavanaugh, K. C., D. A. Siegel, D. C. Reed, and P. E. Dennison. 2011. Environmental controls of giant-kelp biomass in the Santa Barbara Channel, California. Marine Ecology Progress Series 429:1–17.

Cherel, Y., C. Fontaine, P. Richard, and J. P. Labat. 2010. Isotopic niches and trophic levels of myctophid fishes and their predators in the Southern Ocean. Limnology and Oceanography 55:324–332.

Cherel, Y., L. Kernaléguen, P. Richard, and C. Guinet. 2009. Whisker isotopic signature depicts migration patterns and multi-year intra- and inter-individual foraging strategies in fur seals. Biology Letters 5:830–832.

Clark, M. R., and J. A. Koslow. 2007. Impacts of fisheries on seamounts. Seamounts: ecology, fisheries, and conservation 12:413–441.

Clarke, M. R. 1986. A handbook for the identification of cephalopods beaks. Clarendon Press.

Collette, B., S. Chang, A. Di Natale, W. Fox, M. J. Jorda, N. Miyabe, R. Nelson, Y. Uozumi, and S. Wang. 2011. *Thunnus maccoyii*. The IUCN Red List of Threatened Species, Version 2013.2.

Condy, P. R. 1978. Distribution, abundance, and annual cycle of fur seals (*Arctocephalus* spp.) on the Prince Edward Islands. South African Journal of Wildlife Research 8:159–168.

Cuthbert, R., G. Hilton, P. Ryan, and G. N. Tuck. 2005. At-sea distribution of breeding Tristan albatrosses *Diomedea dabbenena* and potential interactions with pelagic longline fishing in the South Atlantic Ocean. Biological Conservation 121:345–355.

da Silva, C., S. E. Kerwath, C. G. Wilke, M. Meÿer, and S. J. Lamberth. 2010. First documented southern transatlantic migration of a blue shark *Prionace glauca* tagged off South Africa. African Journal of Marine Science 32:639–642.

David, G. S., R. Coutinho, I. Quagio-Grassiotto, and J. R. Verani. 2005. The reproductive biology of *Diplodus argenteus* (Sparidae) in the coastal upwelling system of Cabo Frio, Rio de Janeiro, Brazil. African Journal of Marine Science 27:439–447.

Deacon, G. E. R. 1982. Physical and biological zonation in the Southern Ocean. Deep-Sea Research 29:1–15.

Doherty, T. S., A. S. Glen, D. G. Nimmo, E. G. Ritchie, and C. R. Dickman. 2016. Invasive predators and global biodiversity loss. Proceedings of the National Academy of Sciences 113:11261–11265.

Doolittle, D. F., W. K. W. Li, and A. M. Wood. 2008. Wintertime abundance of picoplankton in the Atlantic sector of the Southern Ocean. Nova Hedwigia, Beiheft 133:147–160.

Dubiaski-Silva, J., and S. Masunari. 2004. Ontogenetic and Seasonal Variation in the Diet of Marimbá, *Diplodus argenteus* (Valenciennes, 1830) (Pisces, Sparidae) Associated With the Beds of Sargassum Cymosum C.Agardh, 1820 (Phaeophyta) at Ponta das Garoupas, Bombinhas, Santa Catarina. Journal of Coastal Research II:1190–1192.

Dubiaski-Silva, J., and S. Masunari. 2006. Ontogenetic and seasonal variation in the diet of marimbá, *Diplodus argenteus* (Valenciennes, 1830)(Pisces, Sparidae) associated with the beds of Sargassum cymosum (C. Agardh, 1820) (Phaeophyta) at Ponta das Garoupas, Bombinhas, Santa Catarina. Journal of Coastal Research:1190–1192.

Dunne, J. A., R. J. Williams, and N. D. Martinez. 2002. Network structure and biodiversity loss in food webs: robustness increases with connectance. Ecology Letters 5:558–567.

Ebert, D. A. 1991. Diet of the seven gill shark *Notorynchus cepedianus* in the temperate coastal waters of southern Africa. South African Journal of Marine Science 11:565–572.

Ebert, D. A. 2002. Ontogenetic changes in the diet of the sevengill shark (*Notorynchus cepedianus*). Marine and Freshwater Research 53:517–523.

EklÖF, A., and B. O. Ebenman. 2006. Species loss and secondary extinctions in simple and complex model communities. Journal of Animal Ecology 75:239–246.

