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Assessment of the risk of southern hemisphere surface longline fisheries to ACAP species

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SUMMARY

New Zealand has been utilising and refining a spatially explicit assessment of risk to seabirds from commercial fishing since 2009. New Zealand has been intending to extend the risk assessment framework applied to main fishing methods within the New Zealand Exclusive Economic Zone (EEZ) to a broader set of fisheries. This paper presents the progress on this to date, where the methodology has been applied to public tuna RFMO fishing data throughout the southern hemisphere for the 26 ACAP listed seabird species that breed in the southern hemisphere. This version of the risk assessment is updated from that presented to CCSBT in March 2017 due to the inclusion of effort data from north of 25 degrees S from WCPFC.

Across all the seabird species and surface longline fishing effort included in this study, the total estimated annual potential fatalities were 6275 (95% c.i.: 4918–8054). The risk ratio is defined as the ratio of the annual potential fatalities to the population sustainability threshold (PST), a measure of the productivity of seabird species. Black petrel had the highest risk ratio, followed by three wandering-type albatross species (Tristan albatross, Amsterdam albatross, and Antipodean albatross). All other seabird species had risk ratios that were entirely less than one.

This paper is a report on progress, and the data used in this initial iteration contain some deficiencies. In particular, the vulnerability of seabirds to capture was estimated using New Zealand data only; the seabird distributions were simplistic; and effort data that was not reported to RFMOs was not included. The analysis can readily be updated however, if improved data become available.

1. INTRODUCTION

New Zealand has been utilising and refining a spatially explicit assessment of risk to seabirds from commercial fishing since 2009. The overall framework is described by Sharp et al (2011), and variously applied and improved in multiple iterations for seabirds (Waugh et al 2008a, b, developed further by Sharp 2009, Waugh & Filippi 2009, Filippi et al 2010, Richard et al 2011, Richard & Abraham 2013, Richard & Abraham 2017).

As noted at CCSBT ERSWG 10 and 11, New Zealand has been intending to extend the risk assessment framework applied to main fishing methods within the New Zealand Exclusive Economic Zone (EEZ) to a broader set of fisheries as seabirds migrate widely and interact with a wide range of fisheries across multiple EEZ and RFMO. This paper presents the progress on this to date, where the methodology has been applied to public tuna RFMO fishing data throughout the southern hemisphere for the 26 ACAP listed seabird species that breed in the southern hemisphere (Table 1). The intention for this risk assessment is to undertake a second iteration later in 2017 including additional data for tuna RFMOs and other fishing methods operating within the southern hemisphere.

Previous risk assessment approaches have been applied to tuna RFMO fisheries using spatial overlap and Productivity-Susceptibility Analyses (PSA) (Tuck et al 2011, Waugh et al 2012, 2013). Potential

Biological Removal type approaches have been applied within EEZ fishing including surface longline fishing (Dillingham & Fletcher 2008, 2011, Richard et al 2011, Richard & Abraham 2013, Richard & Abraham 2017). The strength of the PBR type approaches is that seabird captures are estimated and compared to population sustainability thresholds, and the results can be disaggregated or aggregated readily to the desired scale. It has the advantage of being fully quantitative: the ratio is a direct comparison between birds killed and the number of birds that can be produced by the population.

Table 1: The species included in the current study. These are the 26 species listed by the Agreement for the Conservation of Albatrosses and Petrels (ACAP) that have breeding colonies in the southern hemisphere. The species are grouped together for the purpose of estimating their vulnerability to capture in fisheries.

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

2. METHODS

2.1 Risk assessment framework

2.1.1 Risk ratio

The methodology used for estimating the risk follows 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, Sharp et al 2013, Sharp 2016).

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

$$RR = APF/PST,$$

where annual fatalities of less than the PST allow for long-term management goals to be achieved. Uncertainty is carried through all parameters in the calculation, so there is uncertainty in the resulting risk ratio. The APFs are estimated from a combination of observer effort data, observed bycatch, seabird distribution data, and fisheries effort data. Essentially, they are 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 Potential Biological Removals (PBR), used to estimate the productivity of marine mammal populations (Wade 1998).

The use of this risk methodology has developed out of the productivity-susceptibility approach that has previously been used (Waugh et al 2011, 2012). It has the advantage of being fully quantitative: the ratio is a direct comparison between birds killed and the number of birds that can be produced by the population.

2.1.2 Population Sustainability Threshold

The Population Sustainability Threshold (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 the Ministry for Primary Industries for the New Zealand seabird risk assessment is that populations are above half the carrying capacity (with 95% certainty), after 200 years. The PST is defined as:

$$PST = \frac{1}{2} \nu r_{max} N,$$

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. 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 Population Sustainability Threshold was derived for New Zealand seabirds (most recently by Richard et al 2017), based on the Potential Biological Removal (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 . However, the default value of the recovery factor for non-endangered population is also 0.5. In the calibration of the PST used here, the threat status of the population is not considered.

2.1.3 Estimating annual potential fatalities

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 (s) and the fishing effort within a group of fisheries (g) was calculated by summing the product of fishing intensity, population size, and the relative density of a species at the location of the fishing:

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

$$O_{sg} = \sum_i p_{si} a_{gi},$$

where N_s is the total population size, O_{sg} 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, and a_{gi} is the number of hooks associated with the fishing event.

Captures of seabirds are recorded by observers when they are on board fishing vessels. The expected number of incidents is assumed to be proportional to the density overlap. In its simplest form, the number of observable capture recorded by observers (C'_{sg}) is then given by:

$$C'_{sg} \sim \text{Poisson}(p_{\text{observable}} q_{sg} \theta'_{sg}),$$

where q_{sg} is the vulnerability of a species, s , to capture in a fisheries group, g , per unit of density overlap, θ'_{sg} . Here the prime symbol is used to indicate observed quantities. The probability, $p_{\text{observable}}$, is the probability that an incident that occurred while an observer was on the vessel would be recorded, for example because it fell off the hook before being brought on board. In a study by Brothers (2010), the interaction of seabirds with surface longline fishing vessels was recorded from 1988 and 2003, in various fishing regions. From the observation of almost 6000 birds taking baits during fishing events, 176 birds were seen caught, but only 85 of them were retrieved on board. From these figures, a binomial error was calculated, which led to a mean estimate of the probability of retrieving a captured bird of 0.48, with a 95% c.i. of 0.41-0.55.

In this initial analysis, we assumed that all the fishing effort had equivalent vulnerability (a species with the same density would be caught at the same rate in all fisheries), and so all the fishing effort was in the same group. In general, however, fishing effort could be assigned to different groups based on the fleet, provided there was enough observer information available to estimate the vulnerability.

In the initial analysis, the observer data that was used was from New Zealand. Not all the seabird species in the study occur within the New Zealand region, and so we placed the seabirds in groups (see Table 1), with the birds in the same group assumed to have the same vulnerability.

From the observer data, we separately estimated the live and dead capture of seabirds within each vulnerability group:

$$C'_{sga} \sim \text{Poisson}(p_{\text{observable}} p_{\text{alive}} q_{sg} \theta'_{sg}),$$

$$C'_{sgd} \sim \text{Poisson}(p_{\text{observable}} (1 - p_{\text{alive}}) q_{sg} \theta'_{sg}),$$

where the index a indicates captures of alive birds, and d indicates dead captures. The alive-capture probability, p_{alive} , was estimated separately for each species group.

