Assessment of the risk of surface longline fisheries in the Southern Hemisphere to albatrosses and petrels, for 2016

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Contents

1 Abstract

Assessing the bycatch of seabirds is a key step in understanding the impact of fishing on their populations. In this analysis, we used the seabird risk assessment methodology that was developed in New Zealand to estimate the bycatch of 26 albatross and petrel taxa that breed in the Southern Hemisphere. The bycatch estimates were related to population productivity to estimate the impact of surface longline fishing on the populations.

This analysis was based on observer data provided by Japan, South Africa, Australia and New Zealand. These countries all record the seabird species caught during observed fishing, allowing the estimation of seabird bycatch at the species level. The observed captures were related to the overlap between the observed fishing and the distributions of the seabirds (with the latter derived from tracking data where possible). From the fitted risk assessment model, captures were then estimated across all surface longline fishing effort in the Southern Hemisphere (as reported to regional fisheries management organisations). These estimates do not include any estimates of cryptic mortality or survival of released seabirds.

Across all the seabird taxa and surface longline fishing effort included in this study, the total estimated annual captures were 41 078 (95% c.i.: 39 432 to 42 746). Among all the taxa, captures of grey-headed albatross were the highest (8 444; 95% c.i.: 7 796 to 9 100). The petrel with the highest captures was white-chinned petrel, with estimated annual captures of 5 392 (95% c.i.: 2 131 to 13 166). There were nine species for which the estimated annual captures in surface longline fisheries exceeded the population productivity: Amsterdam albatross, sooty albatross, Tristan Albatross, Gibson's albatross, grey-headed albatross, Buller's albatross, black petrel, spectacled petrel and wandering albatross.

The results are preliminary at this stage; however, this analysis demonstrates how distribution information, together with observer data of seabird bycatch, may be used to estimate the impact of fisheries bycatch on seabird populations.

2 Introduction

New Zealand has been utilising and refining a spatially explicit assessment of risk to seabirds from commercial fishing (e.g., Richard & Abraham 2013a; Richard et al. 2017; Sharp 2017). The risk assessment method was applied to surface longline fishing, first by using New Zealand bycatch data to estimate seabird bycatch in surface longline fishing throughout the Southern Hemisphere (Abraham et al 2017a,b); second by using observer data from New Zealand and Japan to estimate the bycatch of great albatross species in surface longline fishing throughout the Southern Hemisphere (Daisuke et al. 2018). These studies were intended to demonstrate the method, while acknowledging limitations in the input data, in particular in the distributions of seabirds, and in the use of observer data from a limited number of fleets. The risk assessment method was also used as part of a Common Oceans project, led by Birdlife International, to estimate seabird bycatch of species listed by the Agreement on the Conservation of Albatrosses and Petrels (ACAP).

In this analysis, we updated the work presented to the Commission for the Conservation of Southern Bluefin Tuna (CCSBT) Ecologically Related Species Working Group (ERSWG) 12 by Abraham et al. (2017a), to estimate seabird bycatch and associated risk for 26 albatross and petrel taxa that breed in the Southern Hemisphere (Table 1). These taxa are the 25 species listed by the Agreement for the Conservation of Albatrosses and Petrels (ACAP), and which breed south of 20°S, with Antipodean albatross being split into two subspecies. In the current iteration, there were several key changes to the analysis:

1. Observer data were used from all CCSBT member countries that have species-specific observer data on seabird bycatch (Japan, South Africa, Australia, and New Zealand).

2. Effort data, derived from regional fisheries management organisations (RFMOs), used for estimating total seabird bycatch were revised and updated.

3. For seabird species with sufficient tracking data, the analysis used seabird distribution information that was derived from the tracking data.

4. A separate catchability was estimated for each fleet.

3 Methods

3.1 RISK RATIO

The methodology used for estimating the risk followed the Spatially Explicit Risk Assessment Framework (SEFRA), which is the approach currently used in New Zealand for assessing the risk of commercial fisheries to seabirds (Richard et al. 2013, 2017; Sharp et al. 2013; Sharp 2017).

Following this method, the risk ratio (RR) is estimated as the seabird bycatch in fisheries (specifically referred to here as the estimated annual capture, EAC) divided by a measure of the population productivity, the population sustainability threshold (PST):

$$RR = EAC/PST.$$

Uncertainty was carried through all parameters in the calculation, so there was uncertainty in the resulting risk ratio. The EAC was estimated from a combination of observer effort data, observed bycatch, seabird distribution data and fisheries effort data. The EAC is a statistical extrapolation of bycatch rates from observer data to all fishing, on the assumption that seabird bycatch is proportional to the overlap between seabird distributions and fishing effort. The PST is an estimate of the productivity of seabird populations, and is closely related to the Potential Biological Removal index (PBR), used to estimate the productivity of marine mammal populations (Wade 1998).

The applied risk methodology has the advantage of being fully quantitative: the ratio is a direct comparison between seabirds killed in surface longline fisheries and the number of seabirds that can be produced by the population.

3.2 POPULATION SUYSTAINABILITY THRESHOLD

The PST is an estimate of the maximum number of annual human-caused mortalities that can occur, while allowing populations to achieve a defined management goal. The goal defined by Fisheries New Zealand for the seabird risk assessment in New Zealand waters is that populations are above half the carrying capacity (with 95% certainty) after 200 years (e.g., Richard et al. 2017). The PST is defined as:

$$PST = ½ \, \nu \, r_{max} \, N,$$

where r_{max} is the maximum population growth rate, under optimal conditions, and N is the total population size. The parameter ν is a factor set so that the management goal may be achieved by the seabird population. In this study, ν was set to 0.5, based on simulations by Richard et al. (2013). The maximum growth rate r_{max} was calculated using the demographic invariants method of Niel & Lebreton (2005), based on the optimal adult annual survival rate and age at first breeding.

The PST was derived for New Zealand seabirds (most recently by Richard et al. 2017), based on the PBR of Wade (1998). The PBR is numerically equivalent to the PST, with the exception that the PBR uses a minimum point estimate of the population size, and a point estimate of the maximum growth rate, whereas the PST includes uncertainty in all the parameters. In the PBR, a recovery factor, f, is used in place of ν. Nevertheless, the default value of the recovery factor for non-endangered populations is also 0.5. In the calibration of the PST used here, the threat status of the population was not considered.

3.3 ESTIMATING ANNUAL CAPTURES

The total number of incidental captures of seabirds was estimated by assuming that, for similar species, and for similar fisheries, the number of incidental captures of protected species is proportional to the overlap between the density of the populations and the fishing. Here, the density overlap (θ) between a species or taxon (s) and the fishing effort within a group of fisheries (f) was calculated by summing the product of fishing intensity, population size and the relative density of a species at the location of the fishing:

$$\theta_{sf} = N_s O_{sf},$$

$$O_{sf} = \sum_i p_{si} h_{fi}$$

where N_s is the total population size, O_{sf} is the population-independent overlap, i is an index of the fishing events within the fisheries group, p_{si} is the relative population density at the location of the fishing (p has units of km^{-2} and is calibrated to integrate to one over the Southern Hemisphere), and h_{fi} is the number of hooks associated with the fishing event.