Etnoyer, P., D. Canny, B. Mate, and L. Morgan. 2004. Persistent pelagic habitats in the Baja California to Bering Sea (B2B) ecoregion. Oceanography-Washington DC-Oceanography Society- 17:90–101.

Feldman, G. C., and C. R. McClain. 2008. Ocean color web, Aqua-MODIS reprocessing 1.1. Greenbelt (MD): NASA Goddard Space Flight Center [accessed 2005 October 10].

Feldman, G. C., and C. R. McClain. 2009. Ocean color web. SeaWiFS Reprocessing 5.

Francis, M. P., L. H. Griggs, and S. J. Baird. 2001. Pelagic shark bycatch in the New Zealand tuna longline fishery. Marine and Freshwater Research 52:165–178.

Friedlander, A. M., E. Ballesteros, J. Beets, E. Berkenpas, C. F. Gaymer, M. Gorny, and E. Sala. 2013. Effects of isolation and fishing on the marine ecosystems of Easter Island and Salas y Gómez, Chile. Aquatic Conservation: Marine and Freshwater Ecosystems 23:515–531.

Friedlander, A. M., J. E. Caselle, E. Ballesteros, E. K. Brown, A. Turchik, and E. Sala. 2014. The Real Bounty: Marine Biodiversity in the Pitcairn Islands. PLoS One 9:e100142.

Froese, R., and D. Pauly. 2017. FishBase. World Wide Web electronic publication. www.fishbase.org.

Gallagher, A., E. Orbesen, N. Hammerschlag, and J. Serafy. 2014. Vulnerability of oceanic sharks as pelagic longline bycatch. Global Ecology and Conservation 1:50–59.

Georges, J. Y., and C. Guinet. 2000a. Maternal care in the subantarctic fur seals on Amsterdam Island. Ecology 81:295–308.

Georges, J. Y., and C. Guinet. 2000b. Early mortality and perinatal growth in the subantarctic fur seal (*Arctocephalus tropicalis*) on Amsterdam Island. Journal of Zoology 251:277–287.

Glass, J. P. 2014. The fishery and biology of the rock lobster *Jasus tristani* at the Tristan da Cunha island group. MS Thesis. Cape Peninsula University of Technology.

Guinet, C., and J. Y. Georges. 2000. Growth in pups of the subantarctic fur seal (*Arctocephalus tropicalis*) on Amsterdam Island. Journal of Zoology 251:289–296.

Hamilton, S. L., J. E. Caselle, D. P. Malone, and M. H. Carr. 2010. Incorporating biogeography into evaluations of the Channel Islands marine reserve network. Proceedings of the National Academy of Sciences 107:18272–18277.

Hazin, F. H., P. B. Pinheiro, and M. K. Broadhurst. 2000. Further notes on reproduction of the blue shark, *Prionace glauca*, and a postulated migratory pattern in the South Atlantic Ocean. Ciência e Cultura 52:114–120.

Heydorn, A., and J. Lutjeharms. 1980. Tristan da Cunha… the loneliest isle. African Wildlife 34:12–17.

Heydorn, A. E. F. 1969. The South Atlantic Rock Lobster *Jasus tristani* at Vema Seamount, Gough Island and Tristan da Cunha. Division of Sea Fisheries.

Hofmeyr, G. J. G., M. N. Bester, and F. C. Jonker. 1997. Changes in population sizes and distribution of fur seals at Marion Island. Polar Biology 17:150–158.

Jackson, G. A. 1987. Modelling the growth and harvest yield of the giant kelp *Macrocystis pyrifera*. Marine Biology 95:611–624.

Johnson, C. R., S. C. Banks, N. S. Barrett, F. Cazassus, P. K. Dunstan, G. J. Edgar, S. D. Frusher, C. Gardner, M. Haddon, F. Helidoniotis, K. L. Hill, N. J. Holbrook, G. W. Hosie, P. R. Last, S. D. Ling, J. Melbourne-Thomas, K. Miller, G. T. Pecl, A. J. Richardson, K. R. Ridgway, S. R. Rintoul, D. A. Ritz, D. J. Ross, J. C. Sanderson, S. A. Shepherd, A. Slotwinski, K. M. Swadling, and N. Taw. 2011. Climate change cascades: Shifts in oceanography, species' ranges and subtidal marine community dynamics in eastern Tasmania. Journal of Experimental Marine Biology and Ecology 400:17–32.