The model was fitted to the data and estimated using Bayesian methods. 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 (2010). The model was fitted using two MCMC chains, with a burn-in of 10,000 iterations, and posteriors calculated from 800,000 further iterations, retaining a sample value every 400 iterations. Convergence and mixing were visually checked from the MCMC trace of the parameters.

Having fitted the model, the number of seabird fatalities of species s in fishing effort in the fishing group g could be estimated from the fitted vulnerability and the overlap as:

$$APF \sim \text{Poisson}((1 - p_{alive}p_{survive})q_{sg}\theta_{sg}),$$

where an additional parameter, the survival probability $p_{survive}$, is needed to specify the survival of live-released birds. There is no information available to inform this probability, and so we simply assume that it is uniformly distributed between 0 and 1. This prior implies that, on average, half of the live-released birds survive. Note that whether capture events are observable or not (whether caught birds are retrieved on board the vessel), does not affect the APF, and so the probability $p_{observable}$ does not appear in this expression.

2.2 Data used for assessing risk

2.2.1 Fishing effort

Publically available fishing effort data were obtained from Regional Fisheries Management Organisations (RFMOs), with the number of hooks set by country flag, within each month between 2004 and 2014, in 5 degree latitude and longitude cells. The data were obtained from the CCSBT, IATTC, ICCAT, IOTC, and WCPFC through publicly available websites. The data that are made publicly available were typically truncated in some way (such as not reporting effort in cells with fewer than three vessels fishing), and some adjustments were required. The WCPFC data south of 25 S were scaled up to match totals reported at a higher level of aggregation (so that the total number of hooks is correct, although the spatial distribution may not be); WCPFC data north of 25 S was not scaled in this way (the differences were due to different timings of the extracts); a small amount of IOTC data were reported by days, rather than hooks, and these were omitted; some ICCAT data was omitted following a three vessel rule, however no total effort was available and so these data could not be scaled up; some CCSBT effort was only reported when there was a catch of southern bluefin tuna. The data reported to the other RFMOs were used to scale up the CCSBT effort to account for this unreported effort.

For the current analysis, fishing effort was taken over the three year period from 2012 to 2014 and was used to derive an average effort per month, per 5 degree cell, per RFMO. The annual sum of the monthly-averaged effort data are illustrated in Figure 1. The highest fishing intensity was in the western tropical Pacific, where some 5 degree cells had over 15 million hooks set per year.

Over this three year period, the total annual average number of hooks included in the dataset was 1579 million. This only includes surface longline fishing that was reported to RFMOs, and does not include all fishing within southern hemisphere Exclusive Economic Zones (EEZs).

2.2.2 Observer data

Observer data, including the number of hooks observed and the number of seabirds reported as bycatch, was obtained from the New Zealand Ministry for Primary Industries. The data were reported in the same resolution as the effort data (number of hooks per 5 degree cell, per month). Data were used for the eleven year period 2004 to 2014, covering the same period that the effort data were extracted. Over this period, there were Japanese and New Zealand flagged vessels operating in the New Zealand EEZ that carried observers. The Japanese and New Zealand vessels were not treated differently. The overlap between the fishing effort and each of the species groups (Table 2) was then used to fit the model used for estimating the APFs.

Seabird captures reported by observers were classified as dead or alive, and were identified to the species level, where possible. A small number of captures that could not be identified to the level of the vulnerability group were not included in the analysis.

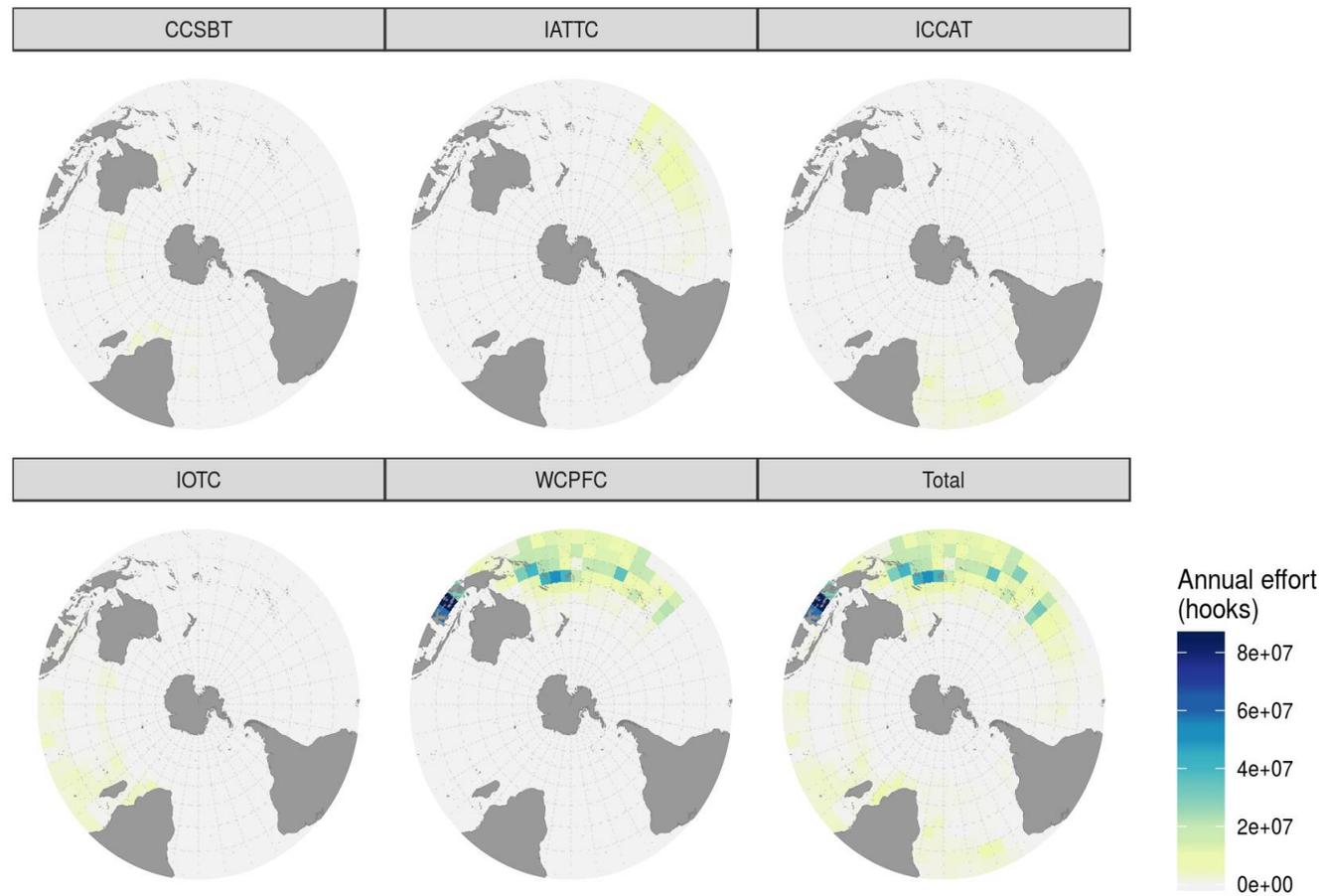


Figure 1: Fishing effort included in the assessment, as obtained from each RFMO, and in total. The colour of each 5 degree cell represents the annual fishing effort within the cell, averaged over the three year period 2012 to 2014.

Table 2: Observer data used for fitting the model. The observations were made in surface longline fisheries in the New Zealand region between 2004 and 2014. For each species group that overlapped with these fisheries, the table gives the number of observed captures (total, alive, and dead), and the overlap of the observed fishing effort with the seabirds in the species group.

