Preliminary assessment of the risk of albatrosses by longline fisheries

Daisuke Ochi[1], Edward Abraham[2], Yukiko Inoue[1], Kazuhiro Oshima[1], Nathan Walker[3] , Yvan Richard[2] and Sachiko Tsuji[1]

Abstract

This document presents the preliminary results of applying the Spatially Explicit Fisheries Risk Assessment framework (SEFRA) developed by New Zealand to assessing the total mortality of great albatross caused by tuna longline operations in the southern hemisphere. The impacts of these mortalities on the sustainability of these albatross species are also considered. Seabird bycatch was modelled as a multiplier of a temporal and spatial overlap between fishing and seabird distributions. Seabird catchability, defined as a combination of seabird-specific vulnerability to fishing gears and gear-specific seabird catchability, was first estimated using the Japanese and New Zealand on-board observer data, and then applied to the total fishing efforts in the southern hemisphere for assessment of total annual bycatch mortality (ABM). High ABMs and high expected risks of longline bycatch to seabird population were identified in particular for wandering albatross, among seven species examined, even though the results were considered preliminary. The analysis demonstrated the capacity of SEFRA to deliver robust estimates of ABM and their impacts to seabird population from limited observer bycatch data, although some constraints and uncertainties, in particular for rarely caught species, remained. The authors intend to further develop the methodology and invite interested WCPFC members to join the collaboration.

1. Introduction

A large proportion of albatross species currently have an endangered or threatened population status (e.g., Phillips et al 2016). Incidental captures of albatrosses by commercial tuna longline operations have been well recognized (e.g., Croxall 2012). The three Regional Fisheries Management Organizations (RFMOs) responsible for tuna fisheries in the Western and Central Pacific, Atlantic and Indian Oceans (WCPFC, ICCAT and IOTC) have established regulations for longline fisheries operating south of 25S in the ICCAT and IOTC Convention Areas and 30S in the WCPFC Convention Area that require the utilization of at least two of the following seabird mitigation measures: tori-poles, night-setting and weighted branch lines (ICCAT Rec. 11-09, IOTC Resolution 12/06 and WCPFC CMM 2017-06).

Despite the significance of albatross bycatch, the assessment of total seabird mortality caused by tuna longline operations has remained a challenge (Anderson 2011). This is due to the inadequacy of the available data and the lack of a comprehensive analytical methodology. Similar challenges prevent the evaluation of the effectiveness of mitigation measures.

---

[1] National Research Institute of Far Seas Fisheries, Japan

[2] Dragonfly Data Science, New Zealand

[3] Fisheries New Zealand, New Zealand

Seabird bycatch is typically a rare event, with large proportions of observed sets having zero bycatch. Even in areas and seasons with high seabird abundance, the observed seabird bycatch rate is highly variable, with a large difference in bycatch rates between vessels (e.g. Tsuji 2017). This variation probably reflects differences between the capacity of fishing masters to avoid seabird bycatch, and differences in gear configuration, fishing operations and use of bycatch mitigation measures. The large fluctuations of observed bycatch rates, and substantial vessel effects, make the use of standard statistical methodologies challenging.

ICCAT SCRS estimated that the level of observer coverage required to obtain reliable estimates of the bycatch of rare species, including seabirds, is above 85%. Using a simulation approach, Tsuji (2017) indicated that the coefficient of variation of estimated bycatch would be around 30%, under 50% observer coverage. The coverage of existing observer programs has been far below this, which indicates it is not realistic to aim for reliable estimates of seabird bycatch, solely from observer data.

There are two characteristics that make seabird bycatch by longline distinct from other species bycatch: i) the distribution and population abundance of seabirds are available independently, and ii) the seabird bycatch rate is significantly influenced by the type of mitigation measures used and the way they are implemented. The Spatially Explicit Fisheries Risk Assessment framework developed by New Zealand (SEFRA; Richard et al 2017, Richard et al 2013, Sharp et al 2013, Sharp 2016) succeeded in integrating independent seabird distribution and population abundance information in analyzing seabird-fisheries interactions. Seabird bycatch was modelled as a multiplier of a temporal and spatial overlap between fishing and seabird distributions (scaled by seabird populations to give an estimate of localised abundance). Seabird catchability was defined as a combination of seabird-specific vulnerability to fishing gears and gear-specific seabird catchability. Overlap between seabirds and fisheries has been used to assess interactions between seabirds and fisheries (e.g., Karpouzi et al 2007, Inoue et al 2011, Tuck et al 2011, Waugh et al 2011, Inoue et al 2012, Waugh et al 2012). The SEFRA method uses overlap to derive quantitative estimates of bycatch for seabird species.

This analysis explores the applicability of SEFRA to assessing the total mortality of great albatrosses (*Diomedea sp.*) caused by tuna longline operations in the southern hemisphere, and their impacts on seabird population sustainability. The potential use of this methodology in evaluating the effectiveness of mitigation measures is also explored in the discussion. The work presented here is preliminary, but will be the foundation of a more detailed analysis.

2. Method

2.1 General concept and model description

The overall scheme of the estimation procedure is shown in Figure 1. The basic concepts, the structure of bycatch model and the method used for estimating total seabird mortality remains, in principle, the same as described in the seabird risk assessment method developed by New Zealand (Abraham et al 2017a,b). In this paper, we focus on estimating the total annual bycatch mortality. We do not consider cryptic mortalities due, for example, to birds being caught during the set but falling off the hook before being retrieved on board the vessel (Brothers 2010).

The seabird bycatch estimate used data collected by on-board observers of Japan and New Zealand (NZ). Here, the bycatch is represented as the multiplicative interaction of three main components: the seabird population abundance; the overlap of seabird distribution with fishing effort; and seabird catchability. The analysis utilized quarter and 5x5 degree squares as the principle time-area strata. Within individual strata, this was equivalent to the formulation of CPUE, and assumes that the bycatch is proportional to both the catchability q and the local population n.

The bycatch estimation model also includes several additional factors to represent the data obtained from on-board observers (shown in the small box titled with grey background in Figure 1), i.e. the separation of live and dead captures, and the adjustment of bycatch records not identified to exact species. In general, the model also allows for the representation of cryptic mortalities that are not able to be recorded by observers.

In other words, the expected numbers of live birds λ_{live} and dead birds λ_{dead} of species s by fishing f are given as:

$$\lambda_{live} = q(s,f)N(s)O_e(s,f)p_{live}(f)p_{obs}p_{id}$$
$$\lambda_{dead} = q(s,f)N(s)O_e(s,f)\big(1-p_{live}(f)\big)p_{obs}p_{id}$$
$$\begin{cases} C_{alive}(s,f) \sim Poisson\big(\lambda_{live}(s,f)\big) \\ C_{dead}(s,f) \sim Poisson\big(\lambda_{dead}(s,f)\big) \end{cases}$$

where C and λ correspond to observed and expected number of bycatch, and $q, N, O_e, p_{live}, p_{obs},$ and p_{id} indicate catchability, total seabird population, overlaps between seabird and fishery distributions, proportion of live bycatch, proportion of bycatch detected by on-board observers, and proportion of bycatch number identified to exact species.

The catchability q was considered to be influenced by two factors, i) the susceptibility of species s caused by its behavior and ecological characteristics, and ii) the tendency of fishing group f to capture seabirds that may differ according to the gear configuration, operation procedures, and/or bycatch mitigation used.

Two different approaches were examined. The first one (Scenario 1) defined q as a multiplier of species-specific susceptibility $q(s)$, fisheries-specific seabird catchability $q(f)$, and their interaction as:

$$q(s,f) = q_s(s)q_f(f)q_{s:f}(s,f)$$

A variation (Scenario 2) was to assume that the species-specific susceptibility was different between fisheries:

$$q(s,f) = q_f(f)q_s(s|f)$$

The population number N was given as offset in the model. However, a sensitivity test was conducted, using the demographic information as a prior, but estimating N within the model. This allowed consistency between the model assumptions and population data to be checked. Parameterisations of the catchability that lead to a large discrepancy between prior and posterior were considered to be unreliable and were not explored further.