Jones, H. P., B. R. Tershy, E. S. Zavaleta, D. A. Croll, B. S. Keitt, M. E. Finkelstein, and G. R. Howald. 2008. Severity of the effects of invasive rats on seabirds: a global review. Conservation Biology 22:16–26.

Jonsen, I. D. 2016. Joint estimation over multiple individuals improves behavioural state inference from animal movement data. Scientific Reports 6:20625–20625.

Jonsen, I. D., J. M. Flemming, and R. A. Myers. 2005. Robust state-space modeling of animal movement data. Ecology 86:2874–2880.

Jordan, F., I. Scheuring, and G. Vida. 2002. Species positions and extinction dynamics in simple food webs. Journal of Theoretical Biology 215:441–448.

Joung, S. J., G. T. Lyu, K. Y. Su, H. H. Hsu, and K. M. Liu. 2017. Distribution Pattern, Age, and Growth of Blue Sharks in the South Atlantic Ocean. Marine and Coastal Fisheries 9:38–49.

Kerley, G. I. H. 1983. Comparison of seasonal haul-out patterns of fur seals, *Arctocephalus tropicalis* and *Arctocephalus gazella* on subantarctic Marion Island. South African Journal of Wildlife Research 13:71–77.

Kernaléguen, L., B. Cazelles, J. P. Y. Arnould, P. Richard, C. Guinet, and Y. Cherel. 2012. Long-term species, sexual and individual variations in foraging strategies of fur seals revealed by stable isotopes in whiskers. PLoS One 7:e32916–e32916.

Kirkman, S. P., M. N. Bester, G. J. G. Hofmeyr, P. A. Pistorius, and A. B. Makhado. 2002. Pup growth and maternal attendance patterns in Subantarctic fur seals. African Zoology 37:13–19.

Kirkman, S. P., M. N. Bester, A. B. Makhado, and P. A. Pistorius. 2003. Female attendance patterns of Antarctic fur seals at Marion Island. African Zoology 38:402–405.

Klages, N. T. W., and M. N. Bester. 1998. Fish prey of fur seals *Arctocephalus* spp. at subantarctic Marion Island. Marine Biology 131:559–566.

Kohler, N. E., P. A. Turner, J. J. Hoey, L. J. Natanson, and R. Briggs. 2002. Tag and recapture data for three pelagic shark species: blue shark (*Prionace glauca*), shortfin mako (*Isurus oxyrinchus*), and porbeagle (*Lamna nasus*) in the North Atlantic Ocean. Collect. Vol. Sci. Pap. ICCAT 54:1231–1260.

Krumhansl, K. A., D. K. Okamoto, A. Rassweiler, M. Novak, J. J. Bolton, K. C. Cavanaugh, S. D. Connell, C. R. Johnson, B. Konar, and S. D. Ling. 2016. Global patterns of kelp forest change over the past half-century. Proceedings of the National Academy of Sciences 113:13785–13790.

Lafferty, K. D. 2004. Fishing for lobsters indirectly increases epidemics in sea urchins. Ecological Applications 14:1566–1573.

Laubscher, R. K., R. Perissinotto, and C. D. McQuaid. 1993. Phytoplankton production and biomass at frontal zones in the Atlantic sector of the Southern Ocean. Polar Biology 13:471–481.

Lea, M. A., C. Guinet, Y. Cherel, G. Duhamel, L. Dubroca, P. Pruvost, and M. A. Hindell. 2006. Impacts of climatic anomalies on provisioning strategies of a Southern Ocean predator. Marine Ecology Progress Series 310:77–94.

Ling, S., C. Johnson, S. Frusher, and K. Ridgway. 2009a. Overfishing reduces resilience of kelp beds to climate-driven catastrophic phase shift. Proceedings of the National Academy of Sciences 106:22341–22345.

Ling, S. D., C. R. Johnson, S. D. Frusher, and K. R. Ridgway. 2009b. Overfishing reduces resilience of kelp beds to climate-driven catastrophic phase shift. Proceedings of the National Academy of Sciences 106:22341–22345.

Llido, J. 2005. Event-scale blooms drive enhanced primary productivity at the Subtropical Convergence. Geophysical Research Letters 32.

Lübcker, N. 2015. Diet of underyearling southern elephant seals (Mirounga leonina) from Marion Island: a stable isotope investigation. University of Pretoria, South Africa., South Africa.