Captures of seabirds are recorded by observers when they are onboard fishing vessels. The expected number of incidents is assumed to be proportional to the density overlap. The mean capture rate recorded by observers (μ'_{sf}) is then given by:

$$\mu'_{sf} = q_{sf}\theta'_{sf},$$

where q_{sf} is the vulnerability of a species, s, to capture in a fleet, f, per unit of density overlap, θ'_{sf}. The prime symbol was used to indicate observed quantities.

In this analysis, it was assumed that the vulnerability could be represented as a combination of a susceptibility, q_g, that was assumed to be the same for all seabirds within each taxon group g (see Table 1), and a catchability, q_f that was assumed to be the same for all seabirds within each fleet:

$$q_{sf} = q_{g(s)}q_f \, \varepsilon_{g(s)f},$$

where the term ε_{gf} represents the interaction between the catchability and the susceptibility. There were sixteen seabird species groups (see Table 1) and six fleets included in the modelling: Japan (JPN) high seas, South Africa (ZAF) joint venture, South Africa domestic, Australia (AUS) domestic, New Zealand (NZL) joint venture and New Zealand domestic.

Not all captured seabirds could be identified to the species group level: some captures were only recorded as unidentified seabirds, and some captures were only identified to the family level (either albatrosses or petrels). From the mean capture rate, the number of observed captures identified to the taxon-group level, C'_{gf}, is given by:

$$C'_{gf} \sim \text{Poisson}(\, p_{observable}p_f^{bird}p_{F(g)f}^{family}\mu'_{gf}).$$

The probability, $p_{observable}$, is the probability that an incident that occurred while an observer was on the vessel would be recorded; not all incidental captures are recorded, for example, as captured bird may fall off the hook before being brought on board. In this study we assume that $p_{observable} = 1$, and so we are not accounting for cryptic mortality. In previous applications of the risk assessment to surfacelongline fishing, a mean value for $p_{observable}$ of 0.48 (95% c.i.: 0.41–0.55) was used, based on a study by Brothers et al. (2010). The probability p_f^{bird} is the probability that a capture is identified to a level better than a seabird (estimated separately for each fishery), and p_{Ff}^{family} is the probability that a capture is identified to a level better than the family, F (estimated separately for each seabird family and fleet). The number of observed unidentified seabird captures in a fleet, S'_f, can then be estimated as:

$$S'_f \sim \text{Poisson}((1 - p_f^{bird}) \sum_g \mu'_{gf}),$$

and the number of seabird captures that are only identified to the family level, F'_{Ff}, can be estimated as:

$$F'_{Ff} \sim \text{Poisson}(p_f^{bird}(1 - p_{Ff}^{family}) \sum_{g \in F} \mu'_{gf}).$$

There were also captures that were only identified to the genus level. Because the model used here assumed that the susceptibility is the same for all species within the same genus, the captures do not need to be resolved below the genus level. In the previous analysis (Abraham et al. 2017a), the susceptibility was defined to be the same for species groups (such as shy albatrosses); however, unidentified captures were not accounted for. Allowing the vulnerability to vary by species group would require handling unidentified captures below the genus level.

In the previous analysis (Abraham et al. 2017a), alive and dead captures were estimated separately, and total mortalities were estimated by assuming a uniformly distributed prior for the survivability between 0 and 1 (a mean of 50% survival of live-released seabirds). Nevertheless, in the current analysis, not all the data had records of the alive or dead status of the seabirds. For this reason, all captures were assumed to be mortalities. In observed captures of New Zealand seabirds caught in surface longline fisheries between 2012 and 2017, 19.1% of the captures were recorded by observers as released alive.

The model was fitted to the data and estimated using Bayesian methods, within the software Stan. A vague log-normal prior was used for q_{sg}, a vague beta prior was used for p_{alive}, but an informed beta prior was placed on $p_{observable}$ as there was no information available within the model to constrain it. For this parameter, the two shape parameters were calculated from the mean and variance of the probability of retrieving a captured bird, derived from Brothers et al. (2010). The model was fitted using two Markov Chain Monte Carlo (MCMC) chains, with a burn-in of 10 000 iterations; posteriors were calculated from 800 000 further iterations, retaining a sample value every 400 iterations. Convergence and mixing were visually assessed from the MCMC trace of the parameters, and by requiring that the $\hat{R}$ parameter (which compares variation within chains and between chains) was less than 1.1 for all parameters.

Having fitted the model, the number of annual captures of a taxon s in fishing effort in the fishing group g could be estimated from the fitted vulnerability and the overlap as:

$$EAC \sim \text{Poisson} (p_{observable} q_{sf} \theta_{sf}).$$

In estimating both the annual captures and the annual potential fatalities, it is necessary to assume a catchability for fleets that are not represented in the observer data. Possible ways of achieving this assumption would be to randomly draw the unknown fleets using the random effects in the model. In this case, however, we assumed that all the fleets that were not represented in the data followed similar methods with respect to seabird catchability as the Japan high-seas fleet. It was assumed that fishing practices by these Japanese vessels, with respect to seabird bycatch, are similar to other high-seas vessels.

3.4 FISHING EFFORT

Fishing effort data were sourced and prepared by Francis & Hoyle (2019). Their methods are summarised as follows. Surface longline fishing effort data were sourced from five tuna RFMOs. The effort data were requested at a 5-degree spatial resolution, by quarter (with quarter 1 being January through March), by year and by flag. Surface longline data from four of the tuna RFMOs (CCSBT, IATTC, ICCAT, IOTC) were downloaded from public websites, and data were requested from WCPFC. The data request covered data from the beginning of the series to the end of the 2016 fishing year.

There is overlap between the reporting to the different RFMOs (so the same fishing may be reported to two RFMOs). Starting with the CCSBT dataset, the IOTC, ICCAT, IATTC, and WCPFC data were progressively combined. When adding each dataset, they were merged by the strata latitude, longitude, flag, year and month (or quarter), with the larger reported effort in each stratum being retained. In the case of IATTC, the shark dataset was merged with the (standard) IATTC tuna and billfish dataset, as the "shark" dataset contains effort that

caught no tunas or billfish. To avoid confidentiality breaches, WCPFC provided a more complete dataset without a flag field. After merging the WCPFC dataset that had a flag field with those from other tuna RFMOs, the version of the WCPFC dataset that had no flag field was merged. When the total effort was higher in the latter dataset, the differing data were added as a new stratum with flag ''Unknown'' (UNK).

