



Report of the Technical Working Group on CCSBT collaborative risk assessment for seabird bycatch with surface longlines in the Southern Hemisphere 2025

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1. ABSTRACT

A quantitative, spatially-explicit risk assessment for 25 taxa of albatrosses and petrels potentially caught in surface longline fisheries by participating Members of the CCSBT (Australia, Japan, Korea, New Zealand, South Africa, and the Fishing Entity of Taiwan) is described. This 2025 Spatially Explicit Fisheries Risk Assessment (SEFRA) builds on the 2024 SEFRA tabled at the CCSBT's ERSWG15 in April 2024. Diagnostics suggest that the 2025 SEFRA model had converged, fitted the data very well, and did not appreciably update any of the priors for biological inputs. The key concerns raised about the 2024 iteration have been resolved in the 2025 iteration in that: conflicts between estimated overlap of fishing and seabirds and the observed captures (i.e., the presence of observed captures where the estimated overlap was zero) have been resolved; and the fitted models no longer require implausible updates to the priors on biological inputs. Updating the available information on seabird distributions reduced the number of "zero overlap" captures" somewhat but fitting to genus-level (or higher) capture data (as opposed to specieslevel, where available) was by far the most influential change. The broad patterns of estimated risk were similar in the 2024 and 2025 SEFRA models; Gibson's albatross, Amsterdam albatross, Tristan albatross and Sooty albatross were the taxa estimated to be at highest risk in both model iterations. The estimated risk for many seabird taxa was higher in the 2025 SEFRA than in the 2024 SEFRA because the lower risk for many taxa in the 2024 SEFRA appears to be largely an artefact caused by data conflicts and the consequent updates to biological priors. At the scale of 5-degree squares, estimated annual deaths of great albatrosses and mollymawks were highest in the Tasman Sea, south-eastern Indian Ocean and south-eastern Atlantic Ocean. Sooty albatross deaths were highest in the south-eastern Atlantic Ocean and in the Tasman Sea. Deaths of medium petrels were highest around South Africa and off Namibia, and in the south-eastern Pacific Ocean. The 2025 SEFRA was not very sensitive to fitting to capture data aggregated to family-level (or higher) but was somewhat more sensitive to fitting only to data from 2012–2019 (as in the 2024 SEFRA). This is thought to be due to lower catchability in more recent years. A 2025 SEFRA with three time-blocks (2012–2016, 2017–2019, 2020–2023) appears to provide useful estimates of catchability and mortality for great albatrosses and mollymawks. Catchability was estimated to be somewhat lower since 2020 for some fleets. Several uncertainties and caveats remain but the 2025 SEFRA is considered to be a substantial improvement on the 2024 version.

2. BACKGROUND and INTRODUCTION

The issue of substantial interactions between SBT fisheries and seabirds was well recognised even at the time of establishment of the CCSBT in 1994. An initial draft of recommendations on reducing the incidental bycatch of seabirds was developed in 2006 at the 6th meeting of the CCSBT Ecologically Related Species Working Group (ERSWG), which ignited the debate whether the CCSBT can make binding measures for ERS related issues. Subsequently, the 7th meeting of ERSWG could not reach agreement on draft recommendations. The debate around the CCSBT's legal capacity to establish mandatory measures on ERS related matters continued until 2018 when the CCSBT agreed on the Resolution to Align CCSBT's Ecologically Related Species measures with those of other tuna RFMOs at the 25th Annual Meeting, which was updated at the 28th Annual Meeting in 2021.

A Performance Review was conducted in 2008 that criticised the ERSWG and pointed to, at the very least, a need to assess the risks and impacts of SBT fisheries on ERS species and adopt an appropriate mitigation strategy to address those risks and impacts. In response, the 15th Annual Commission meeting in 2008 agreed to develop a non-binding recommendation for the CCSBT covering bycatch mitigation for seabirds, sea turtles and sharks. Additionally, it agreed to develop a Strategic Plan and established Strategy and Fisheries Management Working Group. The Plan was adopted at a Special Meeting held in 2011, which included three items

and seven action items under the ERSWG.

In 2014, the Strategy and Fisheries Management Working Group was re-established to discuss revisions of the action plan. At the same time, following the recommendation of ERSWG, a small technical group, Effectiveness of Seabird Mitigation Measures Technical Group (SMMTG), was established to provide advice to the ERSWG on feasible, practical, timely, and effective technical approaches for measuring and monitoring the effectiveness of seabird mitigation measures in SBT longline fisheries. Both groups tabled their reports in 2015. The ERSWG took the SMMTG recommendations to progress in two directions: 1) undertaking a global assessment of seabird bycatch collaboratively among all tuna RFMOs through the support of the ABNJ Tuna Project Seabirds component that was concluded in 2019 (Abraham et al 2019), and 2) developing an ERSWG work plan. The latter led to the development of the CCSBT Multiyear Seabird Strategy, which was adopted at the 26th Annual Meeting of CCSBT.

A range of actions to be undertaken under each specific objective of the Multi-year Seabird Strategy was developed at the 14th meeting of ERSWG in 2021 and adopted by the 29th Annual meeting of CCSBT, which included an action to "update SEFRA seabird risk assessment" (1E) with New Zealand and Japan volunteering to take a leading role inter-sessionally. This would also allow work to "assess the cumulative impacts of fishing for SBT on seabirds, particularly threatened albatross and petrel species, across tuna RFMOs including developing methods for extrapolating seabird bycatch levels and seabird bycatch rates to identify total mortalities and total mortality rates" (3D) to be undertaken.

New Zealand and Japan held initial discussions in Wellington, New Zealand in June 2022 and agreed on a tentative work plan that included two technical workshops, one online and the other hybrid, and one face-to-face data preparatory meeting (Appendix 1). It was also agreed that the CCSBT collaborative assessment would begin after the completion of a seabird risk assessment of fisheries within New Zealand and would be developed based on the model developed for the New Zealand domestic risk assessment.

Following the decision at the 29th meeting of the Commission to hold one technical workshop before ERSWG-15, the original work plan was modified to hold one combined meeting to review the Spatially Explicit Fisheries Risk Assessment (SEFRA) procedure developed by New Zealand and to agree on basic data requirements in 2023, and one assessment meeting online, but with voluntary participation face-to-face.

The first technical workshop (hybrid) was held in Wellington, New Zealand, from 21 to 22 June 2023 with the participation of Australia, Japan, New Zealand and the Fishing Entity of Taiwan. Agreed outcomes from the meeting can be found in Appendix 2. The meeting agreed the first collaborative assessment would be based on the best available science and knowledge and provide a basis for future regular assessments with continuous improvements. The technical workshop agreed a range of basic assumptions, the time-period subject to the analysis, a range of species to be covered, and the temporal and spatial resolutions. The workshop established two expert teams: 1) for reviewing seabird biological parameters and distribution data, and 2) for incorporating modifications agreed at the workshop and evaluating them, together with the draft work schedule.

A review of biological parameters was shared among the group in January 2024. The New Zealand domestic seabird risk analysis was concluded in October 2023 and the program package including seabird observed catch and effort preparation package was provided in late 2023. Thereafter, the Members processed the observed seabird catch and effort data and ran the model for catchability estimation independently, using each Member's domestic information.

The second technical workshop (hybrid) was held in Wellington, New Zealand, from 27 to 29 February 2024 with participation from Australia, Japan, New Zealand and the Fishing Entity of Taiwan. The workshop reviewed the model outputs step-by-step and evaluated the reliability/feasibility of estimated parameters. The workshop noted problems in estimating species-

specific catch, mainly due to potential errors in observed seabird identification, and a mismatch in overlap caused by partial coverage of bird density distribution information with tracking data.

Consequently, the workshop agreed to further modify the model by incorporating new aggregation as a species complex for those species difficult to identify at species level. Observed capture and observed overlaps were summed across species within the species complex during the model fitting. Therefore, the model would ignore the species identification confusion within a species complex but would make a prediction of total mortality at species level relying on the overlap information (discussed further in section 4.2). The revised procedure was reviewed at an online discussion held on 4 April 2024 that confirmed general consistencies between the predicted and observed catches with the agreed aggregations.

The technical group examined the outputs of the modified model including the estimates of total bycatch mortalities and corresponding risks at an online discussion held on 23 April, 2024. The technical group noted that at least two of the biological parameters (the number of breeding pairs and the probability of breeding for some species) show a large shift away from the priors when the model was run (discussed further in Section 4.3). This would impact on the assessment of catchability estimates and evaluation of relative risks in particular for small albatrosses (mollymawks) and medium petrels, so the model output for those species groups should be interpreted carefully.

This document describes the process and results of the CCSBT collaborative seabird risk assessment for the surface longline fishery using the SEFRA framework. The document includes the methodology used, assumptions, input data and their preparation, initial review results and subsequent model modifications, and the final outputs. The document is focused on the description of facts and observations and does not include interpretations, particularly on potential implications for CCSBT seabird management.

While the outputs of the SEFRA update are expected to provide a basis for addressing other actions in the CCSBT Multi-year Seabird Strategy, including "to agree on a SBT seabird bycatch target for reducing the level of impact of SBT fishing operations on seabird populations" (1A), to "agree on the list of priority species and corresponding management targets, taking into account the status of seabird population, distributional overlaps with SBT fisheries, and significance of SBT fisheries in their mortality" (1D), and "establish a robust definition of high risk areas that takes into account the precautionary approach" (1F), such considerations are left to the individual Members and subsequent discussions at the ERS.

3. METHODS

The SEFRA model was based on that used for CCSBT's 2024 seabird risk assessment (Anon., 2024; Edwards et al., 2025b), and is described here in full for clarity.

3.1 General concept of SEFRA

The Spatially Explicit Fisheries Risk Assessment (SEFRA) framework used in this risk assessment was developed and has been utilised in New Zealand as standard procedure to estimate the risk to seabirds and other protected species caused by commercial fishing (Edwards et al. 2023, Abraham et al. 2017, Sharp 2019) and subsequently applied to the capture of albatrosses and petrels in southern hemisphere longline fisheries (Ochi et al 2018, Abraham et al. 2019). A glossary of model terms is provided in Table 1.

The framework is designed to accommodate multiple species and fisheries simultaneously, constructing risk profiles as a function of spatial and temporal overlap. Application has been primarily within the New Zealand Exclusive Economic Zone (EEZ; e.g., Richard & Abraham 2015, Richard et al. 2017, 2020), but, since seabirds migrate widely across the southern

hemisphere, a comprehensive assessment of the fisheries risk needs to account for all the fishing effort that may be encountered as the birds move through international waters. This has motivated application of the method in a wider context.

The SEFRA approach is a quasi-spatial model where temporal and spatial overlap of the seabird distribution and fishing effort are used to predict catch. Parameterisation of the capture rate per unit of overlap occurs via a fit to fisheries observer capture data, and total captures are calculated by multiplication of the total overlap (including the unobserved component) with this estimated rate (referred to as the catchability). Deaths are calculated from the predicted captures using a mortality multiplier that accounts for the probability of dead capture and cryptic mortality. Following estimation of the total deaths, the SEFRA approach often quantifies risk using a limit reference point referred to as the Population Sustainability Threshold (PST; Sharp 2019).

PST per species *s* is calculated as:

$$PST_{s} = \frac{1}{2} \cdot \varphi \cdot r_{s} \cdot N_{s} \tag{1}$$

where r_s is the theoretical unconstrained maximum population growth rate (i.e., under optimal conditions and in the absence of density dependent constraints), and N_s is the total population size, which we assume in the current setting to be the total number of adults. φ is an adjustment factor used by management to ensure that deaths equal to the PST correspond to a defined population stabilisation or recovery objective. In this risk assessment, φ was set to 1.

Risk ratios per species are calculated as:

$$risk ratio = \frac{D_s}{PST_s}$$
 (2)

However, this assessment only considers fishing using surface longlines by CCSBT members and, therefore, cannot estimate overall risk to the population from fishing. Since the PST reference point is designed to allow a measurement of risk, and includes management related tuning parameters, the comparison of deaths to the PST may be misleading. Following the 2024 CCSBT risk assessment, therefore, the 2025 SEFRA compares deaths with the theoretical maximum growth rate in numbers per year, i.e., $r_{\rm S} \cdot N_{\rm S}$, using relative mortalities defined as:

relative mortality =
$$\frac{D_s}{r_s \cdot N_s}$$
 (3)

The relative mortality approach typically provides the same relative ranking as that achieved using the PST reference point, because the φ term is commonly assumed to be the same for all species during comparative assessments.

3.2 Seabirds potentially at risk of capture in the CCSBT fishery

Estimates of seabird population size are typically reported as the number of breeding pairs per colony. The number of adults per species (*s*) was therefore calculated from the global sum of the number of breeding pairs and the probability of breeding:

$$N_s^{\text{adults}} = 2 \cdot \frac{N_s^{\text{BP}}}{P_c^{\text{B}}} \tag{4}$$

The number of adults available to be caught by longline fishing fleets of CCSBT Members during any month of the year was determined from the probability that they are in the southern hemisphere (SH), the probability that they are breeding, and whether they are likely to be attending the nest whilst doing so. The number of available adults per species and month (m) is:

$$N_{s,m} = N_s^{\text{adults}} \cdot \left(1 - P_s^{\text{B}} \cdot P_{s,m}^{\text{nest}}\right) \cdot P_{s,m}^{\text{SH}}$$
(5)

Outside the breeding season the probability of <u>attending the nest is typically breeding is-</u>zero, (i.e. $and P_{s,m}^{nest} = 0$), and all adults in the southern hemisphere are considered available to fishing gear.

The number of adults available for capture by CCSBT longline fleets ($N_{s,m}$; Equation 5) was used for predicting captures and fitting the model, whereas the total adult population size (N_s^{adults} ; Equation 4) was used for calculation of the risk ratios and relative mortalities.

3.3 Spatial overlap

The SEFRA model requires that the individuals available to be caught are represented as a spatial distribution. In this case, spatial distributions were estimated from tracking data (see Section 4.2). The spatial distribution is treated as a fixed data input and described using a density term $(d_{s,m,x})$ per species s, grid cell x and month m. Specifically, if $y_{s,m,x}$ is the number of birds in grid cell x and A_x is the area of grid cell x in square kilometres, then:

$$d_{s,m,x} = \frac{y_{s,m,x}}{A_x \cdot \sum_x y_{s,m,x}} \tag{6}$$

The value $y_{s,m,x}/\sum_x y_{s,m,x}$ is effectively being treated as the multinomial sampling probability of an individual being in grid cell x during that month. The absolute density, in number of birds per square kilometre, is therefore:

$$\mathbb{D}_{s,m,x} = d_{s,m,x} \cdot N_{s,m} \tag{7}$$

If fishing effort $(a_{f,m,x})$ for each fishery group f is allocated to grid cell x, and assuming a uniform distribution of birds and fishing effort within that cell, then we can construct an overlap metric that measures the opportunity for interaction between a bird population and fishing effort:

$$overlap_{f,s,m,x} = a_{f,m,x} \cdot d_{s,m,x}$$
(8)

The overlap provides a measure of the exposure of birds to fishing effort at a particular time and place, relative to the population as a whole. To estimate the catchability, SEFRA uses the density overlap, $\mathbb{O}_{f,s}$, given by:

$$\mathbb{O}_{f,s} = \sum_{m,r} a_{f,m,x} \cdot \mathbb{D}_{s,m,x} \tag{9}$$

The density overlap is a summation across grid cells and months, per species and fishery, and provides an input to the regression model.