Lutjeharms, J. R. E., N. M. Walters, and B. R. Allanson. 1985. Oceanic frontal systems and biological enhancement. Pages 11–21 in W. R. Siegfried, P. R. Condy, and R. M. Laws, editors. Antarctic Nutrient Cycles and Food Webs. Springer, Berlin Heidelberg New York.

MacAlister Elliott and Partners Ltd. 2011. Final Report, Tristan da Cunha fishery for rock lobster *Jasus tristani*.

Makhado, A. B., M. N. Bester, S. P. Kirkman, P. A. Pistorius, J. W. H. Ferguson, and N. T. W. Klages. 2008. Prey of the Antarctic fur seal *Arctocephalus gazella* at Marion Island. Polar Biology 31:575–581.

Makhado, A. B., M. N. Bester, S. Somhlaba, and R. J. M. Crawford. 2013. The diet of the subantarctic fur seal *Arctocephalus tropicalis* at Marion Island. Polar Biology 36:1609–1617.

McCafferty, D. J., I. L. Boyd, T. R. Walker, and R. I. Taylor. 1998. Foraging responses of Antarctic fur seals to changes in the marine environment. Marine Ecology Progress Series 166:285–299.

Montealegre-Quijano, S., and C. M. Vooren. 2010. Distribution and abundance of the life stages of the blue shark *Prionace glauca* in the Southwest Atlantic. Fisheries Research 101:168–179.

Morato, T., W. W. L. Cheung, and T. J. Pitcher. 2006. Vulnerability of seamount fish to fishing: fuzzy analysis of life-history attributes. Journal of Fish Biology 68:209–221.

Morato, T., S. D. Hoyle, V. Allain, and S. J. Nicol. 2010. Seamounts are hotspots of pelagic biodiversity in the open ocean. Proceedings of the National Academy of Sciences 107:9707–9711.

Murie, D. J. 1995. Comparative feeding ecology of two sympatric rockfish congeners, *Sebastes caurinus* (copper rockfish) and *S. maliger* (quillback rockfish). Marine Biology 124:341–353.

Newsome, S. D., M. T. Clementz, and P. L. Koch. 2010. Using stable isotope biogeochemistry to study marine mammal ecology. Marine Mammal Science 26:509–572.

Oosthuizen, W. C., P. J. N. De Bruyn, M. Wege, and M. N. Bester. 2016. Geographic variation in subantarctic fur seal pup growth: linkages with environmental variability and population density. Journal of Mammalogy 97:347–360.

Orsi, A. H., I. I. I. Whitworth, and W. D. Nowlin Jr. 1995. On the meridional extent and fronts of the Antarctic Circumpolar Current. Deep Sea Research I 42:641–673.

Pollock, D. E. 1991. Spiny lobsters at Tristan da Cunha, South Atlantic: inter-island variations in growth and population structure. South African Journal of Marine Science 10:1–12.

Polovina, J. J., E. Howell, D. R. Kobayashi, and M. P. Seki. 2001. The transition zone chlorophyll front, a dynamic global feature defining migration and forage habitat for marine resources. Progress in Oceanography 49:469–483.

Queiroz, N., N. E. Humphries, G. Mucientes, N. Hammerschlag, F. P. Lima, K. L. Scales, P. I. Miller, L. L. Sousa, R. Seabra, and D. W. Sims. 2016. Ocean-wide tracking of pelagic sharks reveals extent of overlap with longline fishing hotspots. Proceedings of the National Academy of Sciences USA 113:1582–1587.

Queiroz, N., F. P. Lima, A. Maia, P. A. Ribeiro, J. P. Correia, and A. M. Santos. 2005. Movement of blue shark, *Prionace glauca*, in the North-east Atlantic based on mark–recapture data. Journal of the Marine Biological Association of the United Kingdom 85:1107–1112.

Raymond, B., S. A. Shaffer, S. Sokolov, E. J. Woehler, D. P. Costa, L. Einoder, M. Hindell, G. Hosie, M. Pinkerton, P. M. Sagar, D. Scott, A. Smith, D. R. Thompson, C. Vertigan, and H. Weimerskirch. 2010. Shearwater Foraging in the Southern Ocean: The Roles of Prey Availability and Winds. PLoS One 5:e10960.

Reed, D., L. Washburn, A. Rassweiler, R. Miller, T. Bell, and S. Harrer. 2016. Extreme warming challenges sentinel status of kelp forests as indicators of climate change. Nature Communications 7.