Species	Captures	Alive	Dead	Overlap
Buller's albatross	331	137	194	57 209
Shy albatross	121	14	107	268 531
Wandering albatross	69	23	46	11 853
Grey petrel	44	0	44	95 870
Black-browed albatross	35	5	30	182 609
White-chinned petrel	34	1	33	631 806
Black petrel	22	11	11	1 349
Salvin's albatross	8	1	7	35 007
Westland petrel	7	0	7	12 971
Royal albatrosses	5	2	3	28 211
Giant petrels	2	0	2	21 159
Sooty albatross	1	0	1	7 690
Chatham Is. albatross	0	0	0	2 603
Grey-headed albatross	0	0	0	43 559

2.2.3 Seabird distributions and populations

Seabird distributions were made by combined range and colony information. For each species, the range was obtained from the American Bird Conservancy (ABC), via their fisheries and seabirds project (<http://www.fisheryandseabird.info/>, version 2015.07). For each species, colony information was obtained from the Agreement for the Conservation of Albatrosses and Petrels (ACAP), through their science officer. For each species this gave a list of colonies and their populations. For consistency, we took the total number of breeding pairs of each species to be the total of the number at each colony. A breeding start and end month was also defined for each species.

From the total number of breeding pairs, a total population was derived by scaling up by a multiplier that accounts both for the proportion of adults breeding and for the proportion of the population that are adult. The proportion of the population that was adult was derived from a simple population model that takes account of the age at first breeding and the adult survival of each species (Richard et al 2013).

Outside of the breeding season, the relative distribution was defined as a uniform distribution throughout the range of each species: a constant relative abundance was assumed, which integrated to one over the range of the species. During the breeding season, the juvenile and adult non-breeding birds were distributed uniformly through the range, while the breeders were distributed within a radius of each colony. This followed the method used by Waugh et al (2013). The density of breeders associated with a colony was assumed to be given by:

$$d = \exp^{\ln(0.01)r/R}$$

where r is the distance from the colony (km) and R is a maximum radius (which we took to be 1,500 km for all the species - note that Waugh et al (2013) used a different distance for each species, and different distances could be derived for each species if tracking data were available). With this expression, the density of birds associated to a colony falls away to 1% at 1,500 km from each colony. Any land was then excluded from the distributions and they were normalised so that they integrated to the number of

breeding adults at the colony. The distributions from all colonies of each species were combined, and were then normalised to integrate to one. A distribution was derived by combining the distributions for the non-breeders and the breeders.

The species richness (number of species occurring within each 5 degree cell) of the 26 ACAP species included in this study is highest between 40 and 50 degrees south, with the highest richness occurring near New Zealand and South Africa (Figure 2).

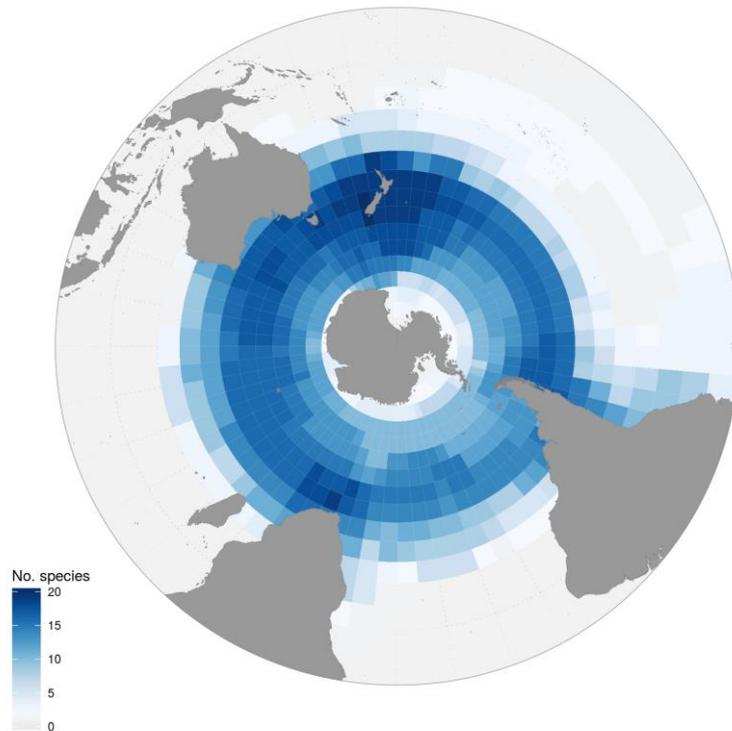


Figure 2: Species richness of the ACAP seabirds. For each 5 degree cell, the intensity of the colour is related to the number of species that occur within that cell. The species richness shows a strong peak at 40 to 50 degrees south, with the highest richness being close to New Zealand and South Africa.

2.2.4 Demographic data

The values of the demographic parameters for the 26 species 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, proportion of adults breeding, and body mass, and values of similar species were used for survival and age at first breeding when no value was available. 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 (2015). To obtain optimal values of adult annual survival rate and age at first breeding for the calculation of r_{max} , a taxonomic and allometric model was fitted to the values of 35 taxa for adult annual survival and 48 species for age at first breeding, as a function of body mass, 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 was found to underestimate the total population size because juvenile survival is generally lower than that of adults (Richard & Abraham 2013). The correction factor μ accounts for this bias, and was estimated from simulations of population dynamics, taking values between 1.45 for black petrel and 1.77 for Antipodean albatross (Richard & Abraham 2013).

The input demographic parameters, and the derived demographic parameters used for the calculation the PST are given in Annex A (Tables A-1, A-2, and A-3).

3. RESULTS

3.1 Vulnerability

As part of fitting the model, the vulnerability of each species group to capture in surface long-line fisheries was estimated, based on the New Zealand longline fisheries (Table 3). The highest vulnerability was of black petrel, followed by wandering albatross and Buller's albatross. The vulnerability of other species was an order of magnitude less. Species with vulnerabilities of two orders of magnitude less than black petrel included Salvin's albatross, royal albatross, black-browed albatross, giant petrels, white-chinned petrel, and grey-headed albatross.

The vulnerability is affected by any errors in either the population size or the bird distributions, and it is likely that the wide variation in vulnerability between closely related species reflects unrealistic distributions. Birds being caught where their distribution was assumed to be low would cause the vulnerability to be higher, and conversely, birds being caught where their distribution was high would cause the vulnerability to be lower.

These errors are exacerbated by the vulnerability only being estimated in the New Zealand region, as the region of overlap with New Zealand surface longline fisheries is small relative to the distribution of the birds. Any errors in the estimation of the vulnerability caused by the distribution being incorrectly specified around New Zealand would then be extrapolated across the southern hemisphere.

Table 3: Estimated vulnerability to capture of the different species groups. The vulnerability is the estimated bycatch rate, normalised by the bird abundance, and has units of birds caught per hook per unit bird abundance. The species groups are defined in Table 1.