3.4 Prediction of captures per species

Multiplication of the density overlap $(\mathbb{O}_{f,s})$ with the catchability $(q_{f,z})$ yields the model predicted captures per species and fishing fleet:

$$C_{f,s} = q_{f,z} \cdot \mathbb{O}_{f,s} \tag{10}$$

The catchability itself is a function of fishery group (f) and species group (z) covariates:

$$\log(q_{f,z}) = \beta_0 + \beta_f + \beta_{z|f} \tag{11}$$

where the fishery group coefficient β_f is centred on the intercept term, with deviations around this intercept constrained to sum to zero. Species group coefficients ($\beta_{z|f}$) were specific to the fishery group and were similarly constrained to sum to zero. This allowed the catchability per species group to deviate from the fishery group effect in a fishery group-specific manner.

The probability of live capture was a function of fishery group (f) and species group (z) covariates:

$$logit(\Psi_{f,z}) = \gamma_0 + \gamma_f + \gamma_{z|f}$$
(12)

where γ_0 is an intercept term and with coefficients γ_f and $\gamma_{z|f}$ similarly constrained to sum to zero. Predictor coefficients for the catchability (β_f and $\beta_{z|f}$) and live capture (γ_f and $\gamma_{z|f}$) were given standard normal priors, whereas the intercept terms β_0 and γ_0 , were given improper (unbounded) prior distributions.

3.5 Prediction of captures per capture code

The model predicts captures per species. However, observed captures of seabirds are not always identified to a species level. In order to fit the model to observed captures, it is necessary to assign the predicted captures per species to one or more capture codes that reflect the taxonomic resolution of identifications by observers (Table 3). For example, captures of Gibson's albatross may have been identified to a species complex level (capture code DWC), a genus (DIZ) or family (ALZ) level, or as an unspecified bird (BLZ).

A vector of probability terms is used to predict the captures per capture code: π , which are a set of probabilities describing the taxonomic resolution to which a species capture is identified (Edwards et al., 2025b). These probabilities are estimated per fishery group, but the f subscript is omitted for clarity of presentation:

$$\mathbf{\pi} = \{\pi_{\text{subgenus}}, \pi_{\text{genus}}, \pi_{\text{family}}, \pi_{\text{class}}\}$$
(14)

The 'subgenus' probability term refers to captures recorded at either the species-level or as part of a species complex (Table 3). This approach requires the condition that there is at most one capture code per species at each taxonomic resolution. This required adjusting the capture codes used for the initial data preparation, with the removal of species-specific capture codes for the royal albatrosses, i.e., southern royal (*Diomedea epomophora*) and northern royal albatross (*D. sanfordi*), and black-browed albatrosses, i.e., black browed (*Thalassarche melanophris*) and Campbell black-browed albatross (*T. impavida*). Otherwise, the capture codes used for the initial data preparation were consistent with those used in the 2024 risk assessment (Anon., 2024).

We can also define:

$$\pi_{\text{subgenus}}^{+} = \pi_{\text{subgenus}}$$

$$\pi_{\text{genus}}^{+} = \pi_{\text{subgenus}} + \pi_{\text{genus}}$$

$$\pi_{\text{family}}^{+} = \pi_{\text{subgenus}} + \pi_{\text{genus}} + \pi_{\text{family}}$$

$$\pi_{\text{class}}^{+} = \pi_{\text{subgenus}} + \pi_{\text{genus}} + \pi_{\text{family}} + \pi_{\text{class}} = 1$$

$$(15)$$

These are the cumulative probabilities, i.e., the probability of a capture being recorded at that taxonomic resolution or higher, or to "at least" that resolution. For example, for southern royal albatross, $\pi_{\rm genus}$ gives the probability that a captured individual was identified as a great albatross (DIZ), and $\pi_{\rm genus}^+$ the probability that the individual was identified as either a great albatross (DIZ) or an unspecified royal albatross (DRA).

The cumulative probabilities have the property that:

$$\pi_{\text{subgenus}}^+ \le \pi_{\text{genus}}^+ \le \pi_{\text{family}}^+ \le \pi_{\text{class}}^+$$
 (16)

As described above, the π and π^+ probability vectors are specific to a fishery group. Within each fishery group, the probability vectors can be shared amongst groups of species, e.g., shared amongst all species within a genus, family, or species group (z).

Using either the π or π^+ probability vectors we can now predict the observed captures per capture code from the model predicted captures per species. We use the following notation.

The observed data are:

- C_k : captures per capture code k, referred to as "empirical captures";
- C_k^+ : cumulative sum of the captures per capture code k (i.e., the sum of all observed captures to capture code k or a higher taxonomic resolution);

and the model predictions are:

- \hat{C}_s : captures per species s;
- \hat{C}_k : captures per code k;
- \hat{C}_k^+ : cumulative sum of the captures per code k.

The relationship between observations C_k and C_k^+ can be written explicitly using a twodimensional matrix. A simplified example is provided here (using capture codes from Table 3), assuming that only Gibson's albatross (DIW), Salvin's albatross (DKS), and sooty albatross (PHU) are being assessed. Note there is no species-level capture code for Gibson's albatross (DIW). In this example, the relationship between the observed captures per capture code (C_k) and the cumulative sum of the observed captures (C_k^+) is:

The relationship between the model predicted captures per species (\hat{C}_s ; right-hand side of equation) and the predicted captures per capture code (\hat{C}_k ; left-hand side of equation) is:

$$\begin{vmatrix}
\hat{C}_{DKS} \\
\hat{C}_{PHU} \\
\hat{C}_{DWC} \\
\hat{C}_{DIZ} \\
\hat{C}_{THZ} \\
\hat{C}_{THZ} \\
\hat{C}_{PHZ} \\
\hat{C}_{ALZ} \\
\hat{C}_{BLZ}
\end{vmatrix} = \begin{vmatrix}
0 & \pi_{\text{subgenus}(DKS)} & 0 \\
0 & 0 & \pi_{\text{subgenus}(PHU)} \\
\pi_{\text{subgenus}(DIW)} & 0 & 0 \\
0 & \pi_{\text{genus}(DIW)} & 0 & 0 \\
0 & \pi_{\text{genus}(DKS)} & 0 \\
0 & 0 & \pi_{\text{genus}(PHU)} \\
\pi_{\text{family}(DIW)} & \pi_{\text{family}(DKS)} & \pi_{\text{family}(PHU)} \\
\pi_{\text{class}(DIW)} & \pi_{\text{class}(DKS)} & \pi_{\text{class}(PHU)}
\end{vmatrix} \times \begin{bmatrix} \hat{C}_{DIW} \\ \hat{C}_{DKS} \\ \hat{C}_{PHU} \end{bmatrix}$$
(18)

which has the property that $\sum_k \hat{\mathcal{C}}_k = \sum_s \hat{\mathcal{C}}_s$ because each species-level predicted capture is partitioned between the possible capture codes using probabilities that sum to one; i.e., for each species, $\pi_{\text{subgenus}} + \pi_{\text{genus}} + \pi_{\text{family}} + \pi_{\text{class}} = 1$.

The relationship between model predicted captures per species (\hat{C}_s ; right-hand side of equation) and the cumulative sum of model predicted captures per capture code (\hat{C}_k^+ ; left-hand side of equation) is:

$$\begin{bmatrix}
\hat{C}_{DKS}^{+} \\
\hat{C}_{PHU}^{+} \\
\hat{C}_{DWC}^{+} \\
\hat{C}_{DIZ}^{+} \\
\hat{C}_{PHZ}^{+} \\
\hat{C}_{PHZ}^{+} \\
\hat{C}_{ALZ}^{+} \\
\hat{C}_{BLZ}^{+}
\end{bmatrix} = \begin{bmatrix}
0 & \pi_{\text{subgenus}(DIW)}^{+} & 0 & 0 \\
0 & 0 & \pi_{\text{subgenus}(PHU)}^{+} \\
0 & 0 & 0 \\
0 & \pi_{\text{genus}(DIW)}^{+} & 0 & 0 \\
0 & 0 & \pi_{\text{genus}(PHU)}^{+} \\
0 & 0 & \pi_{\text{genus}(PHU)}^{+} \\
0 & \pi_{\text{family}(DIW)}^{+} & \pi_{\text{family}(PHU)}^{+} \\
1 & 1 & 1
\end{bmatrix} \times \begin{bmatrix} \hat{C}_{DIW} \\ \hat{C}_{DKS} \\ \hat{C}_{PHU} \end{bmatrix}$$
(19)

which has the property that $\hat{\mathcal{C}}^+_{BLZ} = \sum_s \hat{\mathcal{C}}_s$. This is useful because the total number of bird captures is independent of the estimated π terms. Equality of model prediction $\hat{\mathcal{C}}^+_{BLZ}$ and the observed value \mathcal{C}^+_{BLZ} ensures that the model is accurately predicting the total number of bird captures.

3.6 Parameter estimation

Equations 18 and 19 can both be used to construct a likelihood for the model fit, based on predicted and observed 'empirical captures' (Equation 18), or predicted and observed cumulative captures (Equation 19). As described above, cumulative capture data have the property that the cumulative captures at a class level (\hat{C}_{BLZ}^+) is equal to the total catch across species, and is independent of the estimated π terms. In the 2024 risk assessment, the model was fitted to cumulative captures data, on the expectation that the sum of the captures should be a more reliable data point than captures at finer taxonomic resolutions. A consequence of the approach is that, when calculating the cumulative sum, the data are being pseudoreplicated. In the 2025 risk assessment, fits to cumulative and empirical captures were both explored, and predicted and observed cumulative captures compared for both likelihoods to ensure that the model was accurately predicting total captures.

The model was fitted to the capture data using a Poisson likelihood conditioned on either the cumulative captures:

$$C_k^+ \sim Poisson(\hat{C}_k^+)$$
 (20)

or empirical captures:

$$C_k \sim Poisson(\hat{C}_k)$$
 (21)

A Binomial likelihood function, conditioned on the number of captures for which life status was recorded, was used to estimate the probability of a capture being alive $(\Psi_{f,z})$.

Estimated parameters are listed in Table 1. Estimation of the vector of π values allows the model to predict \hat{C}_k^+ from \hat{C}_s and π^+ (Equation 19), as well as \hat{C}_k (Equation 18). Biological parameters $N_s^{\rm BP}$ and $P_s^{\rm BP}$ were estimated, with strongly informed priors, whereas $P_{s,m}^{\rm SH}$ and $P_{s,m}^{\rm nest}$ were fixed on input. Estimation of $N_s^{\rm BP}$ and $P_s^{\rm B}$ allows incorporation of uncertainty in these parameters (through the prior distribution), and is justified because these parameters are the most important determinants of the number of birds available for capture (Equations 4 and 5). The model is able to fit the captures data by changes in either $q_{f,z}$ or $N_{s,m}$, and by

estimating $N_{s,m}$ we can use it as a diagnostic of the model fit. In a correctly specified model, we would not expect $N_s^{\rm BP}$ or $P_s^{\rm B}$ to be updated from their prior values. If this occurs, it can indicate a deficiency in either the data or the structural assumptions, which can then be investigated. Usually, it would indicate that $q_{f,z}$ is constrained in a way that prevents it from adequately describing the data, requiring the model to update $N_{s,m}$ instead. If only minor updates occur, then these are incorporated directly into the estimates of risk ratios and relative mortality estimate, ensuring internal consistency. For the same reasons $S_s^{\rm opt}$ and $A_s^{\rm curr}$ (see below) are also estimated, because these are used internally by the model for estimation of $r_{\rm max}$. Similar to the other biological parameters, they are provided with informative priors, which we do not expect to be updated. If updates do occur, then this approach allows deficiencies in either the data or the model to be diagnosed, whilst maintaining consistency between the parameters required for calculation of the relative mortality.

All estimation was performed within a Bayesian framework using rstan (Stan Development Team 2020). Two chains were run for 1,000 iterations each, with the first half discarded. Posterior samples from estimated parameters were inspected visually to ensure convergence of the model. Assessment of the model fit to the data was based on comparisons of values of C_k^+ and \hat{C}_k^+ , and C_s^+ and \hat{C}_s^+ . Finally, we inspected updates to the biological inputs, particularly $N_s^{\rm BP}$ and $P_s^{\rm B}$. If either of these demonstrated strong prior updates then this would indicate model mis-specification or data deficiencies.

3.7 Prediction of total deaths

During the fitting process we estimate the catchability $q_{f,z}$ (Equation 11), which describes the rate of observed capture per unit of density overlap. Using this estimated value, we can then predict the total observable captures across all the fishing effort included in the assessment. However, observable captures are only a subset of the total captures resulting from the interaction between fishing effort and birds, as some captures are cryptic, i.e., unobservable even were an observer present.

To calculate the number of deaths from the number of observable captures we used a mortality multiplier ($\kappa_{f,z}$). We assume that captures that occur during setting invariably cause death by drowning, and can be lost (and so unobservable), but that live birds are caught during the haul and are always observable. To estimate the total number of deaths we therefore need $\kappa_{f,z}$ to account for drowned birds that are lost, and live birds that die post-release.

The probability of a bird being alive at capture ($\Psi_{f,z}$) was estimated as part of the model fit; for this assessment it was assumed that almost all seabirds that were caught subsequently died (post-release survival was given a mean value of $\omega=0.01$). For birds caught during setting and subsequently lost, it was decided to use the surface longline multipliers (K) from Edwards et al. (2023, see their Table 4), based on the analysis of the dataset from Brothers et al (2010) by Zhou et al (2019).

The total number of deaths for the surface longline fishery groups was therefore predicted from the estimated values of $q_{f,z}$ and $\Psi_{f,z}$ using:

$$D_{f,s} = q_{f,z} \cdot \mathbb{O}_{f,s} \cdot \left(\Psi_{f,z} \cdot (1 - \omega) + \left(1 - \Psi_{f,z} \right) \cdot K \right) \tag{22}$$

where:

$$\kappa_{f,z} = \Psi_{f,z} \cdot (1 - \omega) + \left(1 - \Psi_{f,z}\right) \cdot K \tag{23}$$

All deaths were generated using posterior predictive simulation from a Poisson distribution conditioned on the expected value. The number of total deaths per species is a summation of the deaths across fishery groups:

$$D_S = \sum_f D_{f,S} \tag{24}$$

The total deaths can then be compared against relative mortality to calculate the relative species-specific risk (Equation 3).