The spatial and seasonal overlap of seabird and fishery distributions O_e within a 5x5 degree latitude-longitude grid cell a and quarter t for fishery group f and species s was given as an offset:

$$O_e(s,f) = \sum_{a,t} E_e(f,a,t)D(s,a,t)$$

with corresponding observed fishing effort E_e and relative seabird distribution density D.

Using observer data, it is not possible to estimate the cryptic mortalities (the proportion of potential mortality caused by interaction with fisheries, but not detected by on-board observers). During this preliminary analysis, we did not consider cryptic mortalities, and assumed p_{obs} = 1 as a base case. A sensitivity run was carried out by using p_{obs} = 0.48 (95% c.i.0.41–0.55), based on an empirical estimate of cryptic mortality, derived by comparing seabird attacks on hooks behind the vessel with actual bycatch (Brothers 2010).

The probability that a seabird would be released alive, p_{live} was an estimated parameter, and was estimated for each fishery from the observer data.

The identification probability p_{id} was estimated from the observed number of bycatch not identified to exact species U and their expected number λ as follows:

$$\lambda_{unid,live} = (1 - p_{id}) \sum_{s,f} q(s,f)N(s)O_e(s,f)p_{live}(f)p_{obs}$$

$$\lambda_{unid,dead} = (1 - p_{id}) \sum_{s,f} q(s,f)N(s)O_e(s,f)\big(1 - p_{live}(f)\big)p_{obs}$$

$$\begin{cases} U_{live} \sim Poisson(\lambda_{unid,live}) \\ U_{dead} \sim Poisson(\lambda_{unid,dead}) \end{cases}$$

The model parameters of the bycatch explanation model were estimated with Bayesian estimation procedures with MCMC algorithm (NUTS sampler) using Stan (Stan 2.17.0; mc-stan.org). The MCMC was conducted with three chains of 12,000 iterations. To reduce any initial value effects, the values from the last 2,000 iterations (i.e., a total of 6,000 samples), were used as the stationary posterior distributions.

From the converged model, the total Annual Potential Fatalities (APF) of species s and fishery group f was calculated using the posterior distribution of estimated parameters (shown with hat) and the overlap of total effort distribution and seabird distribution O_p as:

$$\hat{\lambda}_{APF}(s,f,a) = \hat{q}(s,f)\hat{N}(s)O_p(s,f,a)\big(1 - \hat{p}_{live}(f)\big)$$

$$APF(s,f,a) = Poisson\left(\hat{\lambda}_{APF}(s,f,a)\right)$$

When the probability p_{obs} is assumed to be one, then the APFs reduce to an estimate of the total Annual Bycatch Mortality (ABM).

2.2 Data used

The analysis examined the bycatch of 7 species of great albatrosses: wandering albatross *Diomedea exulans*, Antipodean albatross *D. antipodensis*, Gibson's albatross *D. gibsoni*, Tristan albatross *D. dabbenena*, Amsterdam albatross *D. amsterdamensis*, southern royal albatross *D. epomophora*, and northern royal albatross *D. sanfordi*. The basic biological parameters of these species utilized in this study are shown in Table 1 (see Abraham et al 2017a for the derivation of these parameters).

The species distribution maps were developed by combining 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/). Colony information of each species was obtained from the Agreement for the Conservation of Albatrosses and Petrels (ACAP) through its science officer. The ABC range maps were augmented by including breeding seabirds around the colony, with the breeding seabirds assumed to be within 1500 km of the colony (following the methods described in Abraham et al 2017a).

To estimate a catchability integrating both species-specific susceptibility and fisheries-specific seabird catchability $q(s,f)$, the on-board observer data on pelagic longline fisheries collected by Japan (Inoue et al 2012) and New Zealand during the years 2003 to 2016 was used. The dataset contained number of seabirds caught by species, disaggregated by capture status between live and dead, the number of hooks observed (observed fishing effort), and the time and location of observed fishing. The data were aggregated by quarter and by 5 degrees of latitude and longitude.

New Zealand observer data were also disaggregated into the domestic pelagic longline operations (referred to as "New Zealand") and those by the Japanese flag operating within the NZ EEZ (referred to as "Japan (Inside NZ)"). As a result, three fishing groups were defined as "New Zealand" – NZ domestic pelagic longline, "Japan (Inside NZ)" – Japan's fleet operations within NZ EEZ, and "Japan (Outside NZ)" – Japan's fleet operations outside NZ EEZ covered by Japan's observer program on its distant water fishing.

The summary of input data is shown in Table 2, and the spatial distribution of observed longline effort in Figure 2. The pattern of species occurrences varied substantially between fishery groups as well as operating locations. It should be noted that there were no observed Amsterdam albatross captures, despite a relatively high level of overlap between the distribution of Amsterdam albatross and longline and fisheries.

2.3 Assessment of total captures and their impacts on population sustainability

Total captures of albatrosses by tuna longline fisheries in the southern hemisphere were estimated by applying the bycatch estimation model developed under the Section 2.1 to the total monthly fishing efforts by 5x5 degrees square. For the Japan and NZ fleets, the average monthly effort for the period of 2012 to 2016 was used, together with the corresponding parameter estimates from the model. The estimated mortalities by other fleets were calculated by applying the parameters obtained for the "Japan (outside NZ)" group to annual average efforts by month by 5x5 degrees square, obtained from Regional Fisheries Management Organisations (RFMOs). This assumes that the other fleets had similar operations, with respect to seabird bycatch, as the Japanese high seas fleet. The spatial distribution of fishing effort utilized for this estimation is shown in Figure 3 (by flag).

Estimated annual bycatch mortality (ABM), corresponding to the dead bycatch, was then compared with the Population Sustainability Threshold (PST) that was defined as a half of the multiplier of the maximum population growth rate (r_{max}) under optimal condition and population size. The PST is an indicative measure of mortality that could be sustained by a seabird population while allowing the population remain at above half the carrying capacity in the long term. For the PST calculation, the total population size N was calculated from the current age at first breeding and current adult annual survival rate,

using Gilbert's (2009) formula; and the maximum population growth rate was obtained using demographic parameters from an allometric model, following Niel and Lebreton (2005)[4]. The prior means and standard deviations of maximum population growth, total population and PST for seven examined species are shown in Table 3.

When assessing the impact of pelagic longline bycatch on seabird populations, the populations **N** were represented as log-normal distributions with mean and standard deviations obtained from references.

3. Results

3.1 Model convergence

For all parameters, the measure Rhat (Gelman et al 2013) was lower than 1.1, indicating that the models converged.

3.2 Species-specific susceptibility and fishery-specific seabird catchability

The catchability estimates were sensitive to the assumptions made about the model structure (Table 4). Japan's fleet operating within the NZ EEZ consistently had the lowest fishery-specific seabird catchability. On the other hand, the fishery-specific seabird catchability of Japan's fleet operating outside NZ EEZ was either higher or lower than that of the NZ domestic fleet, depending on the scenario. Scenario 1 estimated Japan's fleet catchability as about 20% higher than NZ domestic longline, while Scenario 2 indicated it to be about 40% lower than NZ domestic longline. The catchability of Japan's fleet within the NZ EEZ (compared to NZ domestic longline catchability) also substantially differed between two scenarios: it was 22% in Scenario 1 and 7% in Scenario 2.

Both scenarios consistently indicated high bycatch susceptibility for three species: Antipodean, southern royal, and wandering albatross; low for northern royal and Amsterdam albatross; and somewhere in between for the remaining two species. Both scenarios also gave consistent patterns of species – fishery group interaction (Figure 4).

However, this result would be influenced by the ability to identify different species. In particular, for Japan's fleet that operates over a broad range of seasons and areas, it is easier to identify Gibson's albatross than Antipodean albatross, and southern royal than northern royal albatross, which may be reflected in the resulted ranking of the species x flag catchability.