For the current analysis, we used effort data from 2016 for estimating total captures (see Figure 1 for the distribution of the total number of hooks set). During 2016, the total number of hooks set in the Southern Hemisphere was 862 million. Of these hooks, 172 million hooks were set south of 25°S. The hooks only included surfacelongline fishing that was reported to RFMOs, and did not include all fishing within Southern Hemisphere Exclusive Economic Zones (EEZs). The highest fishing intensity was in the western tropical Pacific Ocean, north of 10°S, where some 5-degree cells had over 20 million hooks set per year. The cell with the highest intensity, south of 10°S was to the east of South Africa (centred on 32.5°E, 37.5°S), with 15.7 million hooks set in this cell during 2016.

3.5 OBSERVER DATA

Observer data, including the number of hooks observed and the number of seabirds reported as bycatch were obtained from Fisheries New Zealand, the National Research Institute of Far Seas Fisheries, Japan, the Department of Agriculture, Forestry and Fisheries, South Africa, and the Australian Bureau of Agricultural and Resource Economics and Sciences (Table 2). The observer data were reported in the same resolution as the effort data (per 5-degree cell, per quarter). The data used in the estimation covered the period 2012 to 2018. In New Zealand and South Africa, there was fishing by Japanese vessels under a joint venture arrangement. These vessels were treated as separate fleets during the model estimation.

There was a total of 4925 observed seabird captures included in the analysis. There were three captures that were of species other than the study taxa, and these were not included. Observed captures were dominated by *Thalassarche* albatross, with a total of 2704 observed captures.

3.6 DEMOGRAPHIC DATA

The values of the demographic parameters for the 26 taxa were gathered from the scientific literature, peer-reviewed articles and government research reports. Estimates were obtained for age at first breeding, adult annual survival rate, and proportion of adults breeding. Values of similar species were used for survival and age at first breeding when no value was available for one of the study species. When only a point estimate was available, an uncertainty was assigned using a set of rules that was based on the quality of the information, as described in Richard & Abraham (2017).

The total population size was calculated from the annual number of breeding pairs, the age at first breeding and adult annual survival rate from the literature, and the proportion of adults breeding, using the formula:

$$N = 2 \mu N_{BP} S^{1-A} / P_B,$$

where N is the total population size, N_{BP} the annual number of breeding pairs, S the adult annual survival, A the age at first breeding, P_B the proportion of adults breeding, and μ a correction factor. The coefficient S^{1-A} is the ratio of the total population size to the number of adults, derived from a simple population model in which annual survival is assumed to be constant between age classes. This assumption was found to underestimate the total population size, because juvenile survival is generally lower than that of adults (Richard & Abraham 2013b). The correction factor μ accounts for this bias, and was estimated from simulations of population dynamics, using values between 1.45 for black petrel and 1.77 for Antipodean albatross (Richard & Abraham 2013b).

3.7   SEABIRD DISTRIBUTIONS

Seabird distributions were derived from tracking data following methods similar to those by Carneiro et al. (2019), but with several key differences, reflecting the requirements of the analysis. Tracking data were obtained from a request to tracking data owners, through the Birdlife International Seabird Tracking Database (http://www.seabirdtracking.org/).

Each deployment was first processed to remove the first three days (to reduce a bias caused by seabirds being tagged at the colony). Second, any gaps of longer than 24 hours in the tracking data were discarded, by splitting the deployment into separate tracks. Third, each track was interpolated regularly in time (hourly intervals) to obtain a set of points that were equally-spaced in time. The number of interpolated points falling within each 5-degree square was counted, and this gridded track distribution was normalised to integrate to one. Because this analysis was at a 5-degree scale, no kernel density estimation was carried out, as the resolution of the 5-degree grid is lower than typical kernel densities. Because of the coarse spatial scale, tracks with positions derived from Global Positioning System (GPS) or Geolocators (GLS) were treated in the same way.

For each species and breeding site, the tracks were grouped into tracks from breeding, non-breeding, and juvenile seabirds. Tracks that were initially for breeding seabirds, but that continued outside their breeding season, were split with each part assigned to the corresponding season. Furthermore, for petrels, tracks of breeding seabirds were split at 3 000 km from their colony, with the portion of the tracks beyond this distance being assigned to non-breeding seabirds.

For each species and site, the tracking distribution of juveniles with less than 15 tracks or 5 000 tracking points was derived from the average of the distribution of non-breeding adults and of the distribution of juveniles, weighted by the respective number of points in each distribution.

Tracking data were not available or insufficient for some combinations of species, site, and population class and so range maps were also required. For juveniles and non-breeding adults, a simple distribution with a uniform density across the range of the species was derived, based on range maps (BirdLife International and Handbook of the Birds of the World 2018). These range maps were the same for all breeding sites. For breeding adults, the range map was supplemented by adding breeding seabirds, based on an exponential decay function around the colony, so that 90% of their movement occurred within 1 500 km from the colony.

For all species, sites and classes, the distribution was derived as a weighted average of the tracking and range distributions, weighted by the number of hourly points used to derive the tracking distribution (the range distribution was assigned a weight of 5 000). For species, sites and classes with considerable tracking data, the range maps had little weight.

A simple demographic matrix model (with number of breeding pairs, age at first breeding, juvenile survival, adult survival, proportion of successful/unsuccessful breeding seabirds and non-breeding seabirds breeding the following year or not) was used to estimate the proportion of the population at each breeding site that were juvenile, adult breeders or adult non-breeders, within each of the quarterly periods. The gridded track distributions were weighted by the proportion of seabirds in each class, and then combined to provide a normalised distribution for each species and breeding site.

Finally, the population-weighted distributions from each colony were combined to obtain a distribution for the species as a whole.

Across all taxa, there were 7.2 million hours of tracking data available to the analysis. Of the 26 taxa, there were 24 taxa that had at least some tracking data available (no tracking data was requested for either of the two giant petrel species). There were 2.2 million hours of tracking data available for black-browed albatross, and 1.8 million hours available for wandering albatross. Nevertheless, there were three species (southern royal albatross, Campbell black-browed albatross and spectacled petrel) that had less than 10 000 hours of

tracking data. Distinguished by life stage, there were 21 species with more than 10 000 hours of tracking data available for breeding adults; 18 species with more than 10 000 hours of tracking data available for non-breeding adults; and 9 species with more than 10 000 hours of tracking data available for juveniles.

Species richness (the number of species occurring within each 5-degree cell, with a density higher than one bird per 1000 km^2) of the 26 seabird taxa included in this study was highest between 40 and 50°S, with the highest richness occurring near New Zealand, Chile and South Africa (Figure 2).