3.8 Theoretical maximum intrinsic growth rate (r_s)

For the relative mortality reference point, we are required to estimate a distribution for $r_s = \ln(\lambda_s)$. This was achieved using allometric theory. Following the approach of Niel & Lebreton (2005), and dropping species subscripts, mean generation time is first approximated as:

$$\bar{T} = A + \frac{S}{\lambda - S}$$

Allometric theory defines the optimal generation time such that:

$$T_{[opt]} \cdot \ln(\lambda) = k$$

where $k \approx 1$ is a constant. Therefore, under constant fecundity and assumed optimal conditions we can write:

$$\frac{k}{\ln(\lambda)} = A + \frac{S^{\text{opt}}}{\lambda - S^{\text{opt}}}$$

$$\Rightarrow \lambda = \exp\left(k \cdot \left(A + \frac{S^{opt}}{\lambda - S^{opt}}\right)^{-1}\right)$$
(25)

which can be solved numerically. This provides the so-called demographic-invariant solution for λ that has been used for all applications of the SEFRA methodology to date.

To implement this approach, we required information on the optimum survivorship (S_s^{opt}) and the current age at first breeding (A_s^{curr}), with the latter assumed to be indicative of the current environmental conditions. These were treated as estimated parameters within the model, each with strongly informative priors. In this way, local minimisation of Equation 25 (i.e., using a root finding algorithm to estimate λ), could be performed for each posterior sample of S_s^{opt} , A_s^{curr} , P_s^{B} and N_s^{BP} , to calculate the product $r_s \cdot N_s^{\text{adults}}$ as a model output.

3.9 Species groups and fisheries groups

The 2025 risk assessment covered all ACAP albatross species and *Procellaria* petrel species that primarily occur in the southern hemisphere (Table 2), representing 23 of the 31 ACAP albatross and petrel species. Here, Antipodean (*Diomedea antipodensis antipodensis*) and Gibson's albatross (*D. a. gibsoni*) as well as northern (*T. bulleri bulleri*) and southern Buller's albatross (*T. b. platei*) are considered separately, as they likely have different risk profiles, resulting in a total of 25 taxa under assessment. These species were grouped into "species groups" according to their ecology and behaviour. The catchability was shared across species within a species group, assuming that their vulnerability to fishing is determined by these shared behavioural characteristics. Five species groups were initially assumed: wandering albatross; royal albatross; mollymawks; sooty albatross; and medium petrels, with the definition of species groups refined as the risk assessment progressed (see Section 5). Following the 2024 SEFRA, *Macronectes* spp have been excluded because data are limited and

current conservation status is relatively favourable.

The fishery coverage of the assessment was defined as surface longline fisheries operated by the CCSBT members in the southern hemisphere, regardless of target species, in the period from 2012 to 2023 inclusive. Individual members of the CCSBT were each treated as one fishery group, except the joint-venture (JV) operations between New Zealand and Japan, and South Africa and Japan. These JVs were each treated as a separate fishery group to the domestic South African and New Zealand fleets, based on differences in their characteristics in Japanese operational style under strict management and surveillance under the joint venture arrangement.

4. DATA

4.1 Seabird biological input parameters

Biological data inputs to the risk assessment model include demographic parameters, generally represented with statistical distributions (referred to as priors), and information on the spatial distributions of the seabird taxa, included as point estimates without uncertainty. Demographic parameters with prior distributions are estimated during the model fit, whereas parameters represented as point estimates are fixed.

Biological inputs to the risk assessment model were reviewed by seabird researchers coordinated through ACAP in 2024 (Anon., 2024; Edwards et al., 2025a). Researchers were selected based on their publication record and known involvement with the species covered by the risk assessment. The review process included compilation of available information relevant to the demographic parameters of interest at a colony level. The review is summarised in **Appendix B**, along with a comprehensive overview of the biological inputs to the risk assessment model and data sources. Prior distributions for breeding pairs were updated this year (2025) for Gibson's albatross, Antipodean albatross, wandering albatross (*D. exulans*), southern royal albatross, black-browed albatross (*T. melanophris*), Campbell albatross (*T. impavida*), shy albatross (*T. cauta*), white-capped albatross (*T. cauta steadi*), Salvin's albatross (*T. salvini*), grey-headed albatross (*T. chrysostoma*) and southern Buller's albatrosses. Prior distributions for the probability of breeding were also updated for Gibson's albatross and Antipodean albatross.

The probability of breeding adults being on nest by month ($P_{s,m}^{\mathrm{nest}}$), and the probability of adults being in the southern hemisphere ($P_{s,m}^{\mathrm{SH}}$) are provided in Table 5 and Table 6. Summary statistics of the prior distributions for annual breeding pairs (N_s^{BP}), probability of adults breeding (P_s^{B}), current age at first reproduction (A_s^{curr}) and optimum survivorship (S_s^{opt}) are provided in Table 7. Summary statistics of prior values of total adult population size (N_s), theoretical unconstrained maximum population growth rate (r_s) are provided in Table 8.

It was cautioned that the bird population dynamics data are incomplete. ACAP reports that gaps in population data remain for globally significant breeding populations at sites that are logistically difficult to access and for species that are particularly difficult to census (ACAP, 2024). Nine albatross or petrel species on nine islands groups, estimated to hold >10% of the species' global population, have not had a population estimate in >10 years. Similarly, four species at seven island groups, which account for >5% of the species' total global breeding population, have not been censused since 2012. As an example, New Zealand is assumed to hold 33% of the world population of light-mantled sooty albatross (*Phoebetria palpebrata*), but, as this species is notoriously difficult to survey, population estimates rely on incomplete data from the 1970s and 1990s, depending on the island group. Other population parameters, such as breeding probability, are even more limited for these poorly surveyed populations. If parameters were unavailable for a given species (e.g., for breeding probability for Chatham Island albatross), a genus-level mean was used instead.

4.2 Seabird distribution information

Density maps used in the 2024 iteration of the risk assessment were taken from Devine et al. (2023), based on spatiotemporal 3-dimension generalized additive models (GAMs) fitted to tracking data. These density maps were reviewed in 2024 as part of the broader review of the biological inputs to the risk assessment model (see Appendix B). For some species, a lack of available tracking data was identified as an issue while, for other species, existing tracking data that had not yet been included was highlighted, resulting in absences of known foraging areas from density maps. Consequently, new density maps were generated for the species in need of improvements.

Density maps were generated for the 2025 SEFRA using a similar approach to Devine et al. (2023), but with refinements in response to feedback from the 2024 review (see Appendix A). First, available tracking data were weighted by the relative size of the colony before model fitting. This ensures that larger colonies have more influence on the species-level density maps. Second, additional tracking data were incorporated into the modelled datasets, including tracking data held by New Zealand's Department of Conservation¹, as well as data from Birdlife International's seabird tracking dataset². For species for which the available tracking data were limited (not all major colonies had data), distribution maps were augmented with mapping layers from Carneiro et al. (2020), weighted according to the proportion of breeding populations that had been tracked. After this step, only four species had distributions that lacked substantial data from the main colonies.

The density maps were for adults only, noting the difficulty in distinguishing older immatures and pre-breeders from adults for some species, even with necropsy (Lonergan et al, 2017). The working group noted that, ideally, the density maps would also cover juveniles and immature birds. However, for many taxa there were no available tracking data for these life stages. In this context, the working group agreed to continue with an 'adults only' approach to the risk assessment model. This approach is precautionary, in the sense that the estimated deaths (which could include sub-adults) resulting from fishing are compared with relative mortality calculated using only the number of adults.

The working group acknowledged that the incorporation of additional tracking data had partially resolved issues raised in the 2024 review relating to an absence of analysed tracking data from major colonies. The working group noted that the updated density maps addressed the absence of known foraging grounds for some taxa, e.g., regions off Western Australia and Chile for Campbell black-browed albatross. The working group also noted that the weighting of tracking data by colony size had appeared to improve the quality of density maps more generally, e.g., reducing the apparent over-estimation of densities of black-browed albatrosses off the Great Australian Bight.

4.3 Seabird bycatch and effort from surface longlines

The assessment utilised the observed monthly catch and effort data provided by the participating Members in the calendar years 2012 to 2023. The spatial resolution of input data was decided by each Member, though ultimately 5x5 degree cells were used in the model fitting. Individual Members compiled their own data using an R package provided by the modeling team. The Member-specific data submissions then collated into a combined dataset which was used to generate inputs for the risk assessment model.

In the 2024 risk assessment, information on observed captures and effort was limited to the

¹ https://docnewzealand.shinyapps.io/albatrosstracker/

² https://data.seabirdtracking.org/

longline fleets of Japan, New Zealand and the Fishing Entity of Taiwan. For the 2025 risk assessment, information was also provided by Australia, South Africa and Korea. These six Members provided observed catch and effort data, as well as total effort data for their surface longline fisheries operating in the southern hemisphere regardless of target species. Indonesia participated in the ERS Tech meeting, but were unable to provide input data for use in the 2025 risk assessment.

Summaries of observed and total effort by Members and fishery group are provided in Table 9 and Figure 1. Additionally, summaries of observed seabird captures are provided in Table 9, Table 10 and Table 11.

Onboard observer programs were impacted by movement constraints during the COVID-19 pandemic, resulting in reduced observer coverage from 2020 to 2022, particularly for fleets operating in the high seas.

Summaries of each Member's dataset

Australia

For Australia, seabird bycatch and effort data from longline vessels were sourced from 1) observer records for seabird bycatch and observed effort from 2012 to 2015, 2) electronic monitoring records for seabird bycatch and observed effort from 2016 to 2023, and 3) logbooks for total fishing effort from 2012 to 2023. All Australian tuna longline vessels were included in the same AU fleet. Electronic monitoring identification of seabird bycatch was coarser, and mostly to a family or higher taxonomic level, compared to observer identification which was mostly to a species level.

Japan

The input data for SEFRA is produced from Japanese observer data. The observer data from 2012 to 2023 were used; however, the data for 2021 and 2022 are absent due to the COVID-19 pandemic. Observed hooks, total number of bycaught seabirds, and number of seabird bycatch by species were used for the SEFRA input data. The observer usually starts observation from the beginning of the hauling operation and continues till that the observation duration becomes 80 % of the total hauling operation. Observed hooks were estimated from the ratio of research duration to duration of hauling operation. While the individuals with a DNA sample were identified by DNA, others were identified from a photo. The individual that was not identified till species was identified as a species group. The fate of individuals was also collected for each species. The data was aggregated by year, month/quarter, and 5x5 degree strata. The total number of bycaught seabirds and the number of seabird bycatch by species were aggregated additionally by fate. Total effort data were extracted from logbook data using the same time period and resolution as the observer data.

Korea

[Paragraph to be provided by Korea]

New Zealand

Assessments of the capture of protected species in New Zealand commercial fisheries rely on observer and fisher-reported data. Fisheries observers document the captures of protected species, and these observer records are linked to fisher-reported effort data. To improve species resolution all captures were first identified by the observer, and some captures were subsequently necropsied. If a bird has been necropsied, then this identification was used in preference. For birds that were not necropsied, an expert identification based on a photograph was used in preference. Finally, for birds that were neither photographed nor necropsied, an imputation process was used (Thompson et al. 2017). New Zealand data comprised that of both domestic fisheries and that undertaken by the Japanese joint venture fleet. A total of 611

observed captures that occurred during the 2012–23 calendar year were included in the New Zealand reporting reporting tables. Of these captures, 412 were Thalassarche, 152 were Procellaria, 47 were Diomedia, and none were Phoebetria.

South Africa

[Waiting for response from South Africa]

Taiwan

The seabird bycatch and effort data from Taiwanese longline vessels spanned 2012 to 2023, and were sourced from two datasets: 1) observer records for seabird bycatch and observed effort, and, 2) logbooks and e-logbooks documenting fishing effort. All Taiwanese tuna longline vessels, regardless of size or target species, were considered the same fleet (TW). While the observer data aimed to identify seabird bycatch to the species level, Gibson's albatross was not differentiated from other species, likely resulting in being recorded as Antipodean albatross or similar species. Observers were restricted to a maximum of eight working hours during hauling, resulting in incomplete hook observations. Hence, the observed number of hooks were provided. Fishing effort data consisted of logbook-recorded number of hooks set from 2012-2016, while e-logbook data provided effort information for 2017 onwards following e-logbook implementation in 2017.

5. RESULTS

The approach taken for the 2025 risk assessment was to first explore alternative modelling approaches fitted to 2024 biological and fishery inputs. This allowed separation of the impacts of changing the modelling approach from the impacts of updating the data inputs to the risk assessment, including the incorporation of the data inputs from Members that contributed data to the 2024 risk assessment.

The 2024 risk assessment was hampered by biologically implausible posterior updates to the number of breeding pairs and the probability of adults breeding for some taxa, with particularly strong updates for a number of mollymawk species (Anon., 2024). These posterior updates allowed the model to fit to captures data by changing the estimated availability of birds. These updates were required because taxa within a species group share estimated catchability terms $(q_{f,z})$, so improvements to model fits for taxa within a species group can only be achieved through changes in the availability of birds, i.e., by increasing $\mathbb{O}_{f,s}$. As such, a particular focus of developments to the risk assessment model was reviewing data inputs to identify potential causes for the strong posterior updates, and testing approaches intended to reduce the strength of these posterior updates.