3.3 Estimation of total albatross longline mortality

The highest mortalities were estimated for wandering albatross, followed by Gibson's albatross, while there was low estimated mortality of Amsterdam and Northern royal albatrosses (Table 5). All of these results should be considered as preliminary.

Total albatross captures were estimated to be highest for the Taiwanese fleet, followed by Japan, NZ, Korea and Australia. Since all fleets included in this model, other than NZ, utilized the same catchability, the differences reflect differences in the overlap between fishing and seabird distributions.

---

[4] The PST calculation is detailed by Richard et al. (2017; Appendix D)

There was no significant difference in the estimated total albatrosses mortality between Scenario 1 and 2. In other words, the estimate of total bycatch was stable despite the sensitivity in fishery-specific catchability to assumptions about the structure of the catchability.

This result should be considered only as indicative, since not all fleets operated in a similar way as Japan, and some fleets operating in distinctively different areas could be subject to different oceanographic conditions that may influence catchability. Ideally, observer records of albatross bycatch would be available from other fleets. Moreover, we have relied on generic information on seabird distributions (rather than using detailed analysis from tracking studies). A further limitation was that the effort data were derived from RFMOs and NZ only. Apart from NZ, pelagic longline fishing within EEZs was not included. We anticipate that it would be straightforward to update this analysis to include seabird bycatch and fishing effort data from other sources, or to include distributions derived from tracking data, if they were available.

When cryptic mortalities were included, the model parameters remained almost the same, however the estimated total albatross mortality more than doubled. Further information on the nature and extent of cryptic mortality in longline fisheries would be needed to resolve the total mortality that are occurring as a direct result of fishing activity. The model also assumed that, on average, half the albatrosses that are released alive survive. There was no information available to estimate the post-release survival of albatross that are caught in pelagic longline fisheries.

3.4 Impacts of longline mortality on albatross sustainability

The potential impacts of longline bycatch mortality on albatross population sustainability were examined by comparing the total annual bycatch mortality (ABM) with the population sustainability threshold (PST). The results are shown in Table 6 and Figure 5. When the ratio of ABM and PST is higher than 1, there is a risk that the longline bycatch would lead to the population falling below half the carrying capacity in the long term.

For wandering albatross, the mean estimated bycatch exceeds this threshold. The population of wandering albatross is in decline (e.g., Pardo et al 2017) with decreases in both adult and juvenile survival associated with increases in pelagic longline effort. Based on the estimates here, the annual bycatch mortality in longline fisheries represents around 1.5% of the total population.

The risk to Gibson's albatross was less than one, however Gibson's albatross, which breeds on Adams Island in New Zealand, has recently had reduced adult survival, low recruitment, and ongoing population decline at a study site (Eliott et al 2018). The generic distribution used did not reflect the tendency of tracked birds to forage in the Tasman sea, where there is high fishing effort.

The risk to Antipodean albatross appeared to be low, with a mean of less than 0.2. There has been a marked decline in the adult survival of Antipodean albatross since 2004, however, with female adult survival being less than 90%. This is associated with an apparent shift in the distribution, with increased foraging in waters off Chile (Elliott and Walker 2017).

The risks to both northern-and southern royal albatross are low (a mean of 0.15 or less for southern royal albatross, and a mean of 0.02 or less for northern royal albatross).

Northern and southern royal albatross are reported as bycatch on the Patagonian Shelf (Jiménez et al 2014, Jiménez et al 2017).

The results for Tristan and Amsterdam albatrosses were difficult to interpret, due to large uncertainty.

For all these species, it is important to reiterate that the results presented here are preliminary, and a different conclusion may be reached when further seabird distribution information is available.

The risks were estimated without cryptic mortalities. Even though actual mortality is uncertain, there is likely to be some birds that are caught on set, but not retained until the haul. This means that the direct impact of pelagic lining on seabirds is likely to be under represented in this analysis, as it does not take into account the impacts of cryptic mortality.

4. Discussion, future work plan and recommendation

This analysis has demonstrated that, by introducing the SEFRA framework, it is possible to estimate species-specific seabird bycatch mortality from limited observed bycatch information. At the same time, the analysis indicated some constraints and uncertainties, in particular for rarely caught species whose information on distribution and bycatch were also limited, to estimate the total bycatch. Even if the resulting estimates are uncertain, they may still be usefully compared with population sustainability parameters.

An advantage of this methodology is that the seabird demographic information, e.g. biological parameters, population status, behavior, distribution, are included with the bycatch assessment, resulting in a consistent assessment. The risk assessment method is simple and robust, and can be readily updated with new information. Consistency could also be achieved by developing an integrated population model (e.g. Tuck et al 2011), including both seabird and fisheries related components.

This analysis extrapolated captures from observed fishing to the total pelagic longline effort available from RFMOs. In other words, the estimated mortality in this analysis only reflects the fishing effort submitted to the RFMOs. Reliability, accuracy and representativeness of fishing effort data would directly translate into the reliability of annual bycatch mortality estimates. We emphasize the importance of ensuring that comprehensive fishing effort and observed seabird bycatch data are maintained by each RFMO. It should also be noted that this analysis only considered pelagic longline fishing. Seabirds will also be caught or entangled with other fishing gears, so the estimates shown here would be underestimates as overall fisheries mortality of seabird bycatch.

The quality of analysis also relies on the quality of available seabird distribution information. Although tracking data may be requested from researchers through Birdlife International[5], currently no analyzed seabird distribution information is readily available for use in fisheries bycatch assessments. If distribution information was available for each colony, and for male and female birds, or for different life stages, this would allow for a correspondingly detailed analysis of bycatch information. The analysis highlights the value of seabird distribution information for managing seabird bycatch issues.

---

[5] http://seabirdtracking.org

While recognizing the importance of efforts to reduce overall bycatch by tuna fisheries, more focused attention may be needed for those seabird species that require urgent conservation actions, and are known to be subject to substantial impacts from fisheries. The SEFRA analysis could help to explore possible pragmatic methods to reduce the bycatch impacts of fisheries on key species.

This preliminary analysis identified differences in fishery-specific catchability between Japan's fleet operating within NZ's EEZ and Japan's fleet operating outside NZ's EEZ. Both follow similar operational procedures, but the former operated with close to 100% observer coverage, and followed strict seabird mitigation requirements. The low catchability of seabirds by the Japanese fleet operating within the NZ EEZ is likely to reflect the reduction in bycatch achievable if mitigation measures were implemented consistently and effectively.

In principle, it would be possible to define fishery groups in different ways for other purposes, e.g. disaggregating by years to monitor trends over time of catchability, or assessing the effectiveness of different mitigation measures in use. At the same time, the fishery-specific catchability was rather sensitive to model structure and assumptions. Before further disaggregating the observer data, it would be important to identify basic requirements, to obtain a robust estimation of fishery-specific catchability, including minimum quantity of input data.

The plan for further refinement includes:

- investigation of robustness, including a determination of what observer data is required in order to obtain reliable and stable catchability estimates;

- improvement of input datasets, especially of seabird distributions;

- Inclusion of effort data from EEZs;

- further improvement of model structures;

- expansion of fisheries (i.e. including fisheries other than pelagic longlines);

- expansion of seabird species; and

- inclusion of observed effort and captures from other fleets.

We would welcome collaboration with any country who wish to become involved in this project. The method requires not require data at more detail than a 5-degree, quarterly resolution. A reference implementation of the assessment is available online[6], so that collaborators will be able to analyze their own data if they have capacity. Otherwise, the necessary technical assistance could be provided.

The authors request the WCPFC member countries consider collaborating in the approach to improve the following iteration of this risk assessment.

5. Acknowledgement

---

[6] https://github.com/seabird-risk-assessment/seabird-risk-assessment

6. References

Abraham, E., Roux, M.-J., Richard, Y., & Walker N. (2017a). *Assessment of the risk of southern hemisphere surface longline fisheries to ACAP species* (13th WCPFC Scientific Committee, EB-IP-13).

Abraham, E. Roux, M.-J., Richard, Y. and Walker N. (2017b). *Assessing the risk of commercial surface longline fisheries in the southern hemisphere to ACAP seabird species* (CCSBT-ERS/1703/12).