Tracking data were missing for many breeding sites (Figure 3). There were no tracking data available for 44.8% of species, life-stage, and breeding site combinations (only including breeding sites that account for more than 5% of the total population). There were limited tracking data (less than 5000 hours) for 59.6% of species, life-stage, and breeding site combinations.

An indication of the completeness of the distribution information is obtained by comparing the location of observed captures with the distribution maps (Table 3). There were some captures were outside of the range of the distribution maps. Overall the proportion of these captures was low (less than 1%); however, for three species (light-mantled sooty albatross, southern royal albatross, spectacled petrel) over 5% of the observed captures were outside of the range of the distributions.

4   Results

4.1   ESTIMATED VULNERABILITY

The vulnerability of each species group and fleet was estimated within the model (Figure 4). The highest vulnerabilities were for captures by the Japan high-seas and the New Zealand domestic fleets. Across taxa, the vulnerability was highest for grey-headed albatross, black petrel, Buller's albatross and Westland petrel. The vulnerability parameters had high uncertainty (close to two orders of magnitude).

The estimated catchability parameters were highest for the Japan high-seas fleet, and were lowest for the joint venture fleets in South Africa and New Zealand (Table 4). Over the period covered by the observer data, these joint venture fleets were Japanese vessels.

4.2   ESTIMATED CAPTURES AND RISK

Across all the seabird taxa and surface longline fishing effort included in this study, the total estimated annual captures were 41 078 (95% c.i.: 39 432 to 42 746; Table 5). The total estimated captures were similar to estimates obtained using other methods (such as fitting a generalised additive model to the observed captures of all seabirds; Birdlife South Africa, in press). Among all taxa, estimated captures of grey-headed albatross were the highest (8 444; 95% c.i.: 7 796 to 9 100). The petrel species with the highest estimated annual captures was white-chinned petrel 7 550 (95% c.i.: 6 550 to 8 630).

To understand the potential impact of these captures on the populations of each species, the estimated captures needed to be considered relative to the population productivity. The PST varied by four orders of magnitude, from 1.8 (95% c.i. 0.9–3.5) for Amsterdam albatross to 91 600 (95% c.i.: 54 100 to 141 000) for white-chinned petrel (Table 5). There were nine taxa for which the median risk ratio was higher than one: Amsterdam albatross, sooty albatross, Tristan albatross, Gibson's albatross, grey-headed albatross, Buller's albatross, black petrel, spectacled petrel and wandering albatross (Figure 5).

The captures within the risk assessment model may be estimated spatially, at the same resolution as the fishing effort data (Figure 6). The assessment model estimated that the highest captures, within any 5-degree cell, were to the south-east of South Africa. This distribution reflected both the occurrence of seabirds with high vulnerability to capture and high fishing effort. The summed risk, across all taxa, was also highest to the south-east of South Africa (Figure 7).

5 Discussion

5.1 SPECIES AT RISK

Of the nine taxa that had a median risk ratio higher than one, six taxa had population declines between 1993 and 2013 (Philipps et al. 2016). The other three species had populations that were increasing (Amsterdam albatross, spectacled petrel) or stable (Buller's albatross) over this period.

Amsterdam albatross was the taxa with the highest risk, with mean estimated annual captures of 5.8 (95% c.i.: 2 to 11) captures. Despite the apparent high risk, the population has increased (Thiebot et al. 2016). The apparent paradox of population recovery despite high overlap with fisheries has been noted before for this species, with the suggestion that bycatch mortality has selectively removed seabirds with risky behaviours, leaving seabirds that are less susceptible to bycatch (Barbraud et al. 2013). Spectacled petrel had little tracking data (a total of 2 400 hours, all from adult non-breeding seabirds), and so the distribution largely reflected the range map for this species. This aspect may make estimates of the overlap (and thus estimated captures) less reliable. For Buller's albatross, the population breeding at the Snares Islands has remained stable, but the estimated adult survival was around 0.91 during the period 2005 to 2016 (Sagar et al. 2017). It is possible that bycatch has reduced adult survival, but this decrease has not yet been reflected in the size of the population.

5.2 POTENTIAL LIMITATIONS

The analysis relied greatly on seabird distribution information. Where possible, tracking data were used to derive distributions; however, tracking data were limited for many life-stages and sites. For example, there were only nine species that had more than 10 000 hours of tracking of juvenile seabirds. To address limitations with data gaps, the tracking data were amalgamated with range maps. For breeding seabirds, the range maps were supplemented with an increased density near the colony.

As more tracking data are collected, the quality of the resulting distributions will improve, and there will be less need to use heuristics to fill in data gaps. The current risk assessment method could be used to help prioritise the collection of more tracking data for particular species, breeding stages and sites.

The current analysis also depended on observer effort and bycatch data, with seabird captures identified to the species level. The information is only required at a coarse resolution (5-degree cells, quarterly). Developing accurate reporting at the species level requires investment in taxonomic skills, in systems for processing bycaught seabirds, and potentially in data management.

The analysis in this report relied on the assumption that the catchability of seabirds in high-seas fleets for which no observer data were available was the same as by the Japan high-seas fleet. This assumption is important structurally and difficult to validate.

Effort data used in this analysis are unlikely to be complete. Fishing effort data were obtained from the RFMOs, but not all countries report fishing within their EEZs to the RFMOs. In addition, the range of some of the species extends into the Northern Hemisphere, especially into the North Pacific Ocean. Global data on surface longline effort is needed to fully complete the analysis to encompass all fishing that overlaps with these species.

Previous applications of the risk assessment have explicitly included a cryptic mortality, accounting for seabirds that would be killed by fishing but not recorded by observers (for example, because they fell off the hook). Information on cryptic mortality relies on a single study by Brothers et al. (2010). If this study is used to inform the cryptic mortality, then the risk approximately doubles. We have not included a survivability parameter, so that live released seabirds were assumed to be mortalities. In the New Zealand observer data, 19.1% of the seabirds captured in surface longline fisheries were recorded as released alive (compared with 2.2% in the data from Japan).

The current analysis only included surface longline data, but it could be extended to other fishing methods. This extension would allow a direct comparison of the impacts of surface longline fishing to impacts from other methods. The risk assessment would then allow for a quantitative prioritisation across fishing methods.

5.3   CONCLUSIONS

This was a preliminary analysis, and the results demonstrate how the use of seabird distributions and observer data can be used to estimate bycatch at the species or population level. It is important to estimate bycatch at the species or population level, rather than aggregated to all species, as the impacts of fishing bycatch depend on population status.

Nevertheless, to provide confidence in the results, further research is required. In particular, ongoing development of the seabird distribution data is important. This development includes refining the heuristic distributions that were used to fill the data gaps. In addition, we anticipate that further tracking data will become available in the future for some species, such as black petrel. The improvement of the seabird distribution data will allow testing and refinement of the underlying assumptions of the assessment model, i.e., that bycatch is proportional to overlap.