5.1 Initial model runs and exploratory analyses with inputs to the 2024 risk assessment

The working group selected five one-off changes to the 2024 risk assessment model, with each applied to the data inputs to the 2024 risk assessment. These model runs are described below.

a) Fitting the model to empirical captures

Fitting the model to empirical captures substantially improved the model fit to captures data, most notably reducing the over-estimation of captures identified at coarse taxonomic resolutions, i.e., captures identified to a family level, or recorded as an unspecified bird. Fitting to empirical captures is preferred from a theoretical basis, as there is no need for pseudoreplication of captures data. However, there was no material change in the strength of

posterior updates to the number of breeding pairs ($N_s^{\rm BP}$) or the probability of breeding ($P_s^{\rm B}$), and the estimated catchabilities were insensitive to the change.

b) Composite density maps based on the weighted average of the 2024 density maps and range maps

These composite maps can be interpreted as the use of the density maps for colonies which contributed tracking data in the modelled datasets used to estimate density maps, and the use of range maps for colonies with no available tracking data. The use of the composite maps reduced the prevalence of "zero overlap captures" but did not reduce the strong posterior updates to the number of breeding pairs $(N_s^{\rm BP})$ or the probability of breeding $(P_s^{\rm B})$.

c) Use a single species group for catchabilities for great albatross species

In the 2024 risk assessment, the great albatrosses were split into two species groups, a wandering albatross group, and a royal albatross group. Fitting to genus-level capture data was identified as an avenue of exploration in the 2025 risk assessment (run d). This would assume that there is no information in the captures data to support estimation of sub-genus catchabilities. Model run c was used to assess the impact of collapsing the great albatross species groups into one (in isolation). There were minor changes to the estimated catchabilities with the change, but no material degradation of model fits. This likely reflects the limited captures of royal albatrosses in the 2024 dataset (36 individuals).

d) Fitting to genus-level captures data

Capture data identified to a species or complex level were reassigned to genus-level capture codes. This resulted in 7 capture codes (Table 4) and a truncated π vector:

$$\mathbf{\pi} = \{\pi_{\text{genus}}, \pi_{\text{family}}, \pi_{\text{class}}\}$$

The conversion matrix for calculation of cumulative captures is provided in Table 12. As described above, the two great albatross species groups were also combined, resulting in four genus-level species groups, i.e., great albatrosses, mollymawks, sooty albatrosses, and medium petrels. Fitting the model to genus-level captures data greatly reduced the strength of posterior updates to $N_s^{\rm BP}$ and $P_s^{\rm B}$, with no updates that were considered to be biologically implausible. There was an increase in catchabilities for the mollymawk group, which appeared to compensate for the reduction in density overlap in the absence of the artificial increases in population size from updates to $N_s^{\rm BP}$ and $P_s^{\rm B}$.

e) Genus-specific π vectors

In the 2024 risk assessment, the π vectors were specific to a fishery group, but were shared among all 25 taxa. This assumption may not be appropriate if some taxa are more difficult to identify to finer taxonomic resolutions than others, e.g., similar physical characteristics, rarity of interaction with vessels and so a lack of familiarity on the part of observers, etc. The model was refitted with genus (and fishery group) specific π vectors. There were relatively minor changes to estimated catchabilities, but there was some evidence for differences in the identifiability of captures between genera, with a higher probability of sub-genus identifications for mollymawks and medium petrels compared with great albatrosses and sooty albatrosses. This had the added benefit of being a useful tool for assessing improvements in species identification between time periods for each of the fishing groups.

f) Other trials

Preliminary model runs with species-fishery group interaction terms in the catchability equation were also explored, i.e. with:

$$\log(q_{f,z}) = \beta_0 + \beta_f + \beta_z + b_{f,s}$$

where $b_{f,s} \sim N(0,\sigma)$. This approach allows for variation in catchabilities among taxa within a fishery group. These deviations can also account for errors in estimated overlap resulting from inaccuracies in density maps, and should not result in biased estimates of total catch if observed effort is representative of total effort (spatially and temporally). The introduction of species-fishery group interaction terms in the catchability equation resolved the strong posterior updates to the number of breeding pairs ($N_s^{\rm BP}$) and probability of breeding ($P_s^{\rm B}$). However, this approach is also susceptible to bias resulting from errors in identifications of captures. In this context, the working group preferred the approach of fitting to genus-level captures data.

5.2 Exploratory analyses of data inputs to the 2024 risk assessment

Targeted examination of data inputs to the 2024 risk assessment was conducted concurrently with the initial model runs, to explore potential drivers for the strong updates to demographic parameters encountered in 2024. The working group noted that there were observed captures for a range of taxa that occurred in areas with zero density overlap. These "zero overlap captures" reflect an inconsistency between the capture data and the assumed adult distribution of the relevant populations. Errors in identifications, captures of sub-adults, and errors in the assumed spatial distributions both have the potential to drive posterior updates to the number of breeding pairs ($N_s^{\rm BP}$) or the probability of breeding ($P_s^{\rm B}$), as errors in observed catch and observed overlap (Equation 8) both influence the estimation of catchabilities.

Composite density maps were created by taking the weighted average of the 2024 density maps and range maps (BirdLife International & Handbook of the Birds of the World, 2024), with the density maps weighted by the proportion of breeding pairs from colonies with modelled tracking data. This reduced the prevalence of "zero overlap captures", from 418 to 72 individuals out of a total of 7,537. However, the use of the composite maps did not materially reduce the strength of posterior updates to $N_s^{\rm BP}$ and $P_s^{\rm B}$ by itself.

The 2024 risk assessment model was rerun with the updated density maps prepared for the 2025 risk assessment. This resulted in reductions in posterior updates for a range of taxa: Indian yellow-nosed albatrosses (primarily P_s^B), black-browed albatross (N_{BP} and P_s^B), New Zealand white-capped albatross (N_s^{BP} and P_s^B), Westland petrel (P_s^B) and spectacled petrel (P_s^B). More modest reductions in posterior updates were observed for Campbell black-browed albatross (P_s^B), Southern Buller's albatross (P_s^B), grey petrel (P_s^B), with a modest increase in posterior updates to N_s^{BP} and P_s^B for Salvin's albatross. However, biologically implausible posterior updates remained for white-chinned petrel, Campbell black-browed albatross, greyheaded albatross, southern Buller's albatross, light-mantled sooty albatross and Westland petrel, with more modest updates for grey petrel and spectacled petrel.

For the taxa with remaining biologically implausible posterior updates, additional sources of information on spatial distributions were examined to assess consistency with the estimated density maps, including eBird sightings data (Sullivan et al. 2019) and tracking datasets in Birdlife's Seabird Tracking Database that were not available for use in the estimation of density maps. In general, there was no clear evidence of inconsistencies in the density maps when compared with the sightings data and additional tracking data. However, there was some

evidence of an underestimation of grey-headed albatross in the Tasman Sea and further south. Additional tracking at Campbell Island and the larger Indian Ocean colonies may address this in the future.

The working group noted that the apparent inconsistency between capture data and the assumed spatial distributions could reflect captures of juveniles and immatures, given that the density maps are for adults only (Section 4.2).

5.3 Selected model

Based on the exploratory analyses and initial model runs using data inputs to the 2024 risk assessment, the working group decided that:

- Fitting to empirical captures was preferred to fitting to cumulative captures, due to superior model fits and the lack of pseudo-replication in captures data.
- Density maps should be combined with range maps (BirdLife International and Handbook of the Birds of the World, 2024) to account for colonies with no available tracking data.
- Models should be fitted to genus-level captures data. Identification of seabirds to a species level at-sea is difficult, particularly if the individual is waterlogged or damaged. It was considered likely that there are errors in identifications in the analysed dataset, particularly when based on at-sea identifications rather than those based on photos or necropsies by experts. Fitting to genus-level captures data is a compromise, in mitigating against bias from errors in identifications at fine taxonomic resolutions, whilst still providing sufficient information to account for variability in catchabilities between taxa.
- As a result of the aggregation of captures data to a genus resolution, the great albatross species groups should be combined, giving four genus level species groups.
- Genus (and species group) specific π vectors should be preferred, subject to confirmation that the updated 2025 dataset provided sufficient information for robust estimation of genus-specific π vectors for all fishery groups.

This model is referred to throughout this report as the "selected 2025 risk assessment model". The use of genus-level captures data, in combination with composite maps based on density maps and range maps, reduced the prevalence of "zero overlap captures" (6 from a total 9,815 captures; Table 13). Overlap from observed and total effort per species and fishery group is provided in Table 14 and Table 15 respectively.

MCMC trace diagnostics (e.g., Figure 2), and \hat{r} (< 1.05)were acceptable for model parameters with minimal posterior updates to both the number of breeding pairs ($N_s^{\rm BP}$; Figure 3) and the probability of breeding ($P_s^{\rm B}$; Figure 4). Model fits to empirical captures were acceptable (Figure 5, Figure 6, Table 17), and comparisons of cumulative captures indicated that the model was also accurately predicting total observed captures.

Estimated catchabilities demonstrated strong variability between fishery and species groups (Table 18, Figure 7), with wide credible intervals for fishery group and species group combinations with no, or less frequent, captures, e.g. sooty albatrosses.

Estimated π vectors demonstrated strong differences between fishery groups (Figure 8), which may reflect differences in sources of identifications. For example, New Zealand's fishery groups have relatively high probability of genus-level identifications for genera with observed captures, which may reflect the use of necropsy-based identifications. In contrast, the probabilities of genus-level identifications for the Australian fishery group were relatively low, which reflects difficulties in acquiring more resolved species identifications based on

electronic monitoring footage alone. Across the fishery groups, there was also a tendency for higher probabilities of genus-level identifications for *Thalassarche* and *Procellaria* species.

Estimated total mean annual deaths, cryptic deaths, and relative mortalities are provided in Table 19. Estimated relative mortality were typically highest for *Diomedea* species, and lowest for the *Procellaria* species (Figure 9). The species with the highest estimated relative mortality were (in descending order): Gibson's albatross (0.72, 95% CI 0.48–1.14), Amsterdam albatross (0.38, 0.25–0.60), Tristan albatross (0.36, 0.24–0.55), Sooty albatross (0.32, 0.20–0.49) and New Zealand white-capped albatross (0.24, 0.16–0.38). Cryptic mortality rates were effectively the same for all species, given the assumption that all birds were assumed to be dead at-vessel, and so relative mortality rankings were equivalent when considering relative mortality from "observable" deaths only (Figure 10).

The spatial distribution of total estimated deaths per species group is provided in Figure 13, with further breakdowns by fishery group provided in Figure 14. The spatial distribution of the mean relative mortality across all species is provided in Figure 15, with species-group specific mean relative mortality available in Figure 16. These maps identify a number of relatively small regions that contribute a high proportion of both estimated deaths and relative mortality, including the Tasman Sea for great albatrosses, mollymawks and sooty albatrosses, as well as the southeast Atlantic for Sooty albatrosses and the east Pacific for petrels.

5.4 Sensitivity run, with family-specific π vectors

A sensitivity run was undertaken based on the selected 2025 risk assessment model but with family-specific π vectors (rather than genus-specific) to assess the sensitivity of outputs to this decision. The quality of model fit was similar to the selected risk assessment model, with no material degradation in model fits resulting from the simplification of the π vector specification. Estimated catchabilities were insensitive to the change in the π vector specification (not shown). Estimated deaths for *Procellaria* petrels were least impacted by the change in π vector specification (Table 20). Estimated mean annual deaths for the great albatrosses and sooty albatrosses were slightly reduced with family-specific π vectors, with a slight increase in estimated deaths for mollymawks. However, these changes to estimated deaths did not materially impact the rankings of estimated relative mortality.

5.5 Selected model fitted to data from 2012 to 2019

To provide a more direct comparison with the 2024 risk assessment model, the selected 2025 risk assessment model was also fitted to data from 2012 to 2019, to match the time series used in 2024. Using this restricted data set, the quality of model fit was similar to the selected risk assessment model but the estimated mean annual deaths and relative mortality increased for all taxa (Table 21). It seems most likely that the higher estimates of mean annual deaths using the restricted data set were driven by differences in catchabilities for some fleets (see Section 5.6).

5.6 Model with temporally varying catchabilities and π vectors

The selected 2025 risk assessment model was also refitted with time-blocked catchability parameters and π vectors, to assess evidence for potential temporal changes in capture rates. Three time periods were assumed: 2012 to 2016; 2017 to 2019; and 2020 to 2023.

There was evidence for increased probabilities of identifications to a finer taxonomic resolution through time (Figure 17), which may reflect increasing seabird-related training for at-sea observers, as well as a move to photo-based identifications by experts (e.g., for the

Japanese fishery group). There were also reductions in estimated catchabilities through time for a number of fishery groups (Figure 18), including: mollymawk catchabilities for the domestic New Zealand fishery group, and South African domestic and Joint Venture fishery groups; Japan's fishery group catchabilities for all species groups, particularly in the period 2020 to 2023. There were also increasing temporal trends through time, including *Procellaria* petrel catchabilities for New Zealand's domestic and South Africa's Joint Venture fishery groups. The working group did note that interpretation of temporal changes in catchability effects is complicated by the time-invariant nature of the biological inputs, as catchabilities are confounded with the size of population available for capture in fisheries.

5.7 Comparisons of model results with previous risk assessments

It is difficult to make direct comparisons between the 2024 and 2025 risk assessments, given the differences in the modelling approach and data inputs. However, the outputs of the two risk assessments are broadly consistent with each other, in terms of the species rankings of estimated risk ratios from the 2025 risk assessment and the 'relative mortalities' from the 2024 risk assessment (Table 22).

The most influential change implemented in the 2025 risk assessment was fitting the model to captures data with genus-level (or higher) taxonomic resolutions. This removed the biologically implausible posterior updates to the number of breeding pairs and probability of breeding that were observed in the 2024 risk assessment, with a corresponding improvement in estimates of relative mortality. This can most clearly be seen for a number of mollymawk species, including Campbell black-browed albatross, grey-headed albatross and southern Buller's albatross (Table 22). The estimated deaths are also markedly different for some species, e.g., the order of magnitude decrease for grey-headed albatross in the absence of the (artificial) increase in adult population size through posterior updates to biological parameters, and the order of magnitude increase for Atlantic yellow-nosed albatross driven by increased overlap with the updated density maps.

More generally, the results of the 2025 risk assessment results are also consistent with previous iterations (Abraham et al. 2019; Anon et al. 2024, Peatman et al. WCPFC report), and other studies (Richard et al. 2024), including the 1) relatively high risk to species from the wandering albatross complex, 2) the higher risk in the Tasman Sea, and 3) the consistent assessment of Gibson's albatross, Amsterdam albatross, Tristan albatross and Sooty albatross as being among the taxa at highest risk.

6. CONCLUSIONS, REMAINING ISSUES and NEXT STEPS

6.1 Progress since the 2024 SEFRA

- This report summarises a quantitative risk assessment for 25 taxa of seabirds caught in surface longline fisheries (no matter the target species) by six participating Members of the Commission for the Conservation of Southern Bluefin Tuna.
- This assessment is based on the Spatially Explicit Fisheries Risk Assessment (SEFRA) approach and builds on the SEFRA model developed collaboratively by Members and discussed by the CCSBT's ERSWG15 in April 2024. Diagnostics suggest that the 2025 SEFRA model had converged, fitted the data very well, and did not appreciably update any of the priors for biological inputs.
- The key concerns raised about the 2024 iteration have been resolved in the 2025 iteration in that: conflicts between estimated overlap of fishing and seabirds and the observed captures (i.e., the presence of observed captures where the estimated overlap was zero) have been resolved; and the fitted models no longer require implausible updates to the priors on population size or the probability of breeding in a year.
- Modifications to the 2024 SEFRA model were made one at a time such that the impact of each change to data and model structure could be assessed.
- Updating the available information on seabird distributions reduced the number of "zero overlap captures" somewhat but fitting to genus-level (or higher) capture data (as opposed to species-level, where available) was by far the most influential change.
- The broad patterns of estimated risk were similar in the 2024 and 2025 SEFRA models; Gibson's albatross, Amsterdam albatross, Tristan albatross and Sooty albatross were the taxa estimated to be at highest risk in both model iterations. However, the estimated risk for many seabird taxa was higher in the 2025 SEFRA than in the 2024 SEFRA. The lower risk for many taxa in the 2024 SEFRA is thought to be largely an artefact caused by the updates to biological priors.
- At the scale of 5-degree squares, estimated annual deaths of great albatrosses and mollymawks were highest in the Tasman Sea although there were other higher-catch areas in the south-eastern Indian Ocean and the south-eastern Atlantic Ocean. Sooty albatross deaths were highest in the south-eastern Atlantic Ocean and, to a lesser extent, in the Tasman Sea. Deaths of medium petrels were highest around South Africa and off Namibia, and in the south-eastern Pacific Ocean. Spatial patterns vary among taxa finer than these four groups.
- The 2025 SEFRA was not very sensitive to fitting to family-level (or higher) capture data (as opposed to genus-level in the base case or species-level, where available, in 2024); the average absolute change to the estimated risk ratios was <10% (compared with \sim 40% for the change between the 2024 and 2025 SEFRA models).
- The 2025 SEFRA fitted to data from 2012 to 2019 (as for the 2024 SEFRA) had consistently higher estimates of risk (averaging about 20% higher) than the model fitted to the whole time series 2012 to 2023. This is thought to be due to lower catchability in more recent years.
- Diagnostics and inspection of results suggested that a SEFRA model with different time blocks (among which catchability was allowed to vary) had converged, fitted the data very well and provided useful estimates of the taxonomic level of identification of captured birds and catchability / total deaths. Catchability for great albatrosses and mollymawks was somewhat lower in the latest time block (2020 to 2023) for New Zealand domestic and Japanese fleets although there were mixed results for the other fleets and seabird taxa with few clear trends.