Reed, D. C., A. Rassweiler, M. H. Carr, K. C. Cavanaugh, D. P. Malone, and D. A. Siegel. 2011. Wave disturbance overwhelms top-down and bottom-up control of primary production in California kelp forests. Ecology 92:2108–2116.

Robinson, P. W., D. P. Costa, D. E. Crocker, J. P. Gallo-Reynoso, C. D. Champagne, M. A. Fowler, C. Goetsch, K. T. Goetz, J. L. Hassrick, and L. A. Hückstädt. 2012. Foraging behavior and success of a mesopelagic predator in the northeast Pacific Ocean: insights from a data-rich species, the northern elephant seal. PLoS One 7:e36728.

Rodriguez, G. E., A. Rassweiler, D. C. Reed, and S. J. Holbrook. 2013. The importance of progressive senescence in the biomass dynamics of giant kelp (*Macrocystis pyrifera*). Ecology 94:1848–1858.

Ryan, P. G., editor. 2007. Field Guide to the Animals and Plants of Tristan da Cunha and Gough Island Pisces Publications, Newbury.

Ryan, P. G. 2016. Ingestion of Plastics by Marine Organisms. Pages 1–32. Springer Berlin Heidelberg, Berlin, Heidelberg.

Scales, K. L., P. I. Miller, L. A. Hawkes, S. N. Ingram, D. W. Sims, and S. C. Votier. 2014. REVIEW: On the Front Line: frontal zones as priority at-sea conservation areas for mobile marine vertebrates. Journal of Applied Ecology 51:1575–1583.

Scott, S. 2016. A Biophysical Profile of the Tristan da Cunha Archipelago. A report to Pew Charitable Trusts.

Scott, S., and P. Franklin. 2011. Potential effects on Tristan lobster fishery following the wreck of the Oliva on Nightingale Island, Tristan da Cunha. Unpublished report.

Shears, N. T., and R. C. Babcock. 2002. Marine reserves demonstrate top-down control of community structure on temperate reefs. Oecologia 132:131–142.

Smale, M. J., M. R. Clarke, N. T. W. Klages, and M. A. C. Roeleveld. 1993. Octopod beak identification — resolution at a regional level (Cephalopoda, Octopoda: Southern Africa). South African Journal of Marine Science 13:269–293.

Smale, M. J., G. Watson, and T. Hecht. 1995. Otolith atlas of southern African marine fishes. JLB Smith Institute of Ichthyology.

Smith, C. D. 2003. Diet of Octopus vulgaris in False Bay, South Africa. Marine Biology 143:1127–1133.

Smythe-Wright, D., P. Chapman, C. Duncombe Rae, L. V. SHannon, and S. M. Boswell. 1998. Characteristics of the South Atlantic subtropical frontal zone between 5°W and 5°E. Deep-Sea Research I 45:167–192.

Steneck, R. S., M. H. Graham, B. J. Bourque, D. Corbett, J. M. Erlandson, J. A. Estes, and M. J. Tegner. 2002. Kelp forest ecosystems: biodiversity, stability, resilience and future. Environmental conservation 29:436–459.

Tegner, M., P. K. Dayton, P. B. Edwards, and K. L. Riser. 1996. Is there evidence for long-term climatic change in southern California kelp forests? California Cooperative Oceanic Fisheries Investigations Reports 37:111–126.

Tegner, M. J., P. K. Dayton, P. B. Edwards, and K. L. Riser. 1997. Large-scale, low-frequency oceanographic effects on kelp forest succession: a tale of two cohorts. Marine Ecology Progress Series 146:117–134.

Tieszen, L. L., T. W. Boutton, K. G. Tesdahl, and N. A. Slade. 1983. Fractionation and turnover of stable carbon isotopes in animal tissues: Implications for δ13C analysis of diet. Oecologia 57:32–37.

Tristan da Cunha Government, and RSPB. 2012. Biodiversity Action Plan for the Tristan da Cunha Islands (2012–2016). Edinburgh of the Seven Seas, Tristan da Cunha, South Atlantic.

Utter, B. D., and M. W. Denny. 1996. Wave-induced forces on the giant kelp *Macrocystis pyrifera* (Agardh): Field test of a computational model. The Journal of Experimental Biology 199:2645–2654.

Valdez, M. C., E. S. Zaragoza, D. L. Belda, R. Marcos, and R. A. Ramírez. 2003. Effect of climatic change on the harvest of the kelp *Macrocystis pyrifera* on the Mexican Pacific Coast. Bulletin of Marine Science 73:545–556.