This work could be developed to a complete quantitative assessment across all seabirds and all longline fisheries. Achieving this goal would allow prioritisation of risk reduction across seabird species and across fisheries. Addressing these limitations will allow us to develop, for the first time, a comprehensive analysis of the risk posed to pelagic seabird species from surface longline fisheries in the Southern Hemisphere.

6   Acknowledgements

7 References

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8 Appendix

8.1 TABLES

Table 1. Taxa included in the current analysis of bycatch of seabirds in the Southern Hemisphere. The 25 species are listed by the Agreement for the Conservation of Albatrosses and Petrels (ACAP) and have breeding colonies in the Southern Hemisphere. Note that in this analysis, Antipodean albatross is represented as two subspecies, Antipodean and Gibson's albatrosses. Taxa were grouped to estimate their vulnerability to capture in surface longline fisheries.

Taxa group	Taxon	Scientific name
Wandering albatrosses	Wandering albatross	*Diomedea exulans*
	Antipodean albatross	*Diomedea antipodensis antipodensis*
	Gibson's albatross	*Diomedea antipodensis gibsoni*
	Tristan albatross	*Diomedea dabbenena*
	Amsterdam albatross	*Diomedea amsterdamensis*
Royal albatrosses	Southern royal albatross	*Diomedea epomophora*
	Northern royal albatross	*Diomedea sanfordi*
Yellow-nosed albatrosses	Atlantic yellow-nosed albatross	*Thalassarche chlororhynchos*
	Indian yellow-nosed albatross	*Thalassarche carteri*
Black browed albatrosses	Black-browed albatross	*Thalassarche melanophris*
	Campbell black-browed albatross	*Thalassarche impavida*
Grey-headed albatross	Grey-headed albatross	*Thalassarche chrysostoma*
Buller's albatross	Buller's albatross	*Thalassarche bulleri*
Shy albatrosses	Shy albatross	*Thalassarche cauta*
	White-capped albatross	*Thalassarche steadi*
Chatham Island albatross	Chatham Island albatross	*Thalassarche eremita*
Salvin's albatross	Salvin's albatross	*Thalassarche salvini*
Sooty albatrosses	Sooty albatross	*Phoebetria fusca*
	Light-mantled sooty albatross	*Phoebetria palpebrata*
Giant petrels	Southern giant petrel	*Macronectes giganteus*
	Northern giant petrel	*Macronectes halli*
White-chinned petrel	White-chinned petrel	*Procellaria aequinoctialis*
Westland petrel	Westland petrel	*Procellaria westlandica*
Black petrel	Black petrel	*Procellaria parkinsoni*
Grey petrel	Grey petrel	*Procellaria cinerea*
Spectacled petrel	Spectacled petrel	*Procellaria conspicillata*

Table 2. Observed hooks and captures, by fishing fleet and seabird genus. Observed effort is thousands of hooks, captures are absolute numbers and capture rate is seabirds per one thousand hooks. Fleets were: AUS, Australia; JPN, Japan; NZL: New Zealand; ZAF, South Africa. New Zealand and South Africa had domestic and joint venture (JV) fleets. Periods covered by the observer data were from 2013 to 2018 for the ZAF domestic fleet, from 2012 to 2015 for the NZL JV fleet, from 2012 to 2016 for the JPN high-seas fleet, and from 2012 to 2017 for the NZ and AUS domestic fleets. The 'All' column includes captures that were not identified to the genus level. Not included were three 3 of seabird species not part of this study, and 24 captures that were outside the range of the corresponding distributions.

Fleet	Hooks	Great albatrosses	*Thalassarche* spp.	*Phoebetria* spp.	Giant petrels	*Procellaria* spp.	All	Capture rate
All	27 298	269	2704	109	122	561	4925	0.137
AUS domestic	2578	0	8	0	0	0	19	0.002
JPN high seas	12 029	243	2276	109	122	292	4181	0.252
NZL domestic	1133	25	189	0	0	47	262	0.230
NZL JV	2384	0	82	0	0	2	84	0.035
ZAF domestic	1061	1	18	0	0	7	26	0.024
ZAF JV	8109	0	131	0	0	213	353	0.042

Table 3. Observed captures that were outside the range of the distribution maps, indicating limitations in the distributions. Shown for each species are the number of captures outside the range, the total number of captures, and the percentage of the total captures that were outside the range. The number of captures for all species only includes captures that were identified to the species level.

Species	No. outside range	Total	% outside range
Light-mantled sooty albatross	10	53	15.87
Campbell black-browed albatross	4	107	3.60
White-chinned petrel	4	405	0.98
Southern royal albatross	3	18	14.26
Spectacled petrel	2	19	9.52
Grey-headed albatross	1	627	0.16
All species	24	2741	0.87

Table 4. Estimated catchability for the different fishery fleets within the model. Shown are for each fleet the mean and 95% credible interval of the catchability parameter. Fleets were: AUS, Australia; JPN, Japan; NZL: New Zealand; ZAF, South Africa. New Zealand and South Africa had domestic and joint venture (JV) fleets. Fleets are sorted in order of decreasing mean catchability.

Fleet	Mean	95% c.i.
JPN high seas	6.14	2.52–12.59
NZL domestic	1.39	0.53–2.94
ZAF domestic	0.26	0.07–0.67
AUS domestic	0.19	0.04–0.53
ZAF JV	0.17	0.05–0.42
NZL JV	0.05	0.01–0.13

Table 5. Risk assessment of surface longline fisheries in the Southern Hemisphere for the seabird taxa included in the current risk assessment. Shown for each taxon are the mean or median and 95% credible interval (c.i.) of the Population Sustainability Threshold (PST), the Estimated Annual Captures (EAC), and the risk ratio, EAC/PST. A risk ratio exceeding 1 indicates that the number of captures is higher than the population sustainability threshold. Values are sorted in order of decreasing median risk ratio.