6.2 Uncertainties and caveats for the 2025 SEFRA model

- All SEFRA models are highly reliant on information on the distribution of seabirds. Better distributions than were available in 2024 were used but these are not perfect. Some recorded captures occur outside the predicted distributions, mostly at a subgenus level. This was interpreted as an indication that the overlap estimates were inconsistent with the captures at a sub-genus level, which could explain the strong updates to biological inputs observed in the 2024 SEFRA. This inconsistency could come from incorrect bird identifications, captures of sub-adults, or poor distribution maps, both of which may remain as issues.
- The 2025 SEFRA uses captures aggregated to genus level. This mitigates against potential bias due to misidentification of seabirds at finer taxonomic resolutions, and substantially reduces the number of captures outside of predicted distributions. The estimated captures are then disaggregated to species level based on the estimated overlap, relying heavily on the distribution of individual bird species and the quantum and distribution of total fishing effort. There may be some potential to use verified identifications of captures to enhance disaggregation in future.
- Juveniles, immature birds and pre-breeding birds may have different spatio-temporal
 distributions to adults and are likely to have higher catchability than adults, although
 data are not available to split captures by life stage. A precautionary approach has been
 adopted by assuming all captures are adults and captures are compared with the adult
 population size.
- The model is highly reliant on observer (or electronic monitoring) data, including bird identifications being correct at the genus level and accurate recording of captures and observed effort. Calculating total deaths assumes there is no "observer effect" on fisher behaviour.
- Catchability is assumed constant in space and (except for time-blocked model) in time, and within genus. There is limited data to explore this in relation to both yearly estimates of population for all seabirds included in the model and sufficient capture information for all fisheries groups. As such, we have not explored this but, if there were broad-scale differences in catchability, this would cause bias. Gaps in observer coverage were found to degrade precision of estimates on the 2024 SEFRA but were not assessed to cause bias.
- Although cryptic mortality is known to occur, the available information to calculate appropriate scalars is relatively sparse and relates only to birds hooked during setting. Similarly, the survival of birds that escape or are released alive is relatively poorly understood; as a precautionary approach, all captures are assumed dead.
- The time-blocked models assume constant biological inputs (population size and productivity) across all time blocks. In reality, population size, productivity or distribution may vary, leading to some potential bias in estimates of catchability or captures.

6.3 Next steps

The transition to phase 2, the global (southern hemisphere) risk assessment under the CCSBT Seabird Project, funded by FAO/GEF Common Oceans Program, started immediately following the ERS-Tech meeting in April 2025. In practice, it is expected that the global southern hemisphere assessment will use the model finalised at that meeting without any modifications.

Data would be sought from other nations fishing in the southern hemisphere in a process led by the Project Manager of the CCSBT Seabird Project, Dr Ross Wanless, coordinating with interested Members and the project partner BirdLife International.

An update on progress with the CCSBT SEFRA, and the transition to the global assessment, will be reported to the Extended Commission of the Thirty-Second Annual Meeting of the Commission (6–9 October 2025), although the format for this update has yet to be determined. Therefore, formally, this technical report and description of the 2025 SEFRA will be made available outside CCSBT only after the completion of the Extended Commission. However, this does not prevent the CCSBT Seabird Project Manager from engaging with potential data-contributing, non-CCSBT Members immediately, noting data confidentiality arrangements within CCSBT. The agreed 2024 SEFRA Technical Report (Attachment 4 of ERSWG 15 report) can be used as the base material, noting the substantial progress made during the 2025 ERS-Tech process.

The SEFRA model can be updated at any time when new information becomes available. No timetable for such updates is presented here although it is anticipated that the risk assessment will be updated periodically as may be required by the CCSBT Multi-year Seabird Strategy.

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Tables

Table 1: Glossary of model terms.

Notation	Description
Subscripts	
f	Fishery group
S	Species
z	Species group
k	Capture code
m	Month
x	Spatial location or grid cell
Estimated parameters	
$N_{\scriptscriptstyle \mathcal{S}}^{\;\mathrm{BP}}$	Number of breeding pairs
$P_{_{\mathcal{S}}}{}^{\mathrm{B}}$	Annual probability of breeding
$\mathcal{S}_s^{\mathrm{opt}}$	Annual optimum survivorship
$A_s^{ m curr}$	Current age at first breeding
eta_0 , eta_f , $eta_{z f}$	$q_{f,z}$ regression coefficients
$\gamma_0, \gamma_f, \gamma_{z f}$	$arPsi_{f,z}$ regression coefficients
$oldsymbol{\pi}_f$	Vector of capture assignment probabilities
Derived parameters	
$N_{\scriptscriptstyle S}^{ m \ adults}$	Total number of adults
$N_{s,m}$	Number of adults available to fishing
N_{s}	$N_{s,m}$ summed across months
$\mathbb{D}_{s,m,x}$	Density of adults available to fishing
$q_{f,z}$	Catchability
$\Psi_{f,z}$	Probability of capture being alive
$C_{f,S}$	Number of captures per species
$C_{f,k}$	Number of captures per capture code
$\kappa_{f,z}$	Mortality multiplier
$D_{f,s}$	Number of deaths
Inputs covariates	
$P_{s,m}^{\mathrm{SH}}$	Probability of an adult being in the southern hemisphere
$P_{s,m}^{ m nest}$	Probability of a breeding adult being on the nest
$d_{s,m,x}$	Relative density of adults per square kilometre
$a_{f,m,x}$	Fishing effort
K	Cryptic mortality multiplier
ω	Probability of post-release survivorship
Derived covariates	
$\mathbb{O}_{f,s}$	Density overlap

Table 2: Species and species groups used in the southern hemisphere risk assessment model. Species codes are from the FAO-ASFIS species list where possible (https://www.fao.org/fishery/en/species/search). The species group definitions provide a covariate input for estimation of the catchability.

Code	Common name	Scientific name	Species group
DIW	Gibson's albatross	Diomedea antipodensis gibsoni	Great albatross
DQS	Antipodean albatross	Diomedea antipodensis antipodensis	Great albatross
DIX	Wandering albatross	Diomedea exulans	Great albatross
DBN	Tristan albatross	Diomedea dabbenena	Great albatross
DAM	Amsterdam albatross	Diomedea amsterdamensis	Great albatross
DIP	Southern royal albatross	Diomedea epomophora	Great albatross
DIQ	Northern royal albatross	Diomedea sanfordi	Great albatross
DCR	Atlantic yellow-nosed albatross	Thalassarche chlororhynchos	Mollymawk
TQH	Indian yellow-nosed albatross	Thalassarche carteri	Mollymawk
DIM	Black-browed albatross	Thalassarche melanophris	Mollymawk
TQW	Campbell black-browed albatross	Thalassarche impavida	Mollymawk
DCU	Shy albatross	Thalassarche cauta	Mollymawk
TWD	New Zealand white-capped albatross	Thalassarche cauta steadi	Mollymawk
DKS	Salvin's albatross	Thalassarche salvini	Mollymawk
DER	Chatham Island albatross	Thalassarche eremita	Mollymawk
DIC	Grey-headed albatross	Thalassarche chrysostoma	Mollymawk
DSB	Southern Buller's albatross	Thalassarche bulleri bulleri	Mollymawk
DNB	Northern Buller's albatross	Thalassarche bulleri platei	Mollymawk
PHU	Sooty albatross	Phoebetria fusca	Sooty albatross
PHE	Light-mantled sooty albatross	Phoebetria palpebrata	Sooty albatross
PCI	Grey petrel	Procellaria cinerea	Medium petrel
PRK	Black petrel	Procellaria parkinsoni	Medium petrel
PCW	Westland petrel	Procellaria westlandica	Medium petrel
PRO	White-chinned petrel	Procellaria aequinoctialis	Medium petrel
PCN	Spectacled petrel	Procellaria conspicillata	Medium petrel

Table 3: Capture codes used in the preparation of data inputs for the 2025 southern hemisphere risk assessment model.

			Taxonomic
Code	Common name	Scientific name	resolution
DKS	Salvin's albatross	Thalassarche salvini	Species
DER	Chatham Island albatross	Thalassarche eremita	Species
DIC	Grey-headed albatross	Thalassarche chrysostoma	Species
PHU	Sooty albatross	Phoebetria fusca	Species
PHE	Light-mantled sooty albatross	Phoebetria palpebrata	Species
PCI	Grey petrel	Procellaria cinerea	Species
PCN	Spectacled petrel	Procellaria conspicillata	Species
DRA	Royal albatrosses	Diomedea epomophora & D. sanfordi	Complex
DYN	Yellow-nosed albatrosses	Thalassarche chlororhynchos & T. carteri	Complex
DST	Shy-type albatross	Thalassarche cauta & T. c. steadi	Complex
DBB	Black-browed albatrosses	Thalassarche melanophris & T. impavida	Complex
DIB	Buller's albatross	Thalassarche bulleri bulleri & T. bulleri platei	Complex
DWC	Wandering albatross complex	Diomedea exulans, D. dabbenena, D. amsterdamensis, D. antipodensis gibsoni & D. a. antipodensis	Complex
PRZ	Procellaria petrel complex	Procellaria parkinsoni, P. westlandica & P. aequinoctialis	Complex
DIZ	Diomedea spp.	Diomedea spp.	Genus
THZ	Thalassarche spp.	Thalassarche spp.	Genus
PHZ	Phoebetria spp.	Phoebetria spp.	Genus
PTZ	<i>Procellaria</i> spp.	Procellaria spp.	Genus
ALZ	Diomedeidae	Diomedeidae	Family
PRX	Procellariidae	Procellariidae	Family
BLZ	Bird	-	Class

 $Table\ 4: Capture\ codes\ used\ in\ the\ 2025\ southern\ hemisphere\ risk\ assessment\ model.$

Code	Common name	Scientific name	Taxonomic resolution
DIZ	Diomedea spp.	Diomedea spp.	Genus
THZ	Thalassarche spp.	Thalassarche spp.	Genus
PHZ	Phoebetria spp.	Phoebetria spp.	Genus
PTZ	Procellaria spp.	Procellaria spp.	Genus
ALZ	Diomedeidae	Diomedeidae	Family
PRX	Procellariidae	Procellariidae	Family
BLZ	Bird	-	Class

Table 5: Probability of a breeding adult being on nest by month ($P_{s,m}^{\text{nest}}$). Darker shaded cells indicate a higher probability.

Common name	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Gibson's albatross	0.50	0.50	0.50	0.40	0.05	0.05	0.05	0.05	0.05	0.05	0.05	0.22
Antipodean albatross	0.40	0.50	0.45	0.45	0.05	0.05	0.05	0.05	0.05	0.05	0.05	0.20
Wandering albatross	0.50	0.50	0.40	0.20	0.05	0.05	0.05	0.05	0.05	0.05	0.05	0.40
Tristan albatross	0.60	0.50	0.50	0.50	0.30	0.30	0.05	0.05	0.05	0.05	0.05	0.40
Amsterdam albatross	0.05	0.40	0.50	0.50	0.40	0.30	0.05	0.05	0.05	0.05	0.05	0.05
Southern royal albatross	0.50	0.50	0.40	0.05	0.05	0.05	0.05	0.05	0.05	0.00	0.40	0.50
Northern royal albatross	0.50	0.40	0.30	0.05	0.05	0.05	0.05	0.05	0.00	0.40	0.50	0.50
Atlantic yellow-nosed albatross	0.30	0.20	0.10	0.05	0.00	0.00	0.00	0.50	0.60	0.50	0.50	0.50
Indian yellow-nosed albatross	0.20	0.10	0.05	0.05	0.00	0.00	0.00	0.10	0.50	0.50	0.40	0.40
Black-browed albatross	0.20	0.05	0.05	0.05	0.05	0.00	0.00	0.00	0.40	0.50	0.50	0.40
Campbell black-browed albatross	0.05	0.05	0.05	0.05	0.00	0.00	0.00	0.20	0.50	0.50	0.40	0.30
Shy albatross	0.10	0.05	0.05	0.05	0.05	0.05	0.10	0.10	0.50	0.50	0.40	0.40
New Zealand white-capped albatross	0.40	0.10	0.05	0.05	0.05	0.05	0.05	0.00	0.00	0.25	0.50	0.50
Salvin's albatross	0.05	0.05	0.05	0.00	0.00	0.00	0.10	0.30	0.50	0.50	0.40	0.10
Chatham Island albatross	0.10	0.05	0.05	0.05	0.00	0.00	0.20	0.40	0.50	0.50	0.40	0.30
Grey-headed albatross	0.30	0.05	0.05	0.05	0.05	0.00	0.00	0.00	0.10	0.50	0.50	0.40
Southern Buller's albatross	0.20	0.50	0.45	0.30	0.05	0.05	0.05	0.00	0.00	0.00	0.00	0.00
Northern Buller's albatross	0.45	0.40	0.05	0.05	0.05	0.00	0.00	0.00	0.00	0.00	0.40	0.50
Sooty albatross	0.20	0.05	0.05	0.05	0.05	0.00	0.00	0.50	0.70	0.70	0.50	0.50
Light-mantled sooty albatross	0.40	0.10	0.05	0.05	0.05	0.05	0.00	0.00	0.10	0.50	0.50	0.40
Grey petrel	0.00	0.50	0.50	0.50	0.40	0.30	0.05	0.05	0.05	0.05	0.05	0.00
Black petrel	0.50	0.40	0.05	0.05	0.05	0.05	0.00	0.00	0.00	0.05	0.30	0.50
Westland petrel	0.00	0.15	0.30	0.40	0.50	0.50	0.45	0.40	0.05	0.05	0.05	0.00
White-chinned petrel	0.40	0.30	0.05	0.05	0.00	0.00	0.00	0.00	0.30	0.40	0.50	0.50
Spectacled petrel	0.10	0.05	0.05	0.00	0.00	0.00	0.00	0.00	0.50	0.50	0.40	0.30

Table 6: Probability of an adult being in the southern hemisphere by month $(P_{s,m}^{SH})$. Darker shaded cells indicate a higher probability.