Anderson, O. R. J., Small, C. J., Croxall, J. P., Dunn, E. K., Sullivan, B. J., Yates, O., & Black, A.. Global seabird bycatch in longline fisheries. *Endangered Species Research, 14*(2), 91–106.

Brothers, N., Duckworth, A. R., Safina, C., & Gilman, E. L. (2010). Seabird bycatch in pelagic longline fisheries is grossly underestimated when using only haul data. *PLoS One, 5(*8), e12491.

Croxall, J. P., Butchart, S. H., Lascelles, B. E. N., Stattersfield, A. J., Sullivan, B. E. N., Symes, A., & Taylor, P. H. I. L. (2012). Seabird conservation status, threats and priority actions: a global assessment. *Bird Conservation International, 22*(1), 1-34.

Elliott, G., & Walker, K. (2017). Antipodean wandering albatross census and population study 2017. Unpublished report for the New Zealand Department of Conservation (20 p).

Elliott, G., Walker, K., Parker, G., & Rexer-Huber, K. (2018). Gibson's wandering albatross population study and census. Unpublished report for the New Zealand Department of Conservation (16 p).

Gelman, A., Carlin, J. B., Rubin, D. B., Vehtari, A., Dunson, D. B., & Stern, H. S. (2013). *Bayesian data analysis (3rd ed.)*. Boca Raton, Fl: CRC Press.

Gilbert, D. (2009). *Calculating the population ratio of total seabirds to adults.* Unpublished report, Ministry for Primary Industries, Wellington, New Zealand.

Inoue, Y., Yokawa, K., Minami, H., Ochi, D., Sato, N., & Katsumata, N. (2012). Distribution of seabird by-catch using data collected by Japanese observers in 1997-2009 in the ICCAT area. *Collect. Vol. Sci. Pap. ICCAT, 68*(5), 1738-1753.

Inoue, Y., Yokawa, K., Minami, H., Ochi, D., Sato, N., & Katsumata, N. (2011). *Distribution of seabird bycatch at WCPFC and the neighboring area of the southern hemisphere*. WCPFC-SC7-2011/EB-WP-07.

Inoue, Y., Yokawa, K., Minami, H., & Ochi, D. (2012). Preliminary view of by-catch hotspot: distribution of seabirds from tracking data, interaction map between seabird distribution and longline effort and by-catch distribution in the ICCAT convention area of the southern hemisphere. *Collect. Vol. Sci. Pap. ICCAT, 68*(5), 1784-1812.

Jiménez, S., Phillips, R. A., Brazeiro, A., Defeo, O., & Domingo, A. (2014). Bycatch of great albatrosses in pelagic longline fisheries in the southwest Atlantic: Contributing factors and implications for management. *Biological Conservation, 171*, 9-20.

Jiménez, S., Domingo, A., Brazeiro, A., Defeo, O., Abreu, M., Forselledo, R., & Phillips, R. A. (2017). Sexual size dimorphism, spatial segregation and sex-biased bycatch of southern and northern royal albatrosses in pelagic longline fisheries. *Antarctic Science, 29*(2), 147-154.

Niel, C. & Lebreton, J. (2005). Using demographic invariants to detect overharvested bird populations from incomplete data. *Conservation Biology, 19*, 825-835*.*

Pardo, D., Forcada, J., Wood, A. G., Tuck, G. N., Ireland, L., Pradel, R., Croxall, J. P. & Phillips, R. A. (2017). Additive effects of climate and fisheries drive ongoing declines in multiple albatross species. *Proceedings of the National Academy of Sciences*, *114*(50), E10829-E10837.

Phillips, R. A., Gales, R., Baker, G. B., Double, M. C., Favero, M., Quintana, F., ... & Wolfaardt, A. (2016). The conservation status and priorities for albatrosses and large petrels. *Biological Conservation, 201*, 169-183.

Richard, Y. & Abraham, E. R. (2013). *Application of Potential Biological Removal methods to seabird populations.* (New Zealand Aquatic Environment and Biodiversity Report 108). Wellington: Ministry for Primary Industries.

Richard, Y., Abraham, E. R., & Berkenbusch, K. (2018). A*ssessment of the risk of commercial fisheries to New Zealand seabirds, 2006-07 to 2014-15*. (New Zealand Aquatic Environment and Biodiversity Report, 191). Wellington: Ministry for Primary Industries.

Sharp, B. R., Cryer, M., Richard, Y., & Abraham, E. R. (2013). New Zealand's approach to assessment of risk to seabirds associated with fishing-related mortality. *5th meeting of the Seabird Bycatch Working Group, 12*.

Sharp, B. R. (2016). *Spatially Explicit Fisheries Risk Assessment (SEFRA): A framework for quantifying and managing incidental commercial fisheries impacts on non-target species and habitats.* Unpublished report, Ministry for Primary Industries, Wellington, New Zealand.

Tsuji, S. (2017). *Statistical consideration on sea-bird mitigation measure evaluation* (CCSBT-ERS/1703/27). Unpublished report, 2017 Intersessional Meeting of the Sub-committee on ecosystems, International Commission for the Conservation of Atlantic Tunas.

Tuck, G. N., Phillips, R. A., Small, C., Thomson, R. B., Klaer, N. L., Taylor, F., ... & Arrizabalaga, H. (2011). An assessment of seabird–fishery interactions in the Atlantic Ocean. *ICES Journal of Marine Science, 68*(8), 1628-1637.

Waugh, S. M., Filippi, D. P., Kirby, D. S., Abraham, E., & Walker, N. (2012). Ecological Risk Assessment for seabird interactions in Western and Central Pacific longline fisheries. *Marine Policy, 36*(4), 933-946.

Table 1. Biological parameters utilized for the seven *Diomedea* species included in the assessment. For each species, the table gives the mean and 95% quantiles of the distributions. The parameters are derived from information provided by ACAP, following methods described by Richard et al (2018).

Parameter	Mean	95% c.i.	Parameter	Mean	95% c.i.
Amsterdam albatross			**Southern royal albatross**		
Survival (current)	0.954	(0.905 – 0.982)	Survival (current)	0.948	(0.931 – 0.963)
Survival (optimal)	0.954	(0.906 – 0.981)	Survival (optimal)	0.948	(0.931 – 0.962)
Age at first breeding	11.5	(10.1 – 12.9)	Age at first breeding	9.5	(8.6 – 10.5)
Breeding probability	0.598	(0.501 – 0.691)	Breeding probability	0.600	(0.500 – 0.693)
Annual breeding pairs	46	(42 – 50)	Annual breeding pairs	7940	(7250 – 8670)
Total population	264	(174 – 452)	Total population	42000	(33000 – 54300)
Antipodean albatross			**Tristan albatross**		
Survival (current)	0.956	(0.941 – 0.969)	Survival (current)	0.906	(0.834 – 0.955)
Survival (optimal)	0.956	(0.940 – 0.969)	Survival (optimal)	0.950	(0.843 – 0.991)
Age at first breeding	11.5	(10.1 – 12.9)	Age at first breeding	10.0	(7.1 – 12.9)
Breeding probability	0.599	(0.498 – 0.692)	Breeding probability	0.599	(0.501 – 0.691)
Annual breeding pairs	3630	(1730 – 6440)	Annual breeding pairs	1110	(1010 – 1210)
Total population	19600	(8760 – 37100)	Total population	9870	(5120 – 22400)
Gibson's albatross			**Wandering albatross**		
Survival (current)	0.959	(0.945 – 0.969)	Survival (current)	0.953	(0.939 – 0.963)
Survival (optimal)	0.959	(0.945 – 0.969)	Survival (optimal)	0.953	(0.940 – 0.964)
Age at first breeding	11.0	(10.1 – 11.9)	Age at first breeding	10.0	(7.1 – 12.8)
Breeding probability	0.599	(0.503 – 0.693)	Breeding probability	0.599	(0.504 – 0.693)
Annual breeding pairs	4890	(4380 – 5450)	Annual breeding pairs	8390	(7350 – 9620)
Total population	25100	(20000 – 31900)	Total population	43900	(32700 – 59300)
Northern royal albatross					
Survival (current)	0.951	(0.928 – 0.968)			
Survival (optimal)	0.951	(0.928 – 0.968)			
Age at first breeding	9.6	(8.6 – 10.5)			
Breeding probability	0.609	(0.509 – 0.703)			
Annual breeding pairs	5790	(5420 – 6150)			
Total population	29700	(23200 – 38800)			

Table 2. Observer data used to parameterise the bycatch estimation model. For each species and fisheries group, the table gives the observed effort, the overlap, the observed captures (total, alive, and dead), and the observed capture rate (calculated relative to the overlap: [captures] x 1000 / ([overlap] x [total population])).