| Taxon	PST Mean	c.i.	EAC Mean	c.i.	EAC/PST Median	c.i.
Amsterdam albatross	1.8	0.9–3.5	5.8	2–11	3.35	0.81–8.95
Sooty albatross	515	341–738	1 350	1100–1620	2.65	1.76–4.14
Tristan albatross	202	72.7–482	377	316–441	2.16	0.77–5.24
Gibson's albatross	327	197–506	550	466–640	1.74	1.05–2.89
Grey-headed albatross	5 500	3770–7880	8 440	7800–9090	1.56	1.07–2.26
Buller's albatross	1 580	904–2500	2 260	2040–2480	1.47	0.89–2.48
Black petrel	127	76.1–197	191	72–390	1.44	0.49–3.68
Spectacled petrel	708	383–1190	948	576–1410	1.37	0.64–2.86
Wandering albatross	575	401–797	696	591–803	1.23	0.85–1.79
Light-mantled sooty albatross	968	686–1320	875	708–1050	0.92	0.62–1.34
Campbell black-browed albatross	1 170	704–1800	812	722–907	0.72	0.44–1.18
Indian yellow-nosed albatross	3 690	1780–7790	1 860	1540–2210	0.56	0.23–1.05
White-capped albatross	5 200	2960–8560	2 060	1870–2260	0.41	0.24–0.71
Shy albatross	610	347–1010	232	196–269	0.4	0.22–0.69
Westland petrel	237	128–396	90.1	45–155	0.38	0.16–0.88
Antipodean albatross	397	240–626	146	117–177	0.38	0.22–0.64
Northern giant petrel	1480	626–2960	493	402–589	0.36	0.16–0.81
Atlantic yellow-nosed albatross	3130	1990–4740	1 080	892–1290	0.35	0.22–0.57
Southern giant petrel	5150	3000–9450	1 460	1210–1710	0.3	0.15–0.5
Black-browed albatross	39 500	27 300–55 700	8 350	7580–9160	0.22	0.15–0.31
Southern royal albatross	602	362–948	126	76–187	0.21	0.11–0.41
Northern royal albatross	507	266–886	96	58–143	0.19	0.09–0.41
Grey petrel	6050	3500–9480	1 000	807–1230	0.17	0.1–0.3
Chatham Island albatross	160	89–269	21.5	0–86	0.09	0–0.66
White-chinned petrel	91 600	54 100–141 000	7 550	6550–8630	0.08	0.05–0.14
Salvin's albatross	1 130	629–1920	10.4	0–39	0.01	0–0.04
Total captures			**41 078**	**39 432–42 746**		

8.2 FIGURES
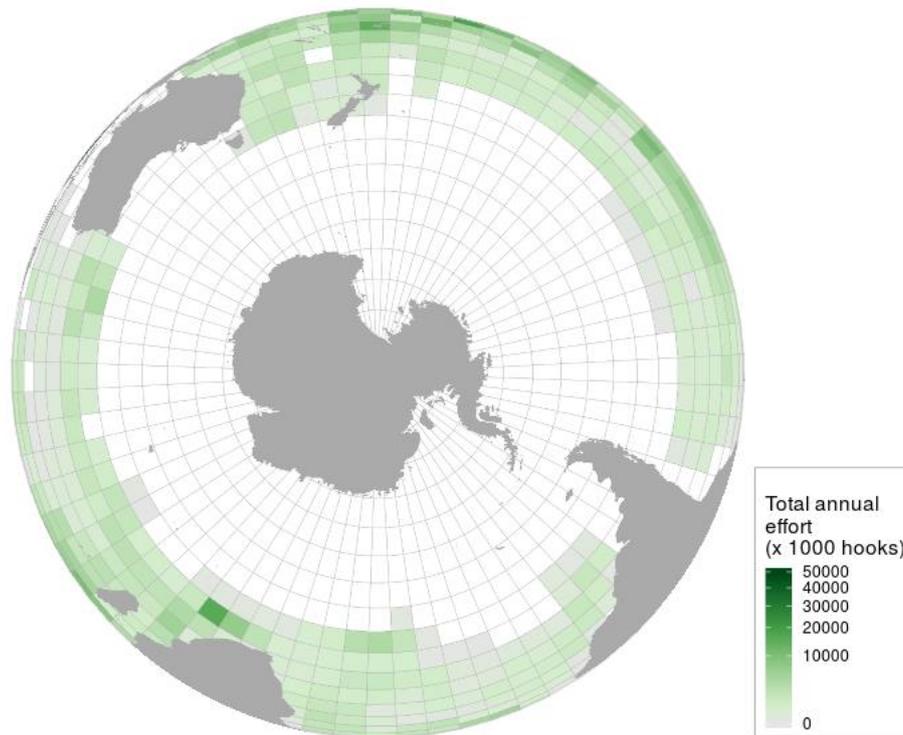
Figure 1. Fishing effort included in the current seabird risk assessment. The colour of each 5-degree cell represents the annual fishing effort within the cell during the 2016 fishing year.
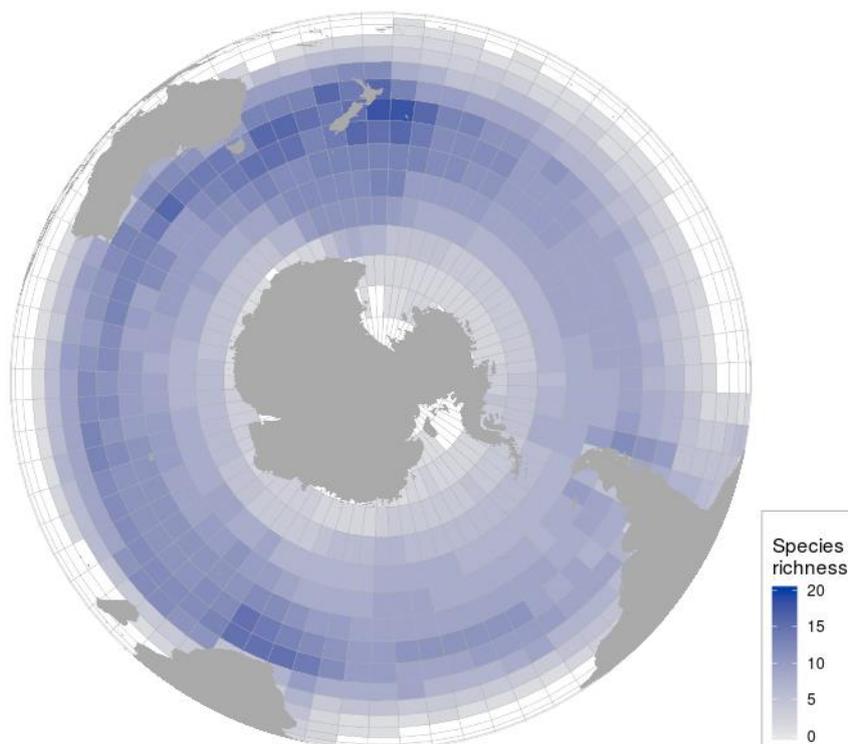