Species group	Mean	95% c.i.
Black petrel	0.0425	0.0257–0.0640
Wandering albatross	0.0148	0.0108–0.0198
Buller's albatross	0.0145	0.0117–0.0181
Westland petrel	0.0017	0.0007–0.0031
Chatham Is. albatross	0.0016	0.0002–0.0047
Grey petrel	0.0012	0.0008–0.0016
Shy albatross	0.0011	0.0009–0.0015
Sooty albatross	0.0009	0.0002–0.0023
Salvin's albatross	0.0007	0.0003–0.0012
Royal albatross	0.0006	0.0002–0.0011
Black-browed albatross	0.0005	0.0003–0.0007
Giant petrels	0.0005	0.0001–0.0010
White-chinned petrel	0.0001	0.0001–0.0002
Grey-headed albatross	0.0001	0.0000–0.0003

3.2 Risk assessment

Across all the seabird species and surface longline fishing effort included in this study, the total estimated annual potential fatalities (APFs) were 6280 (95% c.i.: 4920–8050) (Table 4). The highest estimated APFs were of Buller's albatross, with annual potential fatalities of 1110 (95% c.i.: 752–1560), followed by black-browed albatross and white-capped albatross, with APFs of 1060 (95% c.i.: 679–1540) and 631 (95% c.i.: 464–832) respectively. Some species have very low APFs, for example Amsterdam albatross has an estimated APF of 8 (95% c.i.: 2–14) birds per year.

The APFs should be interpreted relative to the PSTs, however, and these vary by four orders of magnitude, from 6 (95% c.i. 4–10) for Amsterdam albatross, to 93800 (95% c.i.: 63500–142000) for white-chinned petrel (Table 4). The APFs for black petrel entirely exceed the PST (Table 4, Figure 4). With the assumed population size of black petrel, the fatalities in southern hemisphere surface longline fisheries are higher than the sustainable threshold. In this case, the population was taken from the ACAP colony database, and was 960 breeding pairs. This reflects a single count from the breeding colony, while black petrel are known to breed outside of the area that was surveyed. Taking account of a range of sources of information (including at sea surveys and band recovery data), the New Zealand risk assessment used an estimated 2750 (95% c.i.: 1600–5120) breeding pairs for black petrel (Abraham et al, 2015). With a population that was approximately three times higher, the uncertainty of the risk ratio would still exceed one. The assumed distribution of black petrel is partly in the northern hemisphere, outside of our study region, and there may be surface longline fishing within the EEZs of South and Central American countries that overlaps with the distribution of black petrel, but that is not included here.

Table 4. Risk assessment results for each of the 26 ACAP-listed seabird species included in this study, the table gives the mean, or median, and 95% credible interval of the Annual Potential Fatalities (APF), the Population Sustainability Threshold (PST), and the risk ratio. A risk ratio of over one indicates that the APFs are higher than the PST. The table is sorted in order of decreasing median risk ratio. The total APFs of all assessed species are also given.

Species	APF		PST		Risk ratio	
	Mean	95% c.i.	Mean	95% c.i.	Median	95% c.i.
Black petrel	1 000	475–1 700	98	76–127	7.42	3.53–13.81
Tristan albatross	250	158–361	180	92–413	1.64	0.62–3.30
Amsterdam albatross	8	2–14	6	4–10	1.39	0.38–3.20
Antipodean albatross	328	207–475	519	266–929	0.99	0.59–1.63
Buller's albatross	1 110	752–1 560	2 980	1 480–5 400	0.56	0.30–0.99
Wandering albatross	347	219–499	731	534–1 000	0.51	0.31–0.84
Chatham Island albatross	100	9–316	1 510	1 110–2 220	0.19	0.02–0.80
Shy albatross	136	96–181	445	322–641	0.1	0.06–0.15
Westland petrel	14	4–28	4 700	2 210–8 960	0.07	0.02–0.16
Atlantic yellow-nosed albatross	213	134–314	264	153–433	0.07	0.04–0.13
White-capped albatross	631	464–832	11 500	8 260–16 700	0.06	0.04–0.09
Indian yellow-nosed albatross	190	117–283	4 600	2 530–10 300	0.05	0.02–0.10
Salvin's albatross	93	40–172	967	669–1 360	0.03	0.01–0.05
Grey petrel	226	150–321	9 090	5 390–14 300	0.03	0.02–0.05
Sooty albatross	33	5–86	3 490	2 640–5 010	0.03	0.01–0.11
Black-browed albatross	1 060	679–1 540	57 800	40 800–80 600	0.02	0.01–0.03
Northern giant petrel	20	4–48	1 400	644–2 970	0.02	0.00–0.05
Campbell b.-browed albatross	32	17–51	2 090	1 530–2 910	0.02	0.01–0.03
Southern royal albatross	7	1–16	1 630	940–2 620	0.01	0.00–0.02
Northern royal albatross	6	1–15	5 940	3 700–10 300	0.01	0.00–0.02
Southern giant petrel	56	13–130	730	497–1 060	0.01	0.00–0.03
Spectacled petrel	15	7–25	918	688–1 220	0.01	0.01–0.02
Grey-headed albatross	11	1–35	93 800	63 500–142 000	0	0.00–0.00
Light-mantled sooty albatross	4	0–13	1300	809–2 000	0	0.00–0.01
Waved albatross	1	0–3	10 600	7 230–15 400	0	0.00–0.00
White-chinned petrel	389	253–555	1 360	593–2 640	0	0.00–0.01
Total	6 280	4 920–8 050				