Common name	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Gibson's albatross	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
Antipodean albatross	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
Wandering albatross	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
Tristan albatross	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
Amsterdam albatross	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
Southern royal albatross	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
Northern royal albatross	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
Atlantic yellow-nosed albatross	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
Indian yellow-nosed albatross	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
Black-browed albatross	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
Campbell black-browed albatross	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
Shy albatross	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
New Zealand white-capped albatross	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
Salvin's albatross	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
Chatham Island albatross	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
Grey-headed albatross	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
Southern Buller's albatross	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
Northern Buller's albatross	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
Sooty albatross	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
Light-mantled sooty albatross	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
Grey petrel	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
Black petrel	1.00	1.00	1.00	1.00	1.00	0.80	0.80	0.80	0.80	0.80	1.00	1.00
Westland petrel	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
White-chinned petrel	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
Spectacled petrel	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00

Table 7: Prior values for the annual number of breeding pairs ($N_s^{\rm BP}$), proportion of adults breeding ($P_s^{\rm B}$), age at first reproduction ($A_s^{\rm curr}$), and optimum survivorship ($S_s^{\rm opt}$).

			N_s^{BP}		P_s^B	1	A_s^{curr}	S_s^{opt}		
Code	Common name	Mean	95% CI	Mean	95% CI	Mean	95% CI	Mean	95% CI	
DIW	Gibson's albatross	4 421	4 000-4 864	0.60	0.52-0.67	11.9	8.5-16.1	0.96	0.95-0.98	
DQS	Antipodean albatross	3 381	3 065-3 725	0.45	0.35-0.55	13.9	10.5-18.2	0.97	0.95-0.98	
DIX	Wandering albatross	10 131	9 175-11 134	0.49	0.40-0.59	9.9	7.3-13.3	0.97	0.95-0.98	
DBN	Tristan albatross	1 623	1 146-1 973	0.35	0.23-0.48	9.2	6.5-12.7	0.96	0.95-0.98	
DAM	Amsterdam albatross	60	49-73	0.60	0.50-0.69	9.9	7.3-13.2	0.96	0.95-0.98	
DIP	Southern royal albatross	5 818	5 043-6 653	0.53	0.33-0.72	9.2	6.2-13.0	0.96	0.95-0.98	
DIQ	Northern royal albatross	4 257	3 413-5 239	0.53	0.33-0.73	8.9	8.5-9.3	0.97	0.95-0.98	
DCR	Atlantic yellow-nosed albatross	26 808	22 001-32 403	0.60	0.58-0.61	8.9	6.4-12.2	0.95	0.93-0.97	
TQH	Indian yellow-nosed albatross	34 002	27 855-41 039	0.60	0.49-0.69	8.9	6.3-12.1	0.95	0.93-0.97	
DIM	Black-browed albatross	671 369	607 619-738 568	0.84	0.79-0.89	9.9	7.3-13.1	0.95	0.93-0.97	
TQW	Campbell black-browed albatross	14 119	12 768-15 549	0.89	0.75-0.96	9.2	6.2-13.1	0.95	0.93-0.97	
DCU	Shy albatross	15 339	12 529-18 518	0.74	0.64-0.83	8.8	5.8-13.0	0.95	0.94-0.97	
TWD	New Zealand white-capped albatross	85 808	67 480-107 569	0.68	0.56-0.79	8.8	5.8-13.0	0.95	0.94-0.97	
DKS	Salvin's albatross	35 238	31 960-38 794	0.82	0.67-0.94	11.2	8.4-14.7	0.95	0.94-0.97	
DER	Chatham Island albatross	5 294	5 188-5 400	0.77	0.66-0.86	9.9	7.8-12.3	0.96	0.94-0.97	
DIC	Grey-headed albatross	63 034	57 057-69 504	0.41	0.19-0.63	12.9	10.2-16.1	0.96	0.95-0.98	
DSB	Southern Buller's albatross	13 499	12 211-14 878	0.80	0.66-0.92	11.9	9.2-15.1	0.95	0.93-0.97	
DNB	Northern Buller's albatross	19 362	17 529-21 341	0.80	0.69-0.88	11.9	9.3-15.1	0.95	0.93-0.97	
PHU	Sooty albatross	13 359	11 705-14 451	0.73	0.62-0.82	9.2	6.3-13.1	0.97	0.95-0.98	
PHE	Light-mantled sooty albatross	20 905	17 136-25 231	0.73	0.49-0.91	9.2	6.3-13.1	0.97	0.95-0.98	
PCI	Grey petrel	105 660	77 870-140 105	0.89	0.75-0.96	6.9	5.2-9.0	0.94	0.92-0.95	
PRK	Black petrel	5 458	4 873-6 083	0.61	0.53-0.69	7.4	7.0-7.9	0.93	0.92-0.95	
PCW	Westland petrel	6 225	5 514-6 987	0.48	0.34-0.63	7.0	5.0-9.4	0.95	0.93-0.96	
PRO	White-chinned petrel	1 316 786	1 074 335-1 593 474	0.75	0.64-0.83	6.6	4.6-9.2	0.93	0.92-0.95	
PCN	Spectacled petrel	41 988	34 447-50 333	0.79	0.68-0.88	6.6	4.6-9.1	0.94	0.92-0.95	

Table 8: Prior values for the total number of adults (N_s) and the theoretical unconstrained maximum population growth rate (r_s) .

		Λ	J_s (thousand)		r_s
Code	Common name	Mean	95% CI	Mean	95% CI
DIW	Gibson's albatross	14 909	12 750-17 458	0.04	0.03-0.05
DQS	Antipodean albatross	15 263	11 956-19 727	0.04	0.03-0.05
DIX	Wandering albatross	41 429	33 352-51 892	0.05	0.03-0.06
DBN	Tristan albatross	9 690	5 900-15 107	0.05	0.04-0.06
DAM	Amsterdam albatross	202	156-260	0.05	0.03-0.06
DIP	Southern royal albatross	22 877	15 534-36 179	0.05	0.04-0.07
DIQ	Northern royal albatross	16 704	10 850-27 135	0.05	0.04-0.06
DCR	Atlantic yellow-nosed albatross	89 992	73 818-108 954	0.06	0.04-0.07
TQH	Indian yellow-nosed albatross	115 030	88 811-147 884	0.06	0.04-0.07
DIM	Black-browed albatross	1 593 207	1 422 033-1 791 582	0.05	0.04-0.07
TQW	Campbell black-browed albatross	31 907	27 687-38 369	0.06	0.04-0.07
DCU	Shy albatross	41 464	32 765-52 255	0.06	0.04-0.08
TWD	New Zealand white-capped albatross	254 551	189 506-338 493	0.06	0.04-0.08
DKS	Salvin's albatross	86 384	72 536-107 411	0.05	0.04-0.06
DER	Chatham Island albatross	13 835	12 342-16 052	0.05	0.04-0.06
DIC	Grey-headed albatross	340 458	195 740-648 759	0.04	0.03-0.05
DSB	Southern Buller's albatross	33 852	28 455-41 829	0.05	0.04-0.06
DNB	Northern Buller's albatross	48 877	41 987-58 026	0.05	0.04-0.06
PHU	Sooty albatross	36 871	30 880-44 041	0.05	0.04-0.07
PHE	Light-mantled sooty albatross	58 790	42 233-88 017	0.05	0.04-0.07
PCI	Grey petrel	238 644	172 197-326 322	0.07	0.06-0.09
PRK	Black petrel	17 981	15 118-21 433	0.07	0.06-0.08
PCW	Westland petrel	26 630	19 309-37 730	0.07	0.05-0.09
PRO	White-chinned petrel	3 543 560	2 799 132-4 491 550	0.08	0.06-0.10
PCN	Spectacled petrel	106 495	84 283-133 438	0.08	0.06-0.10

Table 9: Total observed captures (Obs n; individuals), observed effort (Obs eff; 1000 hooks) and total effort (Tot eff; 1000 hooks) by fishery group and year, for a) NZL (DOM), NZL (JV), ZAF (DOM), ZAF (JV) and AUS), and b) (continued on next page) JPN, TWN and KOR. DOM denotes domestic fleet, and JV Joint Venture operations.

a) NZL (DOM), NZL (JV), ZAF (DOM), ZAF (JV) and AUS

		NZL (DOM)			NZL (JV)			ZAF (DOM)			ZAF (JV)			AUS	
Year	Obs n	Obs eff	Tot eff	Obs n	Obs eff	Tot eff	Obs n	Obs eff	Tot eff	Obs n	Obs eff	Tot eff	Obs n	Obs eff	Tot eff
2012	24	148	2 5 1 0	33	555	551	0	0	1 572	126	337	2 742	3	487	7 369
2013	24	88	2 287	5	488	488	0	0	1 745	267	719	3 094	0	401	7 312
2014	18	126	1 868	16	653	653	20	23	1 767	170	475	1 265	1	222	7 341
2015	23	122	1808	22	619	622	18	23	1878	120	309	978	2	172	8 560
2016	128	332	2 358	0	0	0	0	0	1 573	37	101	668	3	771	8 094
2017	55	333	2 119	0	0	0	6	7	1 783	77	206	890	6	949	9 098
2018	95	301	2 317	0	0	0	6	7	2 230	15	38	651	14	907	8 249
2019	54	165	2 042	0	0	0	14	18	2 176	24	64	724	10	1 048	8 905
2020	18	197	1 974	0	0	0	77	95	1 661	0	0	0	6	862	8 392
2021	48	184	1 546	0	0	0	9	12	2 116	11	31	197	10	777	8 009
2022	56	68	1 280	0	0	0	10	15	2 356	4	12	163	3	693	7 124
2023	18	50	1 497	0	0	0	38	59	2 932	0	0	0	8	711	7 463
Total	561	2 114	23 604	76	2 314	2 315	198	260	23 789	851	2 292	11 370	66	8 000	95 914

Table 9 continued.b) JPN, TWN, KOR, and total across fishery groups

		JPN			TWN			KOR			Total	
Year	Obs n	Obs eff	Tot eff	Obs n	Obs eff	Tot eff	Obs n	Obs eff	Tot eff	Obs n	Obs eff	Tot eff
2012	120	2 921	139 354	162	11 542	195 190	0	0	52 674	468	15 990	401 962
2013	423	4 745	121 815	355	11 424	232 556	0	0	61 178	1 074	17 864	430 473
2014	746	6 540	105 885	123	9 954	229 415	0	0	54 717	1 094	17 992	402 912
2015	946	5 175	94 939	26	8 554	201 169	0	0	53 628	1 157	14 974	363 581
2016	1 559	6 344	93 383	59	9 229	225 181	0	0	59 769	1 786	16 777	391 026
2017	121	5 164	91 530	42	13 316	281 430	0	0	43 958	307	19 976	430 807
2018	355	5 304	88 059	76	15 005	266 056	0	0	43 974	561	21 563	411 535
2019	1 857	5 265	70 012	71	15 340	301 488	26	530	2 427	2 056	22 431	387 773
2020	136	2 302	65 604	48	12 929	316 198	0	0	0	285	16 385	393 830
2021	0	0	59 565	80	11 581	192 956	0	0	0	158	12 586	264 388
2022	0	0	53 050	266	14 215	249 051	28	386	2 413	367	15 389	315 437
2023	151	3 042	41 394	269	14 246	265 050	18	501	2 478	502	18 609	320 814
Total	6 414	46 804	1 024 590	1 577	147 334	2 955 740	72	1 417	377 216	9 815	210 534	4 514 537

Table 10: Observed captures per capture code and fishery group. DOM denotes domestic fleet, and JV Joint Venture operations.

		NZL	NZL			ZAF	ZAF			
Code	Common name	(DOM)	(JV)	JPN	TWN	(DOM)	(JV)	AUS	KOR	Total
DIZ	<i>Diomedea</i> spp	51	0	430	106	2	0	1	3	593
THZ	Thalassarche spp	358	74	3 853	734	148	316	7	59	5 549
PHZ	<i>Phoebetria</i> spp	0	0	267	115	0	0	0	10	392
PTZ	Procellaria spp	152	2	650	435	43	520	0	0	1 802
ALZ	Diomedeidae	0	0	824	172	5	15	33	0	1 049
PRX	Procellariidae	0	0	167	7	0	0	16	0	190
BLZ	Bird	0	0	223	8	0	0	9	0	240
	Total	561	76	6 414	1 577	198	851	66	72	9 8 1 5

Table 11: Observed captures by at-vessel status, per capture code and fishery group. DOM denotes domestic fleet, and JV Joint Venture operations.

		NZL (DOM)		NZL (JV)		JPN		TWN		ZAF (DOM)		ZAF (JV)		AUS		KOR	
Code	Common name	Alive	Dead	Alive	Dead	Alive	Dead	Alive	Dead	Alive	Dead	Alive	Dead	Alive	Dead	Alive	Dead
DIZ	Diomedea spp	13	38	0	0	58	369	2	100	0	2	0	0	1	0	0	3
THZ	Thalassarche spp	42	316	33	41	60	3785	28	689	24	121	213	68	3	4	0	59
PHZ	<i>Phoebetria</i> spp	0	0	0	0	1	266	0	115	0	0	0	0	0	0	0	10
PTZ	Procellaria spp	19	133	0	2	4	646	20	404	0	41	53	458	0	0	0	0
ALZ	Diomedeidae	0	0	0	0	17	461	10	159	3	2	8	6	11	21	0	0
PRX	Procellariidae	0	0	0	0	4	117	2	2	0	0	0	0	7	8	0	0
BLZ	Bird	0	0	0	0	6	204	1	7	0	0	0	0	3	6	0	0
Total		74	487	33	43	150	5848	63	1476	27	166	274	532	25	39	0	72

Table 12: Conversion matrix for calculation of cumulative captures per capture code.

Code	DIZ	THZ	PHZ	PTZ	ALZ	PRX	BLZ
DIZ	1	0	0	0	0	0	0
THZ	0	1	0	0	0	0	0
PHZ	0	0	1	0	0	0	0
PTZ	0	0	0	1	0	0	0
ALZ	1	1	1	0	1	0	0
PRX	0	0	0	1	0	1	0
BLZ	1	1	1	1	1	1	1

Table~13:~Observed~captures~per~capture~code~from~cells~with~zero~densities~in~all~months.~DOM~denotes~domestic~fleet,~and~JV~Joint~Venture~operations.