Figure 2. Species richness of the seabird taxa included in the current seabird risk assessment. For each 5-degree cell, the intensity of the colour shows the number of species within that cell that has a density higher than one bird per 1000 km^2.
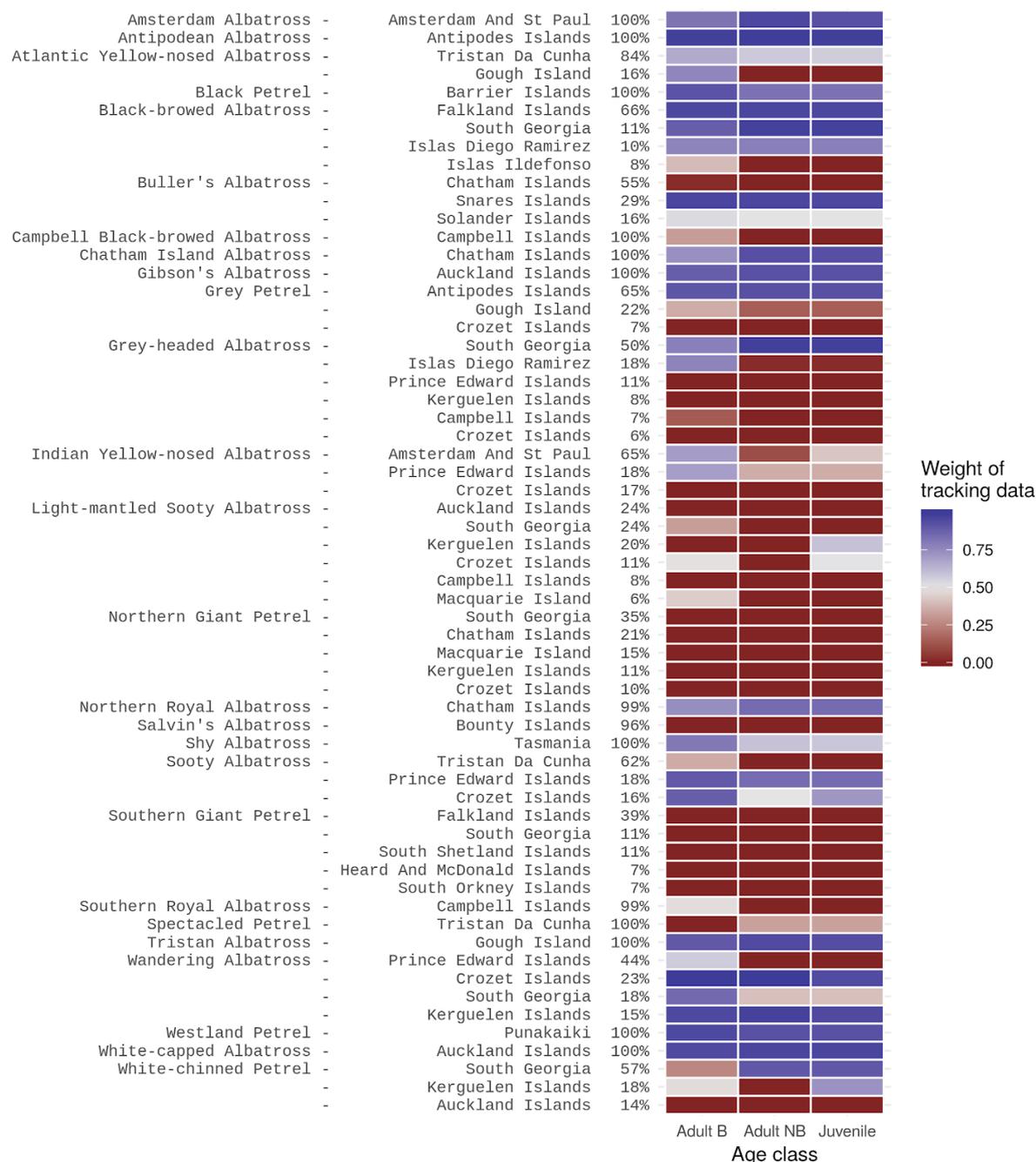
Figure 3: Use of tracking data for generating seabird distributions. For each species and breeding site, the figure illustrates the relative weight of the tracking data in generated the distribution (number of hours), compared with the weight of the combined distribution including a distribution derived from range maps (which were given a weight of 5000). Blue indicates that the distribution is mainly derived from tracking data, and red indicates that the distribution is mainly derived from the range maps. The number by each site indicates the percentage of the total global population breeding at that site. Sites with less than 5% of the global breeding population are not shown.