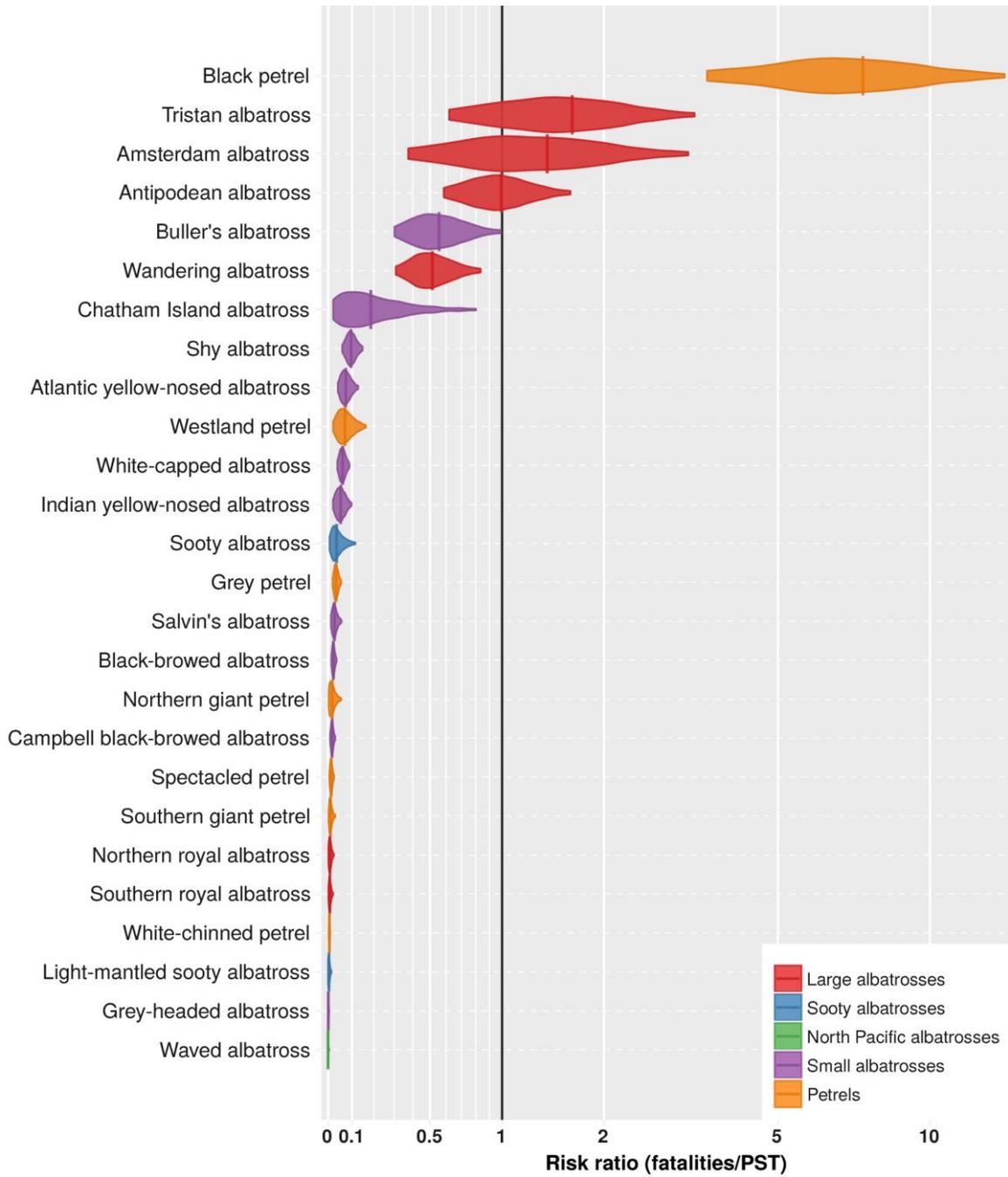


Figure 3: Risk ratio for the 26 ACAP seabird species considered. The risk ratio is the ratio of the annual potential fatalities in southern hemisphere fisheries to the Population Sustainability Threshold (PST). The risk ratio is displayed on a logarithmic scale, with the threshold of the number of potential bird fatalities equalling the Population Sustainability Threshold (PST) indicated by the vertical black line, and the distribution of the risk ratios within their 95% credible interval indicated by the coloured shapes, including the median risk ratio (vertical line). Seabird species are listed in decreasing order of the median risk ratio.

Following black petrel, the three species with the highest risk are wandering-type albatrosses (Tristan albatross, Amsterdam albatross, and Antipodean albatross). In each of these cases, the range of uncertainty in the risk ratio exceeds one, with the mean value exceeding one for Tristan albatross and Amsterdam albatross. The risk to wandering albatross is also high, but is less than one. Despite the different population sizes of these wandering albatross type species, the estimated risk is similar, reflecting the structural assumptions of the model that the bycatch rate is proportional to the overlap (which is itself related to the population size). From 2004, Antipodean albatross at Antipodes Island has experienced a marked decrease in the population within a census area at the breeding colony (Elliott and Walker 2014), associated with both a decrease in survivorship and in breeding success. Over the same period the population of the Gibson's albatross subspecies of Antipodean albatross at the Auckland Islands (*Diomedea antipodensis gibsoni*) has also decreased, while showing a similar decline in survivorship (Francis et al 2015). Tracking data shows the Antipodes Island breeding birds using waters to the east of New Zealand, and migrating to Chile during the non-breeding season, while the Auckland islands breeding birds forage to the south and west of New Zealand, and in the Tasman Sea. These tracking data could be used to help refine understanding of overlap with surface longline fisheries (see Figure 3).

The only other species with a risk ratio that is higher than 0.5 is Buller's albatross. Buller's albatross have been the most frequently observed caught species in New Zealand surface longline fisheries (see Table 2). In this study, these observed captures have then been used to estimate captures in surface longline fisheries in the Tasman Sea. An analysis of long-term banding data concluding that the population of Buller's albatross has shown long-term increase, and so surface longline fisheries were not a threat to the viability of this species (Francis & Sagar 2012). This is consistent with a risk ratio below one. The use of a more detailed spatial distribution, that included tracking data from the different breeding colonies and of different life stages, would help to resolve the extent of the interaction between Buller's albatross and these Tasman Sea fisheries.

An advantage of the risk assessment method is that the APFs may be estimated spatially (Figure 3), at the same resolution as the fishing effort data. When viewed spatially, the current model predicts that the highest captures are within the sub-tropical Pacific and Tasman Sea regions, and on the east coast of South Africa. This distribution reflects both the occurrence of birds with high vulnerability to capture, and of high fishing effort. The summed risk, across all species, is highest in the sub-tropical Pacific (Figure 5), along the migratory route of black petrel. There is also high risk on the east coast of South Africa. It is likely that the method has overestimated the risk to black petrel in this region, as the seabird distributions were not seasonal.

When viewed by RFMO, the estimated APFs are highest on fishing effort reported to CCSBT (Table 5). The captures are most intense on fisheries in the Tasman Sea (see Figure 4), with captures also occurring on CCSBT fisheries across the southern Indian Ocean. It is possible that this conclusion will change if the data are improved, particularly if observer data is available from the relevant CCSBT fisheries.

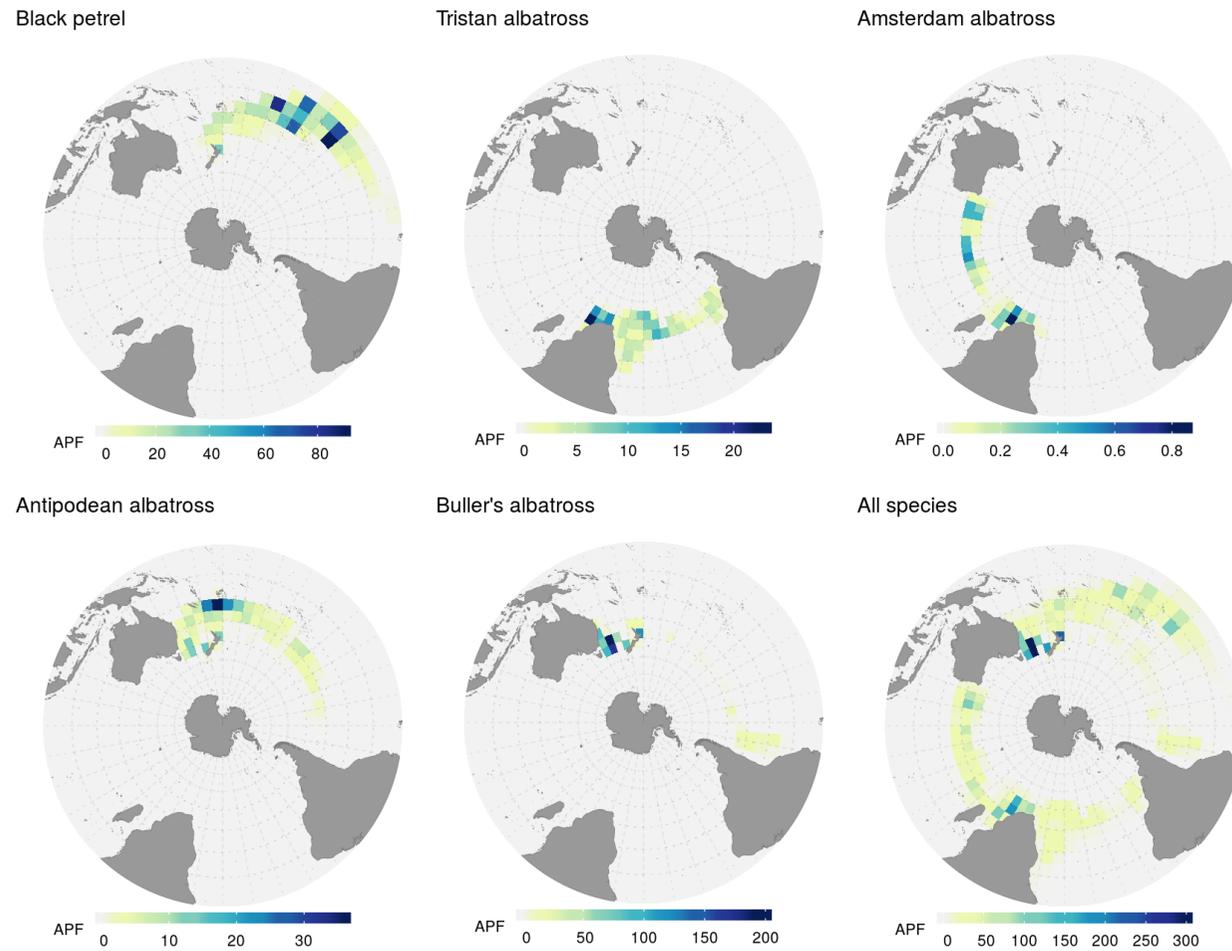


Figure 4: Annual potential fatalities for the five seabird species the most at risk from fisheries in the southern hemisphere, and for all 26 ACAP taxa considered in this study, within each 5 degree cell. The highest estimated annual potential fatalities occur within the Pacific, Tasman Sea, and eastern South African areas.