Fishery group	Effort(1000 hooks)	Overlap	Captures	Alive	Dead	Capture rate
Amsterdam albatross						
Japan (outside NZ)	55562	0.779	0	0	0	0.000
Antipodean albatross						
Japan (inside NZ)	18591	0.107	5	0	5	2.384
Japan (outside NZ)	55562	0.152	4	0	4	1.348
New Zealand	2923	0.116	31	13	18	13.726
Gibson's albatross						
Japan (inside NZ)	18591	3.651	13	1	12	0.142
Japan (outside NZ)	55562	0.152	25	2	23	6.550
New Zealand	2923	0.117	28	5	23	9.520
Northern royal albatross						
Japan (inside NZ)	18591	0.037	0	0	0	0.000
Japan (outside NZ)	55562	0.231	1	0	1	0.146
New Zealand	2923	0.022	1	0	1	1.526
Southern royal albatross						
Japan (inside NZ)	18591	0.037	4	1	3	2.559
Japan (outside NZ)	55562	0.258	63	11	52	5.785
New Zealand	2923	0.022	6	3	3	6.415
Tristan albatross						
Japan (outside NZ)	55562	0.764	14	0	14	1.859
Wandering albatross						
Japan (inside NZ)	18591	0.183	0	0	0	0.000
Japan (outside NZ)	55562	0.475	262	13	249	12.581
New Zealand	2923	0.017	6	1	5	7.846
Unidentified wandering and royal spp.						
Japan (inside NZ)	18591		3	2	1	
Japan (outside NZ)	55562		163	7	156	
New Zealand	2923		12	7	5	

Table 3. Summary of the parameters defining population sustainability thresholds. These are derived from the input biological parameters following the methods outlined by Richard et al (2018).

(a) Scenario 1

Taxa	Total population		r_{max}		PST	
	Mean	95% c.i.	Mean	95% c.i.	Mean	95% c.i.
Amsterdam albatross	262.9	157.3–417.3	0.046	0.021–0.060	3.01	1.20–5.20
Tristan albatross	9789.2	4351.9–19043.4	0.051	0.006–0.085	125.44	12.55–296.83
Southern royal albatross	42196.4	32398.9–54176.3	0.056	0.048–0.064	589.58	438.50–782.96
Northern royal albatross	29623.6	22486.1–38080.6	0.055	0.044–0.064	404.56	286.34–545.02
Gibson's albatross	25084.9	19698.4–31490.6	0.047	0.040–0.053	293.37	221.60–378.99
Wandering albatross	44019.9	32173.4–58518.6	0.054	0.041–0.071	589.37	386.22–876.37
Antipodean albatross	19433.7	7748.9–40006.6	0.046	0.039–0.054	224.05	89.08–465.28

(b) Scenario 2

Taxa	Total population		r_{max}		PST	
	Mean	95% c.i.	Mean	95% c.i.	Mean	95% c.i.
Amsterdam albatross	253.9	154.4–396.9	0.046	0.023–0.060	2.91	1.23–5.02
Tristan albatross	8872.7	4261.8–16415.0	0.051	0.005–0.086	114.19	10.37–259.85
Southern royal albatross	44376.3	34288.4–56553.5	0.056	0.048–0.064	620.63	458.50–825.10
Northern royal albatross	27092.8	20746.7–34781.1	0.055	0.045–0.064	371.35	265.17–502.21
Gibson's albatross	25114.7	19833.6–31454.4	0.047	0.041–0.053	293.64	223.47–379.73
Wandering albatross	45092.4	33226.1–59535.2	0.054	0.042–0.072	603.76	404.74–892.03
Antipodean albatross	24296.0	12314.6–43227.9	0.046	0.039–0.054	280.29	138.76–507.83

Table 4. Species-specific susceptibility and fishery-specific seabird catchability parameters as estimated based on Japan and NZ observer data. For each model scenario (see section 2.1), and for each parameter, the model gives the mean and 95% credible interval (c.i.) of the posterior distribution.

(a) Scenario 1

Parameter	Mean	95% c.i.
Intercept		
	0.0001	(0.00004 – 0.0004)
Fishery group		
Japan (outside NZ)	1.230	(0.288 – 3.436)
New Zealand	1.000	(1.000 – 1.000)
Japan (inside NZ)	0.216	(0.034 – 0.753)
Species		
Antipodean albatross	12.134	(2.963 – 30.328)
Southern royal albatross	11.516	(2.870 – 28.673)
Wandering albatross	10.981	(2.758 – 27.110)
Gibson's albatross	8.960	(2.131 – 23.310)
Tristan albatross	6.787	(0.705 – 22.621)
Northern royal albatross	2.047	(0.126 – 8.833)
Amsterdam albatross	1.000	(1.000 – 1.000)
Species x fishery group		
Wandering albatross in Japan (outside NZ)	19.281	(5.113 – 50.257)
Antipodean albatross in New Zealand	15.047	(3.722 – 37.238)
Southern royal albatross in Japan (inside NZ)	14.944	(2.918 – 41.977)
Antipodean albatross in Japan (inside NZ)	14.148	(2.833 – 38.294)
Gibson's albatross in New Zealand	14.124	(3.661 – 36.383)
Gibson's albatross in Japan (outside NZ)	12.242	(2.809 – 30.839)
Northern royal albatross in New Zealand	10.430	(1.088 – 32.191)
Amsterdam albatross in Japan (inside NZ)	10.138	(0.309 – 36.265)
Amsterdam albatross in New Zealand	9.980	(0.338 – 33.871)
Tristan albatross in Japan (inside NZ)	9.929	(0.392 – 34.445)
Tristan albatross in New Zealand	9.810	(0.287 – 34.700)
Wandering albatross in New Zealand	9.611	(1.832 – 25.623)
Southern royal albatross in Japan (outside NZ)	8.746	(1.779 – 23.105)
Amsterdam albatross in Japan (outside NZ)	8.392	(0.276 – 28.173)
Northern royal albatross in Japan (inside NZ)	7.936	(0.224 – 28.441)
Southern royal albatross in New Zealand	7.898	(1.428 – 21.793)
Tristan albatross in Japan (outside NZ)	6.750	(0.738 – 22.688)
Northern royal albatross in Japan (outside NZ)	3.575	(0.153 – 14.568)
Antipodean albatross in Japan (outside NZ)	2.479	(0.303 – 8.500)
Gibson's albatross in Japan (inside NZ)	1.738	(0.182 – 6.208)
Wandering albatross in Japan (inside NZ)	1.074	(0.013 – 5.411)

Table 4 (continued).