	NZL	NZL			ZAF	ZAF			
Code	(DOM)	(JV)	JPN	TWN	(DOM)	(JV)	AUS	KOR	Total
DIZ	0	0	0	0	0	0	0	0	0
THZ	0	0	0	0	0	0	0	0	0
PHZ	0	0	0	1	0	0	0	0	1
PTZ	0	0	0	0	0	2	0	0	2
ALZ	0	0	0	0	0	0	0	0	0
PRX	0	0	0	0	0	0	0	0	0
BLZ	0	0	0	3	0	0	0	0	3
Total	0	0	0	4	0	2	0	0	6

Table 14: Estimated observed overlap by species and fishery group, with units hooks km⁻². DOM denotes domestic fleet, and JV Joint Venture operations.

		NZL	NZL			ZAF	ZAF			
Code	Common name	(DOM)	(JV)	JPN	TWN	(DOM)	(JV)	AUS	KOR	Total
DIW	Gibson's albatross	0.19	0.59	1.20	0.44	0.00	0.00	0.30	0.00	2.72
DQS	Antipodean albatross	0.13	0.11	0.09	0.18	0.00	0.00	0.02	0.00	0.54
DIX	Wandering albatross	0.01	0.01	0.09	0.36	<0.01	0.01	0.01	< 0.01	0.50
DBN	Tristan albatross	0.00	0.00	0.71	0.61	0.01	0.03	< 0.01	0.18	1.54
DAM	Amsterdam albatross	0.00	0.00	0.13	2.84	<0.01	0.02	<0.01	<0.01	2.99
DIP	Southern royal albatross	0.14	0.28	0.20	0.08	<0.01	<0.01	0.02	<0.01	0.72
DIQ	Northern royal albatross	0.12	0.03	0.04	0.08	<0.01	<0.01	<0.01	<0.01	0.28
DCR	Atlantic yellow-nosed albatross	0.00	0.00	0.37	0.62	0.01	0.03	0.00	0.06	1.09
TQH	Indian yellow-nosed albatross	<0.01	< 0.01	0.40	0.96	<0.01	0.01	0.05	<0.01	1.43
DIM	Black-browed albatross	0.01	< 0.01	0.04	0.13	<0.01	0.01	0.01	<0.01	0.19
TQW	Campbell black-browed albatross	0.07	0.09	0.35	0.30	0.00	0.00	0.10	0.00	0.90
DCU	Shy albatross	0.00	0.00	0.29	0.19	<0.01	0.01	0.14	<0.01	0.63
TWD	New Zealand white-capped albatross	0.18	0.41	0.59	0.46	<0.01	0.03	0.12	<0.01	1.79
DKS	Salvin's albatross	0.10	0.07	0.06	0.11	<0.01	<0.01	0.02	< 0.01	0.36
DER	Chatham Island albatross	0.04	< 0.01	0.08	0.08	0.00	0.00	<0.01	0.00	0.20
DIC	Grey-headed albatross	0.01	0.01	0.08	0.10	<0.01	<0.01	<0.01	0.01	0.21
DSB	Southern Buller's albatross	0.14	0.26	0.60	0.21	0.00	0.00	0.14	0.00	1.36
DNB	Northern Buller's albatross	0.08	0.05	0.10	0.07	0.00	0.00	0.01	0.00	0.31
PHU	Sooty albatross	0.00	0.00	0.40	0.71	<0.01	0.01	<0.01	0.07	1.19
PHE	Light-mantled sooty albatross	0.01	0.04	0.09	0.05	<0.01	<0.01	<0.01	< 0.01	0.20
PCI	Grey petrel	0.05	0.06	0.10	0.11	<0.01	<0.01	<0.01	0.02	0.34
PRK	Black petrel	0.07	0.03	0.16	0.19	0.00	0.00	0.06	0.00	0.51
PCW	Westland petrel	0.29	0.63	0.39	0.21	0.00	0.00	0.04	0.00	1.57
PRO	White-chinned petrel	<0.01	<0.01	0.07	0.20	<0.01	0.02	0.01	0.01	0.30
PCN	Spectacled petrel	0.00	0.00	0.05	0.60	<0.01	<0.01	0.00	0.01	0.65
Total		1.64	2.67	6.68	9.90	0.04	0.18	1.05	0.37	22.52

Table 15: Estimated total overlap by species and fishery group, with units hooks km⁻². DOM denotes domestic fleet, and JV Joint Venture operations.

Code	Common name	NZL (DOM)	NZL (JV)	JPN	TWN	ZAF (DOM)	ZAF (JV)	AUS	KOR	Total
DIW	Gibson's albatross	1.7	0.6	13.4	3.5	0.0	0.0	3.5	<0.1	22.6
DQS	Antipodean albatross	1.4	0.1	1.6	2.0	0.0	0.0	0.3	<0.1	5.4
DIX	Wandering albatross	0.1	<0.1	1.4	6.6	0.1	0.1	0.1	0.2	8.5
DBN	Tristan albatross	0.0	0.0	6.3	6.0	0.5	0.2	<0.1	2.8	15.7
DAM	Amsterdam albatross	0.0	0.0	2.4	28.4	0.1	0.1	<0.1	0.2	31.1
DIP	Southern royal albatross	1.5	0.3	2.6	0.6	<0.1	<0.1	0.2	<0.1	5.3
DIQ	Northern royal albatross	1.7	<0.1	0.6	0.6	<0.1	<0.1	<0.1	0.1	3.0
DCR	Atlantic yellow-nosed albatross	0.0	0.0	6.8	8.6	0.8	0.1	0.0	1.1	17.4
TQH	Indian yellow-nosed albatross	<0.1	<0.1	6.5	9.0	0.2	0.1	0.6	0.4	16.8
DIM	Black-browed albatross	0.1	<0.1	8.0	1.3	0.2	<0.1	0.1	0.1	2.6
TQW	Campbell black-browed albatross	0.7	0.1	4.5	2.2	0.0	0.0	1.1	0.1	8.7
DCU	Shy albatross	0.0	0.0	2.1	2.8	0.1	<0.1	1.1	0.1	6.2
TWD	New Zealand white-capped albatross	1.8	0.4	7.6	5.5	0.3	0.1	1.4	0.1	17.2
DKS	Salvin's albatross	1.1	0.1	1.7	1.2	<0.1	<0.1	0.2	<0.1	4.3
DER	Chatham Island albatross	0.4	<0.1	2.0	1.0	0.0	0.0	<0.1	0.1	3.6
DIC	Grey-headed albatross	0.1	<0.1	0.7	1.1	<0.1	<0.1	<0.1	0.2	2.1
DSB	Southern Buller's albatross	1.5	0.3	7.3	2.3	0.0	0.0	1.6	<0.1	13.0
DNB	Northern Buller's albatross	1.1	<0.1	2.8	1.0	0.0	0.0	0.1	0.0	5.1
PHU	Sooty albatross	0.0	0.0	3.5	8.6	0.3	0.1	<0.1	1.3	13.9
PHE	Light-mantled sooty albatross	0.1	<0.1	1.0	0.7	<0.1	<0.1	<0.1	0.1	2.0
PCI	Grey petrel	0.5	0.1	1.0	1.1	<0.1	<0.1	<0.1	0.3	3.0
PRK	Black petrel	1.0	<0.1	3.6	4.3	0.0	0.0	0.7	0.6	10.1
PCW	Westland petrel	2.6	0.6	5.6	1.9	0.0	0.0	0.5	0.0	11.2
PRO	White-chinned petrel	0.1	<0.1	1.6	2.8	0.2	0.1	0.1	0.2	5.0
PCN	Spectacled petrel	0.0	0.0	1.0	11.9	0.2	<0.1	0.0	0.1	13.2
Total		17.4	2.7	88.3	114.9	2.9	1.0	11.6	8.2	247.0

Table 16: The mean of the year and month specific proportions of each population from 5° cells that overlapped with fishing effort.

		NZL	NZL			ZAF	ZAF			
Code	Common name	(DOM)	(JV)	JPN	TWN	(DOM)	(JV)	AUS	KOR	Total
DIW	Gibson's albatross	0.076	0.109	0.156	0.051	0.000	0.000	0.099	0.002	0.276
DQS	Antipodean albatross	0.059	0.018	0.037	0.033	0.000	0.000	0.008	0.000	0.111
DIX	Wandering albatross	0.003	0.002	0.033	0.043	0.006	0.004	0.003	0.007	0.079
DBN	Tristan albatross	0.000	0.000	0.099	0.080	0.028	0.011	0.001	0.040	0.188
DAM	Amsterdam albatross	0.000	0.000	0.044	0.191	0.005	0.004	0.001	0.015	0.224
DIP	Southern royal albatross	0.061	0.053	0.029	0.009	0.000	0.000	0.008	0.001	0.092
DIQ	Northern royal albatross	0.089	0.004	0.010	0.008	0.001	0.001	0.001	0.002	0.102
DCR	Atlantic yellow-nosed albatross	0.000	0.000	0.107	0.145	0.046	0.011	0.000	0.034	0.256
TQH	Indian yellow-nosed albatross	0.002	0.001	0.070	0.086	0.013	0.006	0.023	0.022	0.168
DIM	Black-browed albatross	0.007	0.001	0.020	0.013	0.008	0.002	0.002	0.002	0.039
TQW	Campbell black-browed albatross	0.031	0.018	0.051	0.029	0.000	0.000	0.030	0.010	0.108
DCU	Shy albatross	0.000	0.000	0.017	0.017	0.005	0.003	0.143	0.003	0.175
TWD	New Zealand white-capped albatross	0.076	0.084	0.079	0.039	0.015	0.009	0.042	0.007	0.214
DKS	Salvin's albatross	0.049	0.010	0.057	0.011	0.002	0.001	0.008	0.001	0.117
DER	Chatham Island albatross	0.026	0.001	0.075	0.020	0.000	0.000	0.001	0.002	0.117
DIC	Grey-headed albatross	0.002	0.001	0.013	0.010	0.001	0.001	0.001	0.004	0.026
DSB	Southern Buller's albatross	0.064	0.052	0.100	0.032	0.000	0.000	0.043	0.000	0.195
DNB	Northern Buller's albatross	0.050	0.008	0.111	0.020	0.000	0.000	0.005	0.000	0.155
PHU	Sooty albatross	0.000	0.000	0.062	0.061	0.020	0.007	0.001	0.021	0.132
PHE	Light-mantled sooty albatross	0.004	0.009	0.011	0.007	0.002	0.001	0.001	0.004	0.022
PCI	Grey petrel	0.018	0.012	0.012	0.011	0.001	0.001	0.000	0.005	0.041
PRK	Black petrel	0.054	0.006	0.095	0.092	0.000	0.000	0.022	0.014	0.246
PCW	Westland petrel	0.107	0.112	0.081	0.029	0.000	0.000	0.017	0.000	0.177
PRO	White-chinned petrel	0.003	0.001	0.041	0.024	0.010	0.005	0.002	0.005	0.068
PCN	Spectacled petrel	0.000	0.000	0.057	0.209	0.013	0.001	0.000	0.006	0.236

Table 17 Comparison of predicted vs observed captures per capture code from the selected 2025 risk assessment model, for a) NZL (DOM), NZL (JV), ZAF (DOM), ZAF (JV) and AUS), and b) (continued on next page) JPN, TWN and KOR. DOM denotes domestic fleet, and JV Joint Venture operations. 95% CIs are provided in parentheses.

a) NZL (DOM), NZL (JV), ZAF (DOM), ZAF (JV) and AUS

			NZL (DOM)		NZL (JV)		ZAF (DOM)		ZAF (JV)		AUS
Code	Common name	Obs	Est	Obs	Est	Obs	Est	Obs	Est	Obs	Est
DIZ	Diomedea spp	4.2	3.6 (2.2-5.3)	0	0 (0-0.2)	0.2	0.1 (0-0.2)	0	0 (0-0.2)	0.1	0.1 (0-0.4)
THZ	Thalassarche spp	29.8	29 (24.7-33.7)	6.2	5.5 (3.8-7.6)	12.3	11.6 (8.9-14.7)	26.3	25.6 (21.5-29.8)	0.6	0.7 (0.2-1.5)
PHZ	<i>Phoebetria</i> spp	0	0 (0-0.2)	0	0 (0-0.1)	0	0 (0-0.1)	0	0 (0-0.2)	0	0 (0-0.1)
PTZ	<i>Procellaria</i> spp	12.7	12 (9.5-14.9)	0.2	0.1 (0-0.4)	3.6	3.1 (1.8-4.6)	43.3	42.3 (37.2-47.7)	0	0.2 (0-0.6)
ALZ	Diomedeidae	0	0.7 (0.2-1.4)	0	0.3 (0-0.8)	0.4	0.7 (0.2-1.5)	1.2	1.5 (0.7-2.5)	2.8	2.4 (1.2-3.7)
PRX	Procellariidae	0	0.3 (0-0.8)	0	0.1 (0-0.3)	0	0.3 (0-0.7)	0	0.4 (0.1-1)	1.3	1 (0.3-1.9)
BLZ	Bird	0	1 (0.3-1.8)	0	0.4 (0.1-1)	0	0.6 (0.2-1.3)	0	0.9 (0.2-1.7)	0.8	1.1 (0.4-2.1)

b) JPN, TWN and KOR

			JPN		TWN		KOR
Code	Common name	Obs	Est	Obs	Est	Obs	Est
DIZ	Diomedea spp	35.8	35.4 (30.7-40)	8.8	8.4 (6.2-10.8)	0.2	0.1 (0-0.4)
THZ	Thalassarche spp	321.1	320.8 (307.2-334.8)	61.2	60.5 (54.3-66.8)	4.9	4.3 (2.8-5.9)
PHZ	<i>Phoebetria</i> spp	22.2	21.9 (18.4-25.5)	9.6	9.1 (6.8-11.5)	8.0	0.5 (0.1-1.1)
PTZ	<i>Procellaria</i> spp	54.2	53.8 (48.1-59.6)	36.2	35.7 (31.1-40.7)	0	0.1 (0-0.2)
ALZ	Diomedeidae	68.7	69.1 (62.6-76.2)	14.3	14.8 (11.8-17.8)	0	0.5 (0.1-1.2)
PRX	Procellariidae	13.9	14.1 (11.4-17.2)	0.6	0.9 (0.2-1.8)	0	0.1 (0-0.2)
BLZ	Bird	18.6	19.5 (16.3-23)	0.7	2.1 (1.1-3.2)	0	0.5 (0.1-1.2)

Table 18: Catchability coefficients estimated from the selected 2025 risk assessment model. DOM denotes domestic fleet, and JV Joint Venture operations. 95% CIs are provided in parentheses.