Figure 4. Estimated vulnerability by species group and fishery fleet, estimated from observed surface longline fishing between 2012 and 2018. Shown are for each species group and fleet, the mean and 95% credible interval of the vulnerability (captures per unit overlap) (JPN: Japan; NZL: New Zealand; ZAF: South Africa; AUS: Australia). The colour of the cells indicates the mean vulnerability (with a darker colour indicating a higher vulnerability).
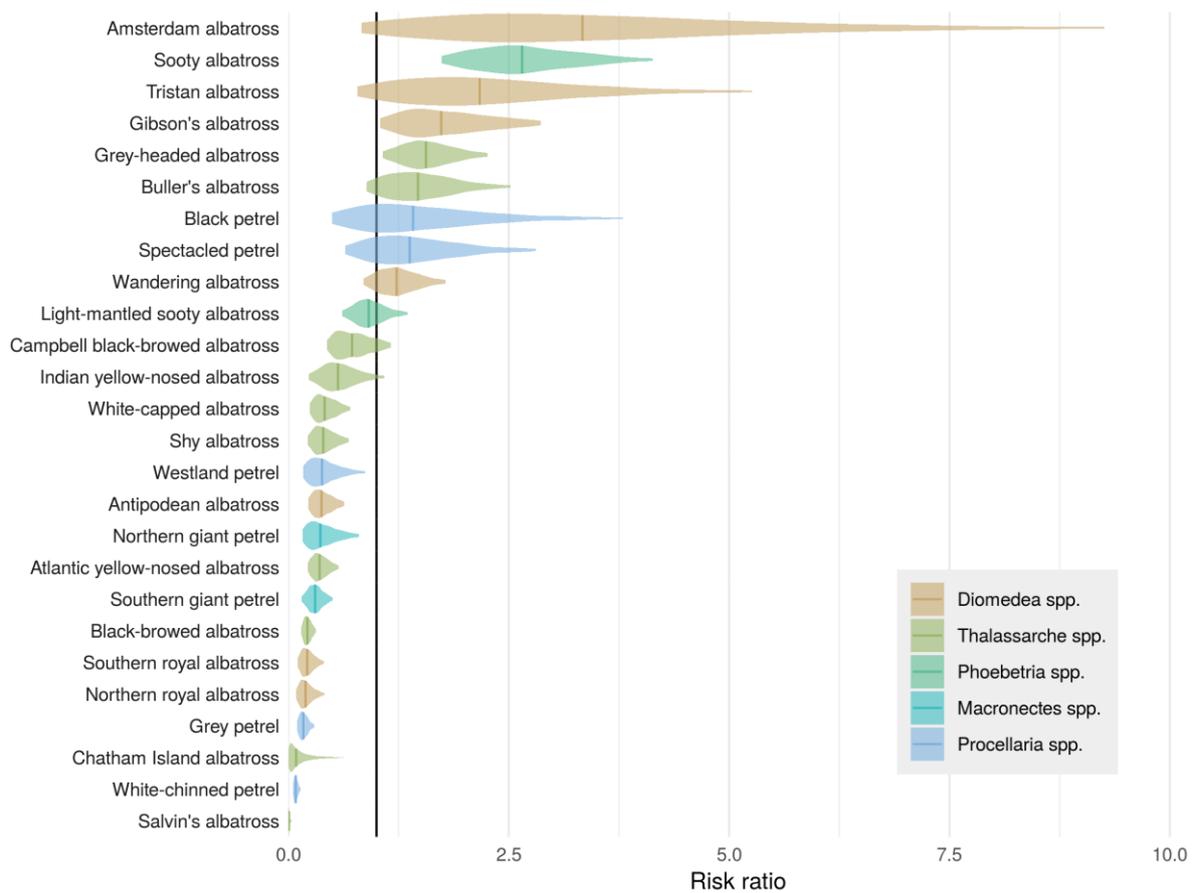
Figure 5. Risk ratio for the seabird taxa included in the current risk assessment (EAC/PST). The risk ratio is the ratio of the estimated annual captures (EAC) in Southern Hemisphere surface longline fisheries to the Population Sustainability Threshold (PST). No cryptic mortality was included and the potential survival of captured seabirds that were released alive was not considered. The black line indicates a risk ratio of 1. For each species, the distribution of the risk ratio within the 95% credible interval is indicated by the coloured shapes, with the vertical line indicating the median risk ratio (vertical line). Seabird species are listed in decreasing order of the median risk ratio.
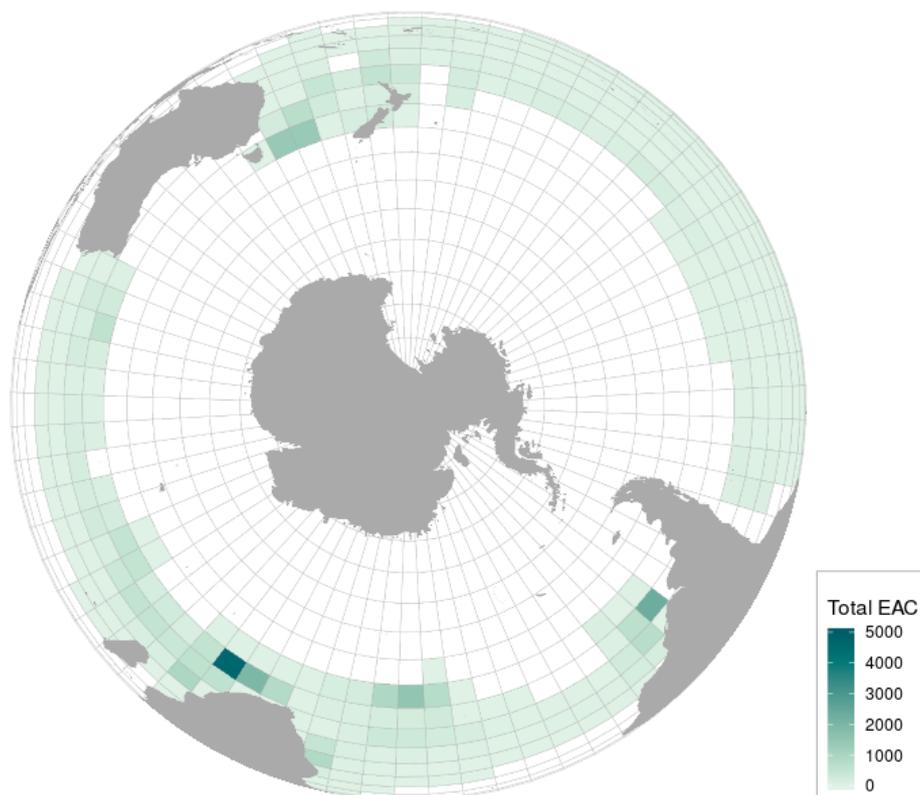
Figure 6. Total estimated captures for the seabird taxa included in the current risk assessment. For each 5-degree cell, the intensity of the colour shows the mean of the sum of Estimated Annual Captures, EAC (without including any cryptic mortality or survival of released seabirds).
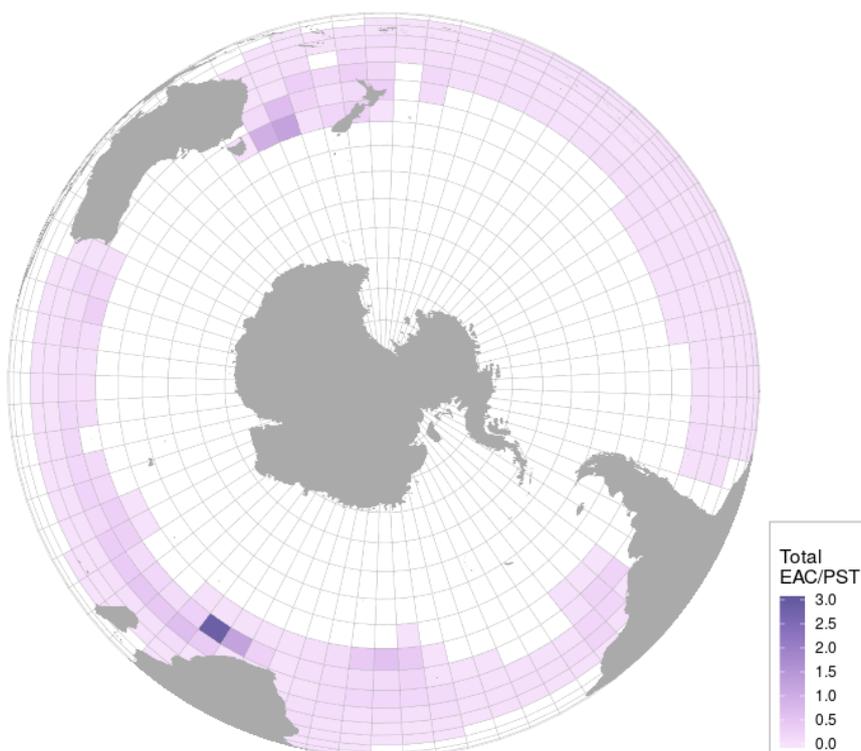
Figure 7. Aggregate risk (Estimated Annual Captures to Population Sustainability Threshold, EAC/PST) to seabirds from surface longline fishing in the Southern Hemisphere. Risk is the summed risk ratio from each of the 26 seabird taxa included in this analysis, not allowing for any cryptic mortality or survival of released seabirds.