    (b) Scenario 2

Parameter	Mean	95% c.i.
Intercept		
	0.011	(0.005 – 0.022)
Fishery group		
New Zealand	1.000	(1.000 – 1.000)
Japan (outside NZ)	0.584	(0.203 – 1.321)
Japan (inside NZ)	0.068	(0.019 – 0.180)
Species x fishery group		
Wandering albatross in Japan (outside NZ)	3.417	(1.674 – 6.434)
Southern royal albatross in Japan (inside NZ)	2.391	(0.689 – 5.874)
Antipodean albatross in Japan (inside NZ)	2.381	(0.703 – 5.971)
Gibson's albatross in Japan (outside NZ)	1.717	(0.776 – 3.332)
Southern royal albatross in Japan (outside NZ)	1.512	(0.715 – 2.823)
Antipodean albatross in New Zealand	1.376	(0.518 – 3.007)
Gibson's albatross in New Zealand	1.116	(0.471 – 2.241)
Tristan albatross in Japan (inside NZ)	1.014	(0.189 – 3.122)
Amsterdam albatross in Japan (inside NZ)	0.997	(0.181 – 3.156)
Amsterdam albatross in New Zealand	0.994	(0.179 – 3.113)
Tristan albatross in New Zealand	0.987	(0.178 – 3.054)
Wandering albatross in New Zealand	0.889	(0.314 – 1.933)
Northern royal albatross in Japan (inside NZ)	0.760	(0.150 – 2.093)
Southern royal albatross in New Zealand	0.750	(0.266 – 1.617)
Amsterdam albatross in Japan (outside NZ)	0.713	(0.150 – 1.959)
Tristan albatross in Japan (outside NZ)	0.649	(0.236 – 1.432)
Northern royal albatross in New Zealand	0.452	(0.117 – 1.111)
Antipodean albatross in Japan (outside NZ)	0.429	(0.139 – 0.964)
Wandering albatross in Japan (inside NZ)	0.400	(0.092 – 1.023)
Gibson's albatross in Japan (inside NZ)	0.340	(0.111 – 0.778)
Northern royal albatross in Japan (outside NZ)	0.182	(0.052 – 0.410)

Table 5. Estimates of total annual albatross mortality by flag and by species, based on applying the bycatch model to fishing effort data from between 2012 and 2016. The fishery-specific catchability of fleets other than Japan and NZ were assumed to be the same as that of the Japan high-seas fleet. The Japanese fleet includes Japanese vessels inside and outside the New Zealand EEZ. For each fleet, and for each species the table gives the mean and 95% credible interval (c.i.) of the posterior distribution of the annual bycatch mortality (ABM).

(a) Scenario 1

Flag	All albatrosses	Amsterdam albatross	Antipodean albatross	Gibson's albatross	Tristan albatross	Northern royal albatross	Southern royal albatross	Wandering albatross
All flags	1070 (834–1345)	1 (0–4)	49 (14–116)	162 (100–242)	86 (29–189)	7 (0–21)	81 (51–119)	684 (482–928)
Australia	23 (13–36)	0 (0–0)	2 (0–7)	10 (3–18)		0 (0–1)	1 (0–4)	10 (4–18)
Belize	39 (23–57)				9 (2–22)	0 (0–0)	1 (0–3)	29 (16–44)
China	80 (53–114)	0 (0–0)	9 (1–26)	45 (24–71)	0 (0–2)	0 (0–1)	1 (0–4)	25 (13–38)
Fiji	17 (8–30)		3 (0–9)	14 (6–25)				0 (0–2)
Japan	130 (96–170)	0 (0–1)	5 (0–15)	20 (9–34)	9 (2–22)	0 (0–2)	14 (6–24)	82 (53–116)
New Zealand	55 (32–88)		19 (5–44)	17 (7–31)		5 (0–18)	10 (2–23)	4 (0–10)
Other flags	28 (17–41)	0 (0–0)	2 (0–6)	8 (2–16)	4 (0–12)	0 (0–1)	2 (0–5)	12 (5–20)
Seychelles	12 (5–21)	0 (0–0)			2 (0–6)	0 (0–0)	0 (0–0)	10 (4–18)
South Africa	25 (14–38)	0 (0–0)			4 (0–10)	0 (0–1)	2 (0–5)	19 (9–30)
South Korea	43 (28–61)	0 (0–1)	0 (0–0)	0 (0–1)	5 (0–13)	0 (0–2)	9 (3–16)	29 (16–44)
Spain	34 (21–51)	0 (0–0)	1 (0–3)	3 (0–8)	7 (1–17)	0 (0–1)	1 (0–4)	22 (12–34)
Taiwan	553 (415–720)	1 (0–3)	6 (0–19)	32 (16–52)	45 (14–101)	1 (0–5)	39 (22–59)	429 (301–585)
Vanuatu	28 (17–42)		2 (0–8)	12 (5–22)		0 (0–0)	0 (0–2)	13 (6–22)

(b) Scenario 2:

| Flag	All albatrosses	Amsterdam albatross	Antipodean albatross	Gibson's albatross	Tristan albatross	Northern royal albatross	Southern royal albatross	Wandering albatross
All flags	1071 (951–1197)	3 (0–9)	52 (28–87)	155 (101–220)	88 (48–141)	17 (5–37)	79 (55–105)	677 (590–772)
Australia	23 (14–34)	0 (0–0)	2 (0–6)	9 (3–17)		0 (0–1)	1 (0–3)	10 (5–17)
Belize	39 (26–52)				9 (3–18)	0 (0–1)	1 (0–3)	28 (18–40)
China	80 (56–107)	0 (0–1)	10 (2–22)	43 (24–66)	0 (0–2)	0 (0–1)	1 (0–4)	25 (15–35)
Fiji	17 (8–28)		3 (0–8)	14 (6–23)				0 (0–2)
Japan	128 (103–154)	0 (0–2)	5 (1–12)	19 (9–32)	9 (3–18)	1 (0–4)	12 (5–21)	81 (62–101)
New Zealand	62 (39–90)		19 (9–32)	17 (7–30)		12 (2–29)	10 (2–22)	4 (0–10)
Other flags	28 (17–39)	0 (0–0)	2 (0–5)	8 (2–14)	5 (1–10)	0 (0–1)	2 (0–5)	11 (5–19)
Seychelles	12 (6–20)	0 (0–1)			2 (0–5)	0 (0–0)	0 (0–0)	10 (4–17)
South Africa	25 (15–35)	0 (0–1)			4 (0–9)	0 (0–1)	2 (0–5)	19 (10–28)
South Korea	43 (30–57)	0 (0–1)	0 (0–0)	0 (0–1)	5 (1–11)	1 (0–3)	8 (3–15)	29 (18–40)
Spain	35 (23–48)	0 (0–0)	1 (0–3)	3 (0–8)	7 (2–14)	0 (0–2)	1 (0–4)	22 (13–32)
Taiwan	551 (483–621)	2 (0–7)	7 (1–16)	30 (16–48)	46 (24–77)	3 (0–8)	38 (24–54)	425 (364–489)
Vanuatu	28 (17–40)		3 (0–7)	12 (5–21)		0 (0–1)	0 (0–2)	13 (6–21)

Table 6. Potential impacts of longline bycatch mortality on albatross populations. For each species, the table gives the mean and 95% credible interval (c.i.) of the Population Sustainability Threshold (PST), the Annual Bycatch Mortality and the Risk ratio (ABM/PST). The table also gives the probability that the annual bycatch mortality exceeded the sustainability threshold, based on the posterior distribution. No cryptic mortality is included in the estimation of the risk.