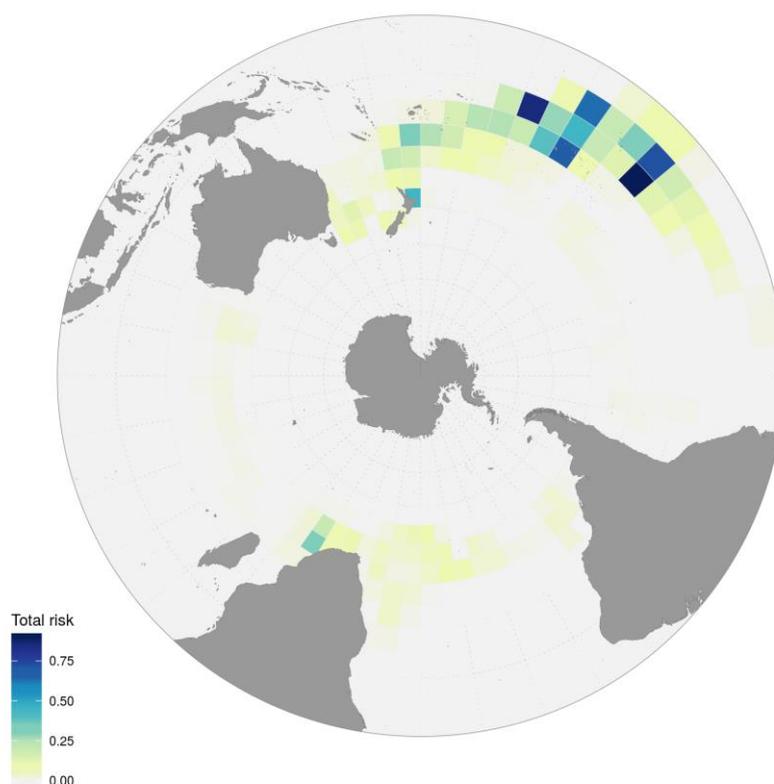


Figure 5. Aggregate total risk to all seabird species from surface longline fishing in the southern hemisphere. The risk is the summed risk ratio from each of the 26 ACAP seabird species, within each 5 degree cell.

Table 5. Breakdown of the Annual Potential Fatalities (APFs) by Regional Fisheries Management Organisation (RFMOs). The table gives the mean and credible interval of the APFs, summed over the 26 ACAP-listed seabird species included in this study, on effort data reported to each RFMO.

RFMO	APF	
	Mean	95% c.i.
CCSBT	2 088	1 644–2 648
WCPFC	1 905	1 380–2 614
IOTC	1 100	844–1424
ICCAT	662	494–875
IATTC	521	364–727
All RFMOs	6 275	4 918–8 054

3.3 Comparison with the New Zealand Seabird Risk Assessment

When the APFs estimated in this study were restricted to the New Zealand region, they were correlated with the APFs estimated in the New Zealand Seabird Risk Assessment (NZSRA; Richard & Abraham 2017) for surface longline fisheries. As would be expected, species that had higher APFs in one study tended to have higher APFs in the other study (Figure 6). For many species, however, the APFs estimated in this study, within the New Zealand region, were lower than the APFs estimated in the NZSRA. The biggest difference was for Campbell black-browed albatross - the mean APF of Campbell black-browed albatross in the New Zealand region was estimated to be 3 birds year (95% c.i.: 0–8) in this study, but was estimated to be 60 (95% c.i.: 32–97) birds per year in surface longline fisheries in the NZSRA. For Antipodean albatross, the mean APF in the New Zealand region was estimated to be 29 (95% c.i.: 15–46) birds per year in this study, but 216 (95% c.i.: 138–313) birds per year in the New Zealand region in the NZSRA.

Differences between the studies that may account for some of this variation include:

- Different species groups. For example, black-browed albatross was not included in the NZSRA, whereas Antipodean albatross was split into the Antipodean and Gibson’s subspecies in the NZSRA.
- A different resolution of the fishing effort. In the NZSRA, surface longline fishing was split by vessel size, at 45 m, to distinguish between the larger vessels that primarily targeted southern bluefin tuna, and the smaller vessel that primarily targeted bigeye tuna; fishing targeting swordfish was also treated separately. In this study, in contrast, all fishing effort was treated together.
- Different spatial resolution. The NZSRA had a resolution of around 2 km, with tracking data being used to inform seabird distributions where it was available. In contrast, this study was carried out with a 5-degree (around 500 km) resolution, and used a simple method for generating seabird distributions.
- A different time period. The NZSRA used observer data from the 2006–07 to 2014–15 fishing years (a New Zealand fishing year runs from October 1 to September 30), and estimated the APFs on fishing effort data from 2012–13 to 2014–15. In contrast this study used observer data from the 2004 to 2014 calendar years, and the APFs were estimated using effort from the 2012 to 2014 calendar years.