Vandeperre, F., A. Aires-da-Silva, J. Fontes, M. Santos, R. Serrao Santos, and P. Afonso. 2014. Movements of blue sharks (*Prionace glauca*) across their life history. PLoS One 9:e103538.

Vaudo, J. J., B. M. Wetherbee, A. D. Wood, K. Weng, L. A. Howey-Jordan, G. M. Harvey, and M. S. Shivji. 2016. Vertical movements of shortfin mako sharks *Isurus oxyrinchus* in the western North Atlantic Ocean are strongly influenced by temperature. Marine Ecology Progress Series 547:163–175.

Vergés, A., P. D. Steinberg, M. E. Hay, A. G. Poore, A. H. Campbell, E. Ballesteros, K. L. Heck, D. J. Booth, M. A. Coleman, and D. A. Feary. 2014. The tropicalization of temperate marine ecosystems: climate-mediated changes in herbivory and community phase shifts. Page 20140846 in Proc. R. Soc. B. The Royal Society.

Walters, A. 2014. Quantifying the trophic linkages of Antarctic marine predators. University of Tasmania, Tasmania, Australia.

Wanless, R. M., A. Angel, R. J. Cuthbert, G. M. Hilton, and P. G. Ryan. 2007. Can predation by invasive mice drive seabird extinctions? Biology Letters 3:241–244.

Wanless, R. M., S. Scott, W. H. H. Sauer, T. G. Andrew, J. P. Glass, B. Godfrey, C. Griffiths, and E. Yeld. 2009. Semi-submersible rigs: a vector transporting entire marine communities around the world. Biological Invasions 12:2573–2583.

Wege, M. 2017. Population trend and foraging ecology of Antarctic and Subantarctic fur seals at Marion Island.

Wege, M., M. P. Etienne, W. C. Oosthuizen, R. R. Reisinger, M. N. Bester, and P. J. N. de Bruyn. 2016a. Trend changes in sympatric Subantarctic and Antarctic fur seal pup populations at Marion Island, Southern Ocean. Marine Mammal Science 32:960–982.

Wege, M., M. Nevoux, P. J. N. de Bruyn, and M. N. Bester. 2015. Multi-state mark-recapture models as a novel approach to estimate factors affecting attendance patterns of lactating subantarctic fur seals from Marion Island. Antarctic Science 27:252–262.

Wege, M., C. A. Tosh, P. J. N. de Bruyn, and M. N. Bester. 2016b. Cross-seasonal foraging site fidelity of subantarctic fur seals: implications for marine conservation areas. Marine Ecology Progress Series 554:225–239.

Weng, K. C., D. G. Foley, J. E. Ganong, C. Perle, G. L. Shillinger, and B. A. Block. 2008. Migration of an upper trophic level predator, the salmon shark *Lamna ditropis*, between distant ecoregions. Marine Ecology Progress Series 372:253–264.

Wernberg, T., B. D. Russell, P. J. Moore, S. D. Ling, D. A. Smale, A. Campbell, M. A. Coleman, P. D. Steinberg, G. A. Kendrick, and S. D. Connell. 2011. Impacts of climate change in a global hotspot for temperate marine biodiversity and ocean warming. Journal of Experimental Marine Biology and Ecology 400:7–16.

Wernberg, T., D. A. Smale, F. Tuya, M. S. Thomsen, T. J. Langlois, T. De Bettignies, S. Bennett, and C. S. Rousseaux. 2013. An extreme climatic event alters marine ecosystem structure in a global biodiversity hotspot. Nature Climate Change 3:78–82.

Wilkinson, I. S., and M. N. Bester. 1990. Continued population increase in fur seals, *Arctocephalus tropicalis* and *A. gazella*, at the Prince Edward Islands. South African Journal of Antarctic Research 20:58–63.

Williams, R., A. McEldowney, and others. 1990. A guide to the fish otoliths from waters off the Australian Antarctic Territory.

Worm , B., H. K. Lotze, and R. A. Myers. 2003. Predator diversity hotspots in the blue ocean. . Proceedings of the National Academy of Sciences USA 100:9884–9888.

Zainuddin, M., H. Kiyofuji, K. Saitoh, and S.-I. Saitoh. 2006. Using multi-sensor satellite remote sensing and catch data to detect ocean hot spots for albacore (*Thunnus alalunga*) in the northwestern North Pacific. Deep Sea Research Part II: Topical Studies in Oceanography 53:419–431.

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