(a) Scenario 1:

Taxa	PST		ABM		Risk ratio		
	Mean	95% c.i.	Mean	95% c.i.	Median	95% c.i.	p(ABM>PST)
Wandering albatross	588.0	390.6–866.7	684	482–928	1.18	0.84–1.55	85.2%
Tristan albatross	123.9	10.2–294.1	86	29–189	0.66	0.29–7.44	22.4%
Gibson's albatross	293.0	221.4–377.4	162	100–242	0.55	0.36–0.81	0.1%
Antipodean albatross	224.2	88.9–471.9	49	14–116	0.21	0.11–0.39	0.0%
Southern royal albatross	589.9	437.5–776.8	81	51–119	0.14	0.09–0.19	0.0%
Northern royal albatross	405.1	287.2–547.5	7	0–21	0.01	0.00–0.05	0.0%
Amsterdam albatross	3.0	1.3–5.2	1	0–4	0.00	0.00–1.57	6.8%

(b) Scenario 2:

| Taxa	PST		ABM		Risk ratio		
	Mean	95% c.i.	Mean	95% c.i.	Median	95% c.i.	p(ABM>PST)
Wandering albatross	602.6	404.8–896.0	677	590–772	1.15	0.74–1.71	74.4%
Tristan albatross	112.1	10.7–252.8	88	48–141	0.82	0.30–8.42	37.2%
Gibson's albatross	293.9	222.8–381.4	155	101–220	0.53	0.33–0.82	0.1%
Antipodean albatross	279.8	133.8–501.7	52	28–87	0.19	0.09–0.41	0.0%
Southern royal albatross	620.7	456.4–820.6	79	55–105	0.13	0.08–0.19	0.0%
Northern royal albatross	371.3	264.9–499.1	17	5–37	0.04	0.01–0.10	0.0%
Amsterdam albatross	2.9	1.2–4.9	3	0–9	0.71	0.00–3.93	36.4%
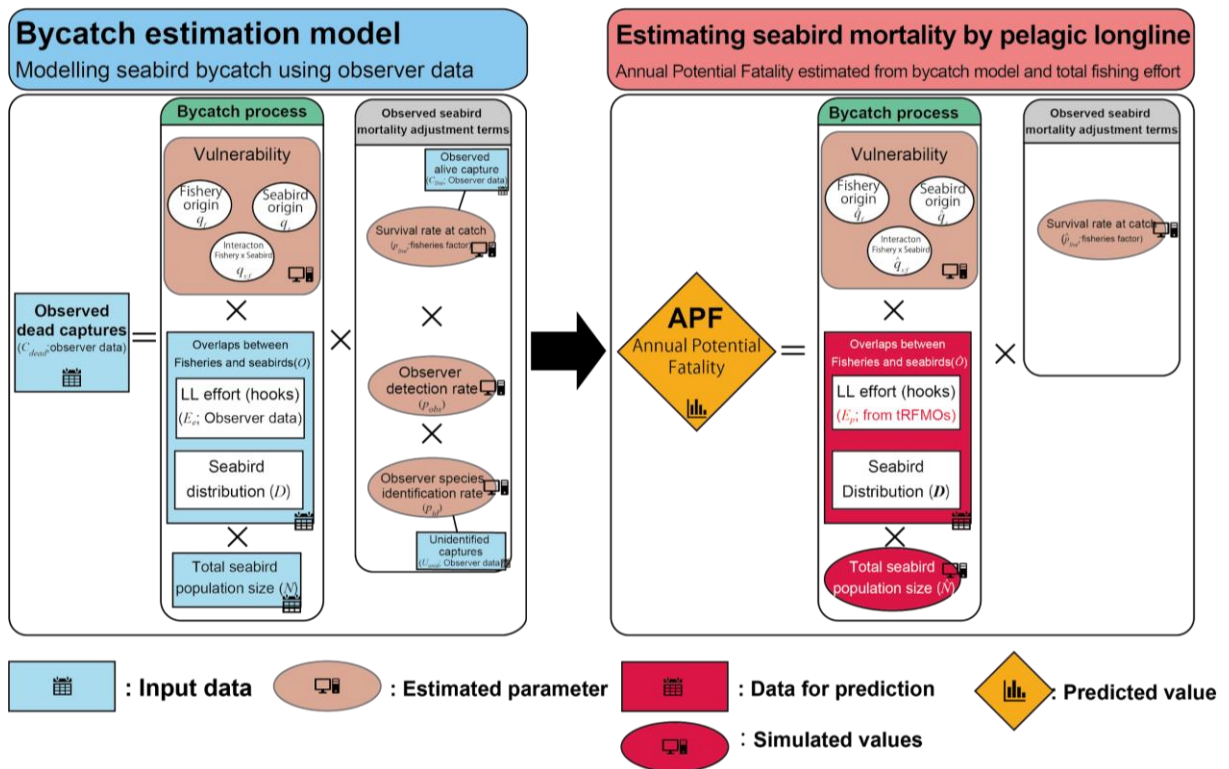
Figure 1. Overall structure of the bycatch estimation, showing the relationship between the observed and estimated quantities in the estimation model. The Annual Potential Fatality (APF) is estimated assuming that the observer detection rate is one, and so there are no cryptic mortalities.
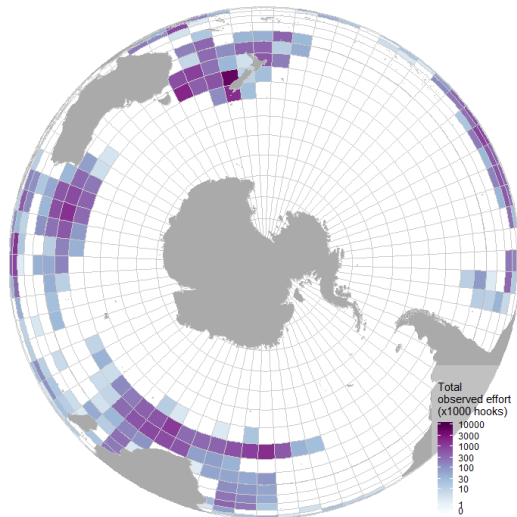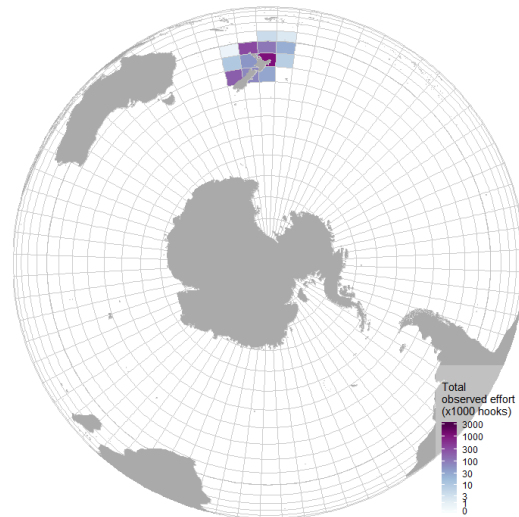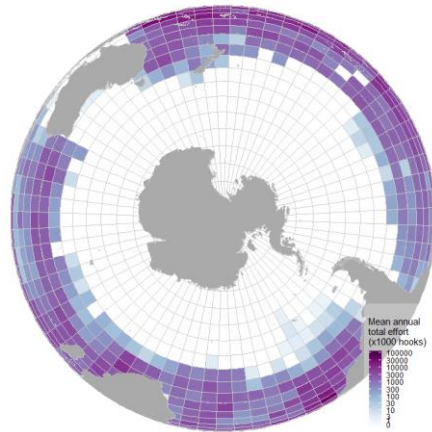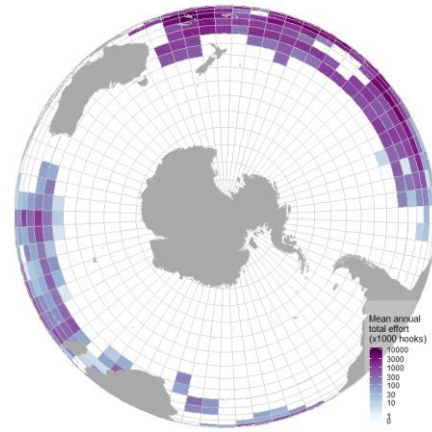
Figure 2. Spatial distribution of observed fishing effort of the Japan longline fishery (JP-LL) and the NZ domestic pelagic longline fishery (NZ-LL). Each 5-degree square shows the annual average number of hooks observed over the period 2003 – 2016.
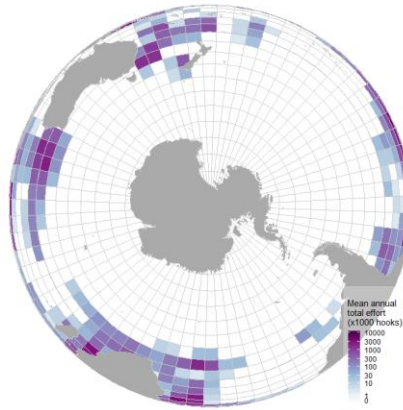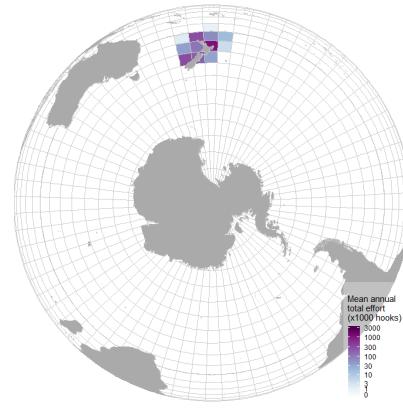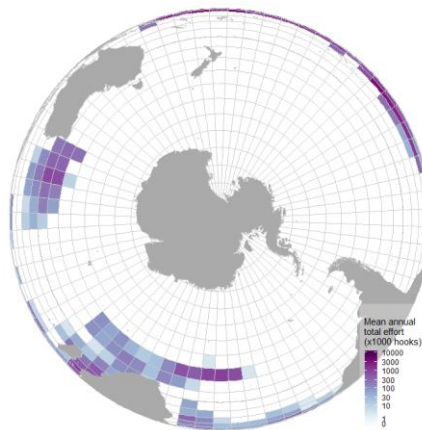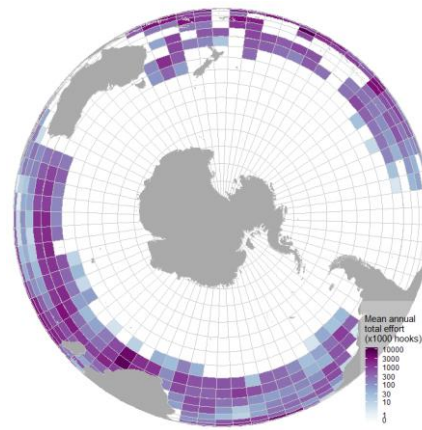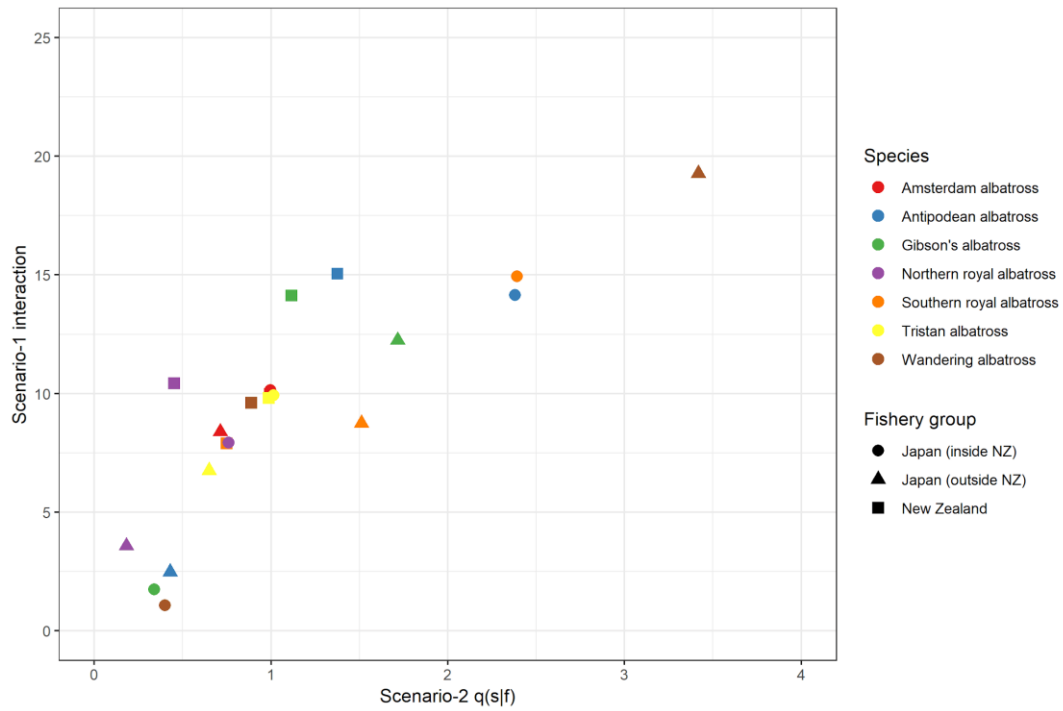
Figure 3. Pelagic longline effort distribution by flag. The colour within each 5-degree cell indicates the average annual number of hooks over the period 2012 to 2016. Fleets with a mean estimated *Diomedea* bycatch of less than fifty per year are not shown.