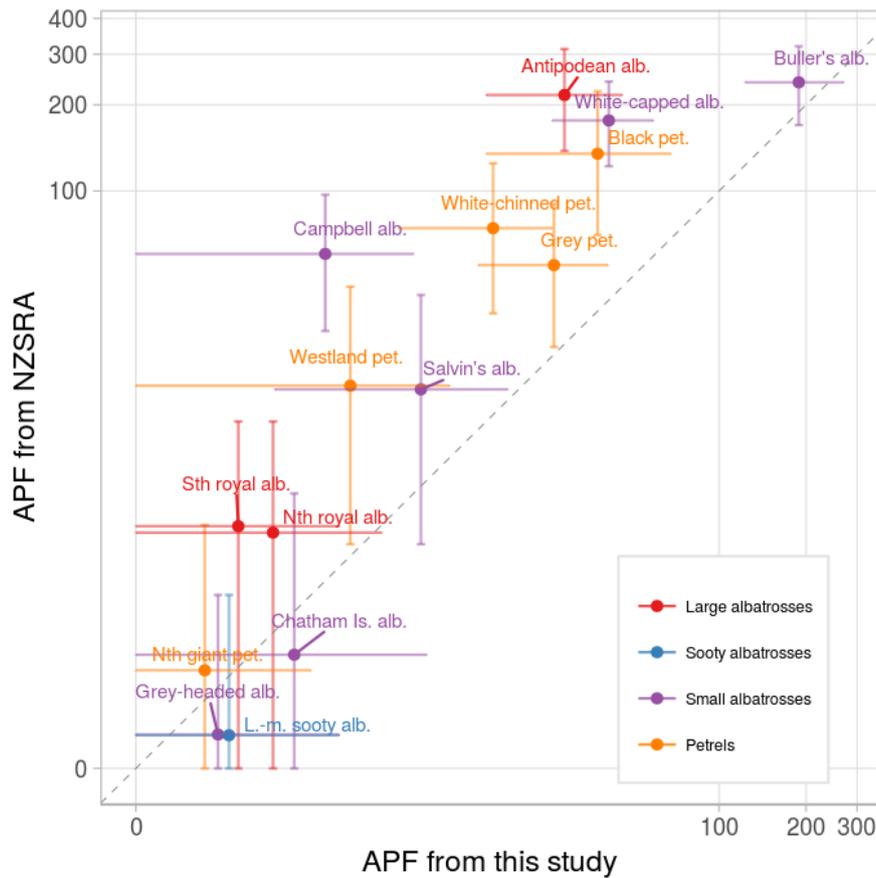


Figure 6: Comparison of the number of annual potential fatalities (APF) in surface longline fisheries in the New Zealand region, as estimated in this study and in the New Zealand seabird risk assessment (NZSRA; Richard & Abraham 2017). The APFs are displayed on a logarithmic scale. The diagonal dashed line shows the 1:1 relationship.

4. DISCUSSION

The risk assessment method is based on available data, using fisheries effort, observed bycatch, and seabird distribution information to estimate total captures of seabirds. From the total captures, the potential impact on each species can be estimated from demographic information. The risk assessment links the bycatch estimation to the demography of each species, allowing species to be identified where the bycatch may be in excess of what the seabird populations can sustain. Conversely, there are some species that may have a high fisheries related bycatch, but the bycatch is at a level that does not threaten the viability of the populations.

The key difference between the approach presented here and other risk assessment methods (such as the productivity-susceptibility analysis (Waugh et al 2012, 2013) and this approach is that this method aims to estimate absolute bycatch. There are many limitations with the data used in this analysis, and so the results should be treated as illustrative, rather being used for management at this stage. The total estimate of an annual fatality of around 5000 ACAP species is over an order of magnitude less than previous estimates (for example, Anderson et al (2011) estimated that at least 160 000 seabirds were killed annually in surface longline fisheries). There are many limitations to the data used in the current study. In particular:

Observer data. The observer data used to inform the vulnerability was from New Zealand only. Capture rates of seabirds may vary widely between fleets, depending on the mitigation used, whether hooks are

set below the depth range of seabirds, whether setting occurs during the day, and other factors that affect seabird capture rates. Ideally, a separate vulnerability would be estimated for each distinct fleet. This would depend either on the availability of observer reported bycatch rates. New Zealand has received positive indications of collaborations with both Japan and Australia to develop parallel assessments using their own effort and observer data. This collaboration approach looks to be a more successful manner in which to progress the refinement of the risk assessment. New Zealand would welcome further collaborations with other States.

Seabird distributions. The seabird distributions used an outer range and colonies. Monthly fisheries data have been used, and so monthly data on seabird distributions could also be used, if they were available. Tracking data are not complete for all breeding colonies or life stages, or for all species, and so analysis and interpretation is still required to derive the distributions. In the current analysis, transient areas (migratory routes) were over-represented and the use of range maps spreads birds away from fishing, potentially decreasing the estimated captures. Ideally, distributions would be available for each month, as the distribution will change depending on the breeding stage of the birds (e.g., Sztukowski 2017). Currently we are in discussion with Birdlife International to access seabird distributions from the Seabird Tracking data to include in the next iteration of the risk assessment.

Effort data. Effort data are not complete. Fishing effort was obtained from the RFMOs, however not all countries report fishing within their EEZs to the RFMOs. In addition, some of the species range into the northern hemisphere, especially into the north Pacific. Global data on surface longline effort is needed to fully complete the analysis.

Seabird demographic data. In this analysis we have used demographic data from ACAP. In some cases, this may need updating. For example, the high risk to black petrel is partly driven by the population estimate provided by ACAP of 960 breeding pairs. This is from a count of a single colony, and so is a minimum estimate. In the New Zealand risk assessment, higher values for the population of 2750 (95% c.i.: 1600–5120) breeding pairs were assumed (Abraham et al, 2015), based on integrating information from a range of sources beyond the colony count. Changes in the estimated population have a direct effect on the resulting risk score. The risk assessment drives a focus on the integrity of the input data, and this is a positive outcome of the process. In addition to the population data, the calculation of the PST also requires an estimate of the maximum population growth rate, under optimal conditions. This is poorly known.

Cryptic multipliers and survivability. The assessment attempts to estimate the total mortality. To achieve this, an estimate is included of cryptic mortality (the number of birds that would be killed by the fishing, but not recovered on board the vessel). This currently uses results from a single study by Brothers et al (2010). Information on cryptic mortality is inherently difficult to collect, however more information would help improve the accuracy of this estimate. We have also included a survivability parameter, which accounts for post-release survival of live-released birds. No information was available for this parameter and it was given a non-informative value (a probability uniformly distributed between zero and one).

A single fishing method only. The current analysis only includes surface longline data. However, it could be extended to other fishing methods. This would allow a direct comparison of the impacts of surface longline fishing to the impacts from other methods. The risk assessment then allows for a quantitative prioritisation across fishing methods. Currently New Zealand also holds public data for fishing under the CCAMLR and SPRFMO, and is investigating a collaboration with Australia to include all fishing methods within their EEZ to include in the next iteration of the risk assessment.

The ultimate goal of this work is a complete quantitative assessment across all pelagic seabirds and all fisheries. This would allow prioritisation of risk reduction across seabird species, and between fisheries or areas. In the short term, we plan to address some of the data limitations discussed above, improving the seabird distributions by including tracking data; obtaining access to more observer data; and improving completeness of the surface-longline effort data. This will allow us to develop, for the first time, a comprehensive analysis of the risk posed to pelagic seabird species from surface longline fisheries.

5. CONCLUSIONS

The risk assessment approach allows available data to be used to assess seabird bycatch at a hemispheric scale. The resulting bycatch and risk estimates may be reported spatially, or by RFMO, and so can be used to inform management responses.

This paper is a report on progress, and the data used contain some deficiencies. The analysis can readily be updated however, if improved data become available. New Zealand seeks further collaborations with other States and organisations to conduct a parallel assessment with their own data with results to be included in the wider risk assessment, or to supply data to improve the observer, effort, or seabird data.

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ANNEX 1: DEMOGRAPHIC PARAMETERS

Table A-1: Age at first breeding and adult annual survival rate of the 26 species of seabirds considered in this study. The values derived from the information gathered in the literature and the values obtained from a taxonomic and allometric model are shown.