Figure 4. Comparison between the Scenario-1 interaction term $q_{s:f}$ and Scenario 2 $q_s(s|f)$. The shapes of the symbols correspond to fisheries defined in the estimation model, and the colors to albatross species.
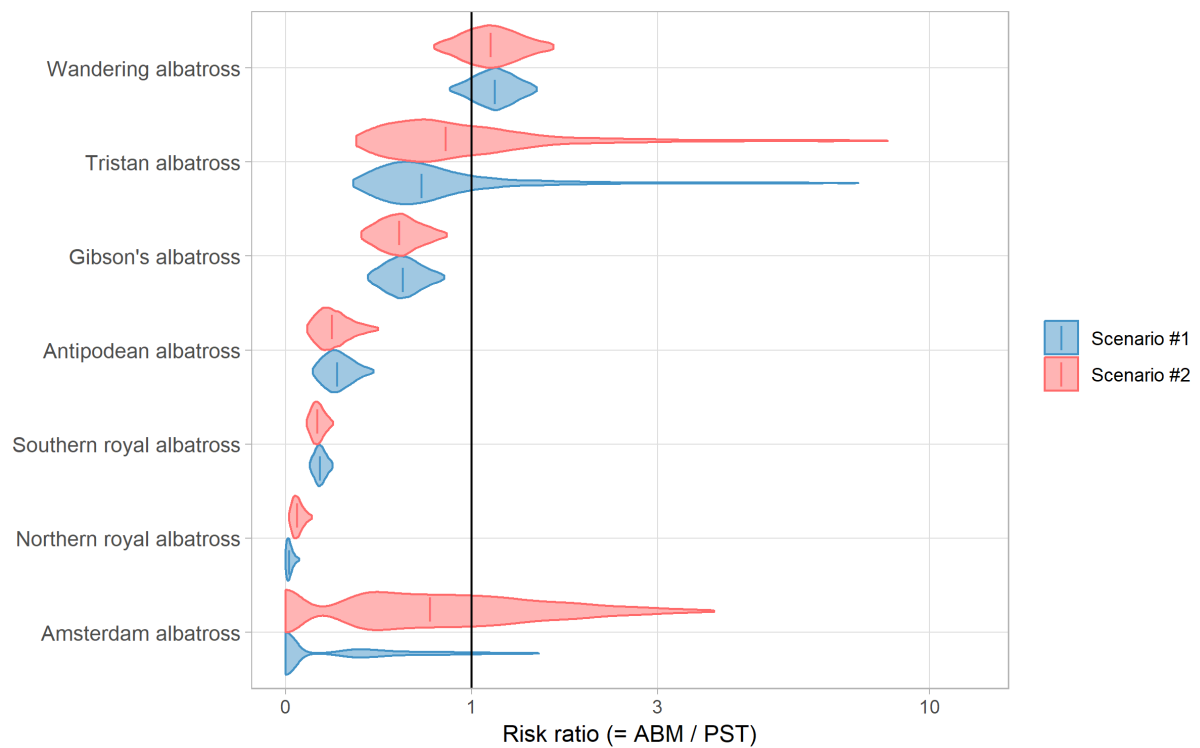
Figure 5. Uncertainties around the estimated impact of pelagic longline mortalities on great albatross. For each species, and for each of the two scenarios, the figure shows the mean and 95% credible interval of the risk ratio (given by the ratio of the Annual Bycatch Mortality (ABM) to the Population Sustainability Threshold (PST). No cryptic mortality is included in the estimation of the risk.