Taxa	Age at first breeding				Adult annual survival rate			
	From literature		Modelled		From literature		Modelled	
Wandering albatross	10	7.2–12.9	11.3	9.7–13.1	0.953	0.94–0.963	0.956	0.945–0.966
Antipodean albatross	11.5	10.1–12.9	10.7	9.3–12.2	0.956	0.942–0.969	0.954	0.944–0.963
Southern royal albatross	9.5	8.6–10.5	10.8	9.3–12.3	0.948	0.931–0.963	0.954	0.943–0.963
Northern royal albatross	9.6	8.6–10.5	10.7	9.3–12.3	0.938	0.909–0.967	0.954	0.944–0.963
Tristan albatross	10	7.2–12.8	10.4	9.1–11.8	0.905	0.831–0.953	0.952	0.943–0.962
Amsterdam albatross	11.5	10.1–12.9	10.7	9.4–12.2	0.953	0.904–0.981	0.954	0.943–0.963
Waved albatross	8.5	6.1–10.9	8.8	7.9–9.7	0.926	0.921–0.93	0.944	0.934–0.952
Atlantic yellow-nosed albatross	9	6.1–11.8	8	7.3–8.7	0.92	0.899–0.938	0.938	0.928–0.947
Indian yellow-nosed albatross	8.9	6.1–11.8	8	7.2–8.7	0.925	0.841–0.971	0.938	0.928–0.948
Grey-headed albatross	10	7.2–12.8	9	8.1–9.9	0.952	0.931–0.968	0.945	0.936–0.953
Black-browed albatross	9	7.1–10.9	9	8.1–9.9	0.945	0.93–0.957	0.945	0.935–0.953
Campbell black-browed albatross	9.5	6.2–12.8	8.8	7.9–9.7	0.944	0.929–0.957	0.944	0.934–0.952
Buller's albatross	12	9.2–14.9	8.4	7.6–9.2	0.955	0.931–0.979	0.941	0.931–0.95
Shy albatross	12	9.1–14.9	9	8.1–10	0.959	0.934–0.975	0.945	0.936–0.954
White-capped albatross	12	9.2–14.8	9	8.1–10	0.959	0.935–0.975	0.945	0.936–0.954
Chatham Island albatross	12	9.2–14.8	9	8.1–10	0.966	0.941–0.982	0.945	0.936–0.954
Salvin's albatross	12	9.2–14.8	9	8.1–10	0.966	0.941–0.982	0.945	0.936–0.954
Sooty albatross	12	9.1–14.9	8.3	7.5–9.1	0.97	0.961–0.979	0.94	0.931–0.95
Light-mantled sooty albatross	12	9.1–14.8	8.4	7.6–9.2	0.97	0.96–0.98	0.941	0.932–0.95
Southern giant petrel	7.5	7–8	9.3	8.3–10.3	0.912	0.837–0.96	0.947	0.938–0.956
Northern giant petrel	8	6.1–9.9	9.2	8.3–10.3	0.886	0.811–0.961	0.946	0.937–0.955
White-chinned petrel	6.5	4.1–8.9	6.9	6.3–7.6	0.935	0.902–0.969	0.929	0.915–0.941
Westland petrel	6.5	4.1–8.9	6.8	6.2–7.4	0.947	0.919–0.974	0.927	0.914–0.94
Black petrel	6.6	6.2–7	6	5.4–6.6	0.926	0.899–0.948	0.917	0.898–0.935
Grey petrel	7	5.1–8.9	6.7	6.1–7.3	0.935	0.902–0.968	0.926	0.912–0.939
Spectacled petrel	6.5	4.1–8.9	6.7	6.2–7.3	0.946	0.919–0.974	0.926	0.912–0.939

Table A-2: Annual number of breeding pairs and proportion of adults breeding of the 26 species of seabirds considered in this study.

Taxa	Annual breeding pairs		Proportion of adults breeding	
	Mean	95% c.i.	Mean	95% c.i.
Wandering albatross	8 390	7 360–9660	0.747	0.641–0.836
Antipodean albatross	4 400	2 480–7240	0.600	0.499–0.693
Southern royal albatross	7 940	7 250–8680	0.599	0.503–0.692
Northern royal albatross	5 790	5 440–6150	0.611	0.51–0.702
Tristan albatross	1 110	1 010–1210	0.748	0.643–0.834
Amsterdam albatross	46	42–50	0.600	0.500–0.693
Waved albatross	10 400	4 970–18500	0.746	0.636–0.837
Atlantic yellow-nosed albatross	30 500	16 800–50500	0.746	0.638–0.835
Indian yellow-nosed albatross	29 600	25 200–35000	0.747	0.640–0.834
Grey-headed albatross	98 700	78 300–123000	0.746	0.641–0.834
Black-browed albatross	535 000	419 000–677 000	0.747	0.635–0.836
Campbell black-browed albatross	21 700	19 800–23 700	0.889	0.759–0.964
Buller's albatross	24 000	14 200–36 800	0.747	0.638–0.833
Shy albatross	13 900	12 800–14 900	0.747	0.642–0.835
White-capped albatross	96 000	88 000–105 000	0.679	0.574–0.769
Chatham Island albatross	5 250	4 790–5 740	0.890	0.751–0.965
Salvin's albatross	41 200	39 500–43 100	0.890	0.754–0.965
Sooty albatross	8 900	6 850–11 500	0.749	0.644–0.838
Light-mantled sooty albatross	9 720	6 730–13 500	0.601	0.500–0.695
Southern giant petrel	49 800	41 300–60 100	0.745	0.637–0.835
Northern giant petrel	10 300	8 290–12 600	0.889	0.752–0.964
White-chinned petrel	978 000	796 000–1 180 000	0.889	0.751–0.964
Westland petrel	2 910	1 920–4 160	0.890	0.754–0.964
Black petrel	1 070	949–1 210	0.795	0.689–0.882
Grey petrel	79 300	55 100–108 000	0.796	0.682–0.881
Spectacled petrel	14 800	9 780–21 200	0.747	0.639–0.835

Table A-3: The derived demographic parameters for each species. The population size is estimated from the number of breeding pairs, to account for non-breeding and juvenile birds, and the maximum growth rate is derived from adult survival and age at first breeding

Species	Total population size (N)		Growth rate (r_{max})	
	Mean	c.i	Mean	c.i.
Wandering albatross	62 100	46 600–83 200	0.047	0.041–0.053
Antipodean albatross	41 800	21 800–73 400	0.05	0.044–0.056
Southern royal albatross	74 400	57 500–96 900	0.049	0.043–0.056
Northern royal albatross	58 900	41 300–82 900	0.05	0.043–0.056
Tristan albatross	14 000	7 400–32 200	0.051	0.045–0.058
Amsterdam albatross	467	300–800	0.05	0.044–0.056
Waved albatross	88 700	38 900–170 900	0.061	0.056–0.067
Atlantic y.-nosed albatross	276 000	131 500–522 400	0.068	0.062–0.074
Indian y.-nosed albatross	270 000	150 200–608 800	0.068	0.062–0.074
Grey-headed albatross	704 000	487 800–1 012 800	0.06	0.054–0.066
Black-browed albatross	3 850 000	2 765 200–5 356 500	0.06	0.055–0.066
Campbell b.-b. albatross	136 000	101 400–186 800	0.062	0.056–0.067
Buller's albatross	184 000	91 300–330 400	0.065	0.059–0.071
Shy albatross	102 000	75 700–149 200	0.06	0.054–0.065
White-capped albatross	772 000	563 700–1 110 000	0.06	0.054–0.065
Chatham Island albatross	29 700	2 2100–42 400	0.06	0.054–0.066
Salvin's albatross	234 000	180 300–328 000	0.06	0.054–0.066
Sooty albatross	59 200	4 1500–82 600	0.065	0.060–0.071
L.-mantled sooty albatross	80 800	50 900–123 300	0.064	0.059–0.070
Southern giant petrel	411 000	258 200–711 700	0.058	0.052–0.064
Northern giant petrel	96 200	44 700–203 400	0.058	0.052–0.064
White-chinned petrel	4 730 000	3 234 300–7 088 900	0.079	0.072–0.087
Westland petrel	13 000	7 600–21 300	0.081	0.074–0.089
Black petrel	4 170	3 300–5 300	0.094	0.084–0.104
Grey petrel	440 000	261 100–693 000	0.083	0.075–0.090
Spectacled petrel	79 200	46 000–126 900	0.082	0.075–0.090