Species group	NZL (DOM)	NZL (JV)	JPN	TWN	ZAF (DOM)	ZAF (JV)	AUS	KOR
Great albatross	5.23 (3.71-7.2)	0.07 (0.01-0.23)	20.15 (15.82-25.9)	4.63 (3.22-6.35)	15.39 (2.2-44.03)	1.88 (0.18-6.48)	0.94 (0.13-3.16)	1.74 (0.4-4.41)
Mollymawk	4.4 (3.53-5.33)	0.57 (0.4-0.79)	12.1 (10.23-13.87)	1.53 (1.33-1.76)	20.07 (16.54-23.9)	16.83 (14.07-19.78)	0.77 (0.47-1.07)	4.12 (2.91-5.41)
Sooty albatross	1.29 (0.16-4.2)	0.23 (0.02-0.86)	23.99 (16.3-34.06)	5.81 (4.18-8.13)	4.93 (0.35-19.64)	2.22 (0.21-8.05)	2.36 (0.12-11.62)	3.6 (1.62-6.17)
Medium petrel	5.38 (4.21-6.77)	0.1 (0.03-0.24)	3.4 (2.72-4.2)	0.59 (0.46-0.75)	4.68 (3-6.76)	9.58 (7.36-12.15)	1.3 (0.74-2.09)	0.09 (0.02-0.24)

Table 19: Estimated total deaths (D_s), cryptic deaths, maximum theoretical growth rate ($r_s \cdot N_s$) and relative mortalities (deaths relative to $r_s \cdot N_s$) from the selected 2025 risk assessment model. 95% CIs are provided in parentheses.

Common name	Total deaths	Cryptic deaths	$r_{\scriptscriptstyle S}\cdot N_{\scriptscriptstyle S}$	Relative mortalities
Gibson's albatross	438 (305-646)	110 (6-280)	598 (450-800)	0.72 (0.48-1.14)
Antipodean albatross	78 (53-115)	20 (5-45)	551 (407-782)	0.14 (0.10-0.21)
Wandering albatross	265 (187-393)	72 (14-158)	1,879 (1,418-2,513)	0.14 (0.10-0.22)
Tristan albatross	166 (102-270)	43 (8-104)	459 (269-770)	0.36 (0.24-0.55)
Amsterdam albatross	3 (2-6)	1 (0-2)	9 (7-13)	0.38 (0.25-0.60)
Southern royal albatross	149 (93-251)	37 (5-101)	1,049 (685-1,810)	0.14 (0.09-0.21)
Northern royal albatross	41 (24-70)	10 (3-24)	804 (524-1,383)	0.05 (0.04-0.07)
Atlantic yellow-nosed albatross	1,071 (771-1,532)	297 (69-657)	5,130 (3,741-7,061)	0.21 (0.15-0.30)
Indian yellow-nosed albatross	1,299 (868-1,897)	361 (56-836)	6,476 (4,542-9,190)	0.20 (0.14-0.30)
Black-browed albatross	2,936 (2,275-3,861)	802 (241-1,634)	83,375 (65,879-106,184)	0.04 (0.03-0.05)
Campbell black-browed albatross	226 (158-338)	65 (6-154)	1,751 (1,309-2,439)	0.13 (0.09-0.20)
Shy albatross	149 (103-226)	42 (7-98)	2,257 (1,547-3,327)	0.07 (0.04-0.10)
New Zealand white-capped albatross	3,445 (2,457-4,820)	965 (158-2,178)	14,038 (9,405-20,602)	0.24 (0.16-0.38)
Salvin's albatross	260 (186-379)	73 (13-163)	3,929 (2,963-5,390)	0.07 (0.05-0.10)
Chatham Island albatross	41 (29-62)	12 (1-29)	698 (572-868)	0.06 (0.04-0.09)
Grey-headed albatross	452 (263-949)	125 (25-347)	12,616 (7,323-27,748)	0.04 (0.03-0.05)
Southern Buller's albatross	344 (236-526)	97 (6-232)	1,541 (1,220-2,001)	0.23 (0.15-0.33)
Northern Buller's albatross	227 (155-330)	64 (6-154)	2,273 (1,781-2,920)	0.10 (0.07-0.15)
Sooty albatross	567 (402-825)	166 (37-340)	1,780 (1,326-2,453)	0.32 (0.20-0.49)
Light-mantled sooty albatross	169 (99-287)	48 (5-127)	2,687 (1,782-4,198)	0.06 (0.04-0.11)
Grey petrel	141 (94-213)	39 (11-83)	17,819 (12,545-26,541)	0.01 (0.01-0.01)
Black petrel	36 (25-52)	10 (3-21)	1,264 (1,073-1,538)	0.03 (0.02-0.04)
Westland petrel	82 (52-131)	23 (5-53)	1,782 (1,214-2,635)	0.05 (0.03-0.07)
White-chinned petrel	3,562 (2,829-4,610)	1,011 (292-2,036)	263,270 (186,430-382,655)	0.01 (0.01-0.02)
Spectacled petrel	132 (88-197)	36 (8-80)	8,140 (5,857-11,532)	0.02 (0.01-0.02)

Table 20: Estimated total deaths (D_s), maximum theoretical growth rate ($r_s \cdot N_s$) and relative mortalities (deaths relative to $r_s \cdot N_s$) for: the selected 2025 risk assessment model; and, the sensitivity run with family-specific π . 95% CIs are provided in parentheses.

	:	2025 risk assessment model Relative			Sensitivity – family-specific $oldsymbol{\pi}$			
Common name	Total deaths	$r_{_{\!S}}\cdot N_{_{\!S}}$	mortalities	Total deaths	$r_{\!\scriptscriptstyle S}\cdot N_{\!\scriptscriptstyle S}$	Relative mortalities		
Gibson's albatross	438 (305-646)	598 (450-800)	0.72 (0.48-1.14)	361 (263-514)	604 (450-817)	0.60 (0.39-0.89)		
Antipodean albatross	78 (53-115)	551 (407-782)	0.14 (0.10-0.21)	66 (46-96)	550 (391-764)	0.12 (0.08-0.17)		
Wandering albatross	265 (187-393)	1,879 (1,418-2,513)	0.14 (0.10-0.22)	223 (159-310)	1,896 (1,427-2,550)	0.12 (0.08-0.17)		
Tristan albatross	166 (102-270)	459 (269-770)	0.36 (0.24-0.55)	136 (87-222)	464 (282-766)	0.30 (0.21-0.43)		
Amsterdam albatross	3 (2-6)	9 (7-13)	0.38 (0.25-0.60)	3 (2-4)	9 (7-13)	0.32 (0.21-0.47)		
Southern royal albatross	149 (93-251)	1,049 (685-1,810)	0.14 (0.09-0.21)	126 (80-203)	1,049 (695-1,754)	0.12 (0.08-0.18)		
Northern royal albatross	41 (24-70)	804 (524-1,383)	0.05 (0.04-0.07)	37 (23-62)	807 (551-1,401)	0.04 (0.03-0.06)		
Atlantic yellow-nosed albatross	1,071 (771-1,532)	5,130 (3,741-7,061)	0.21 (0.15-0.30)	1,110 (799-1,546)	5,132 (3,739-6,851)	0.22 (0.15-0.32)		
Indian yellow-nosed albatross	1,299 (868-1,897)	6,476 (4,542-9,190)	0.20 (0.14-0.30)	1,338 (926-1,951)	6,466 (4,559-9,223)	0.21 (0.14-0.31)		
Black-browed albatross	2,936 (2,275-3,861)	83,375 (65,879-106,184)	0.04 (0.03-0.05)	3,019 (2,339-3,938)	82,522 (65,404-105,457)	0.04 (0.03-0.05)		
Campbell black-browed albatross	226 (158-338)	1,751 (1,309-2,439)	0.13 (0.09-0.20)	238 (165-336)	1,799 (1,354-2,432)	0.13 (0.09-0.20)		
Shy albatross	149 (103-226)	2,257 (1,547-3,327)	0.07 (0.04-0.10)	155 (109-232)	2,235 (1,580-3,387)	0.07 (0.04-0.10)		
New Zealand white-capped albatross	3,445 (2,457-4,820)	14,038 (9,405-20,602)	0.24 (0.16-0.38)	3,576 (2,556-5,067)	14,097 (9,617-20,965)	0.25 (0.17-0.39)		
Salvin's albatross	260 (186-379)	3,929 (2,963-5,390)	0.07 (0.05-0.10)	271 (189-395)	3,981 (3,052-5,209)	0.07 (0.05-0.10)		
Chatham Island albatross	41 (29-62)	698 (572-868)	0.06 (0.04-0.09)	43 (29-62)	699 (568-869)	0.06 (0.04-0.09)		
Grey-headed albatross	452 (263-949)	12,616 (7,323-27,748)	0.04 (0.03-0.05)	466 (276-932)	12,291 (7,274-25,968)	0.04 (0.03-0.05)		
Southern Buller's albatross	344 (236-526)	1,541 (1,220-2,001)	0.23 (0.15-0.33)	362 (250-539)	1,529 (1,219-2,020)	0.23 (0.16-0.35)		
Northern Buller's albatross	227 (155-330)	2,273 (1,781-2,920)	0.10 (0.07-0.15)	234 (164-340)	2,250 (1,765-2,916)	0.10 (0.07-0.15)		
Sooty albatross	567 (402-825)	1,780 (1,326-2,453)	0.32 (0.20-0.49)	441 (334-578)	1,782 (1,335-2,450)	0.25 (0.16-0.37)		
Light-mantled sooty albatross	169 (99-287)	2,687 (1,782-4,198)	0.06 (0.04-0.11)	123 (79-194)	2,658 (1,779-4,204)	0.05 (0.03-0.07)		
Grey petrel	141 (94-213)	17,819 (12,545-26,541)	0.01 (0.01-0.01)	143 (97-213)	17,949 (12,763-26,072)	0.01 (0.01-0.01)		
Black petrel	36 (25-52)	1,264 (1,073-1,538)	0.03 (0.02-0.04)	37 (25-53)	1,269 (1,063-1,515)	0.03 (0.02-0.04)		
Westland petrel	82 (52-131)	1,782 (1,214-2,635)	0.05 (0.03-0.07)	81 (53-132)	1,778 (1,204-2,712)	0.05 (0.03-0.07)		
White-chinned petrel	3,562 (2,829-4,610)	263,270 (186,430-382,655)	0.01 (0.01-0.02)	3,599 (2,796-4,634)	259,021 (186,993-361,680)	0.01 (0.01-0.02)		
Spectacled petrel	132 (88-197)	8,140 (5,857-11,532)	0.02 (0.01-0.02)	133 (92-196)	8,081 (5,787-11,187)	0.02 (0.01-0.02)		

Table 21: Estimated total deaths (D_s), maximum theoretical growth rate, ($r_s \cdot N_s$) and relative mortalities (deaths relative to $r_s \cdot N_s$) for: the selected 2025 risk assessment model; and, the 2025 model fitted to data from 2012 to 2019. 95% CIs are provided in parentheses.

2025 risk assessment model 2025 risk assessment model fitted to data from 2012-19 Relative Relative Total deaths mortalities Common name $r_c \cdot N_c$ Total deaths $r_c \cdot N_c$ mortalities Gibson's albatross 438 (305-646) 598 (450-800) 0.72 (0.48-1.14) 587 (395-904) 603 (459-790) 0.97 (0.62-1.60) Antipodean albatross 78 (53-115) 551 (407-782) 0.14 (0.10-0.21) 93 (61-147) 554 (403-788) 0.17 (0.11-0.26) Wandering albatross 265 (187-393) 1.879 (1.418-2.513) 0.14 (0.10-0.22) 279 (187-408) 1.885 (1.409-2.571) 0.15 (0.10-0.23) Tristan albatross 166 (102-270) 459 (269-770) 0.36 (0.24-0.55) 223 (131-378) 463 (276-779) 0.47 (0.30-0.75) Amsterdam albatross 0.38 (0.25-0.60) 9 (7-13) 0.39 (0.25-0.63) 3(2-6)9 (7-13) 4(2-6)1.049 (685-1,810) Southern royal albatross 1.053 (673-1.855) 0.18 (0.12-0.30) 149 (93-251) 0.14 (0.09-0.21) 195 (116-354) Northern royal albatross 804 (524-1.383) 46 (27-84) 0.06 (0.04-0.08) 41 (24-70) 0.05 (0.04-0.07) 811 (537-1.418) Atlantic yellow-nosed albatross 1,071 (771-1,532) 5,130 (3,741-7,061) 0.21 (0.15-0.30) 1,309 (931-1,819) 5,053 (3,677-6,769) 0.26 (0.18-0.37) Indian vellow-nosed albatross 1,299 (868-1,897) 6.476 (4.542-9.190) 0.20 (0.14-0.30) 1.514 (990-2.247) 6.488 (4.588-9.103) 0.23 (0.16-0.35) Black-browed albatross 2.936 (2.275-3.861) 83.375 (65.879-106.184) 0.04 (0.03-0.05) 3,478 (2,557-4,613) 82.740 (66.062-105.017) 0.04 (0.03-0.06) Campbell black-browed albatross 226 (158-338) 1,751 (1,309-2,439) 0.13 (0.09-0.20) 283 (185-413) 1,767 (1,345-2,465) 0.16 (0.10-0.25) Shy albatross 149 (103-226) 2,257 (1,547-3,327) 0.07 (0.04-0.10) 198 (133-295) 2,276 (1,566-3,316) 0.09 (0.06-0.14) New Zealand white-capped albatross 3,445 (2,457-4,820) 14.038 (9.405-20.602) 0.24 (0.16-0.38) 4.277 (2.989-5.971) 13.935 (9.207-21.511) 0.31 (0.20-0.47) Salvin's albatross 260 (186-379) 337 (228-501) 3.929 (2.963-5.390) 0.07 (0.05-0.10) 3.949 (2.996-5.458) 0.09 (0.06-0.13) Chatham Island albatross 56 (37-83) 41 (29-62) 698 (572-868) 0.06 (0.04-0.09) 695 (552-895) 0.08 (0.05-0.12) Grey-headed albatross 452 (263-949) 12,616 (7,323-27,748) 0.04 (0.03-0.05) 524 (298-1,056) 12,231 (7,519-26,079) 0.04 (0.03-0.06) Southern Buller's albatross 344 (236-526) 1,541 (1,220-2,001) 0.23 (0.15-0.33) 444 (288-693) 1,544 (1,177-2,071) 0.29 (0.19-0.44) Northern Buller's albatross 227 (155-330) 2.273 (1.781-2.920) 0.10 (0.07-0.15) 301 (202-437) 0.13 (0.09-0.19) 2.237 (1.776-2.894) Sooty albatross 567 (402-825) 1,780 (1,326-2,453) 0.32 (0.20-0.49) 630 (430-923) 1,791 (1,307-2,444) 0.35 (0.22-0.56) Light-mantled sooty albatross 169 (99-287) 2,687 (1,782-4,198) 0.06 (0.04-0.11) 181 (108-326) 2,670 (1,770-4,326) 0.07 (0.04-0.12) Grey petrel 141 (94-213) 17,819 (12,545-26,541) 0.01 (0.01-0.01) 142 (89-219) 17,702 (11,989-26,100) 0.01 (0.01-0.01) Black petrel 36 (25-52) 1,264 (1,073-1,538) 0.03 (0.02-0.04) 42 (29-64) 1,258 (1,058-1,569) 0.03 (0.02-0.05) Westland petrel 82 (52-131) 1.782 (1.214-2.635) 0.05 (0.03-0.07) 85 (54-140) 1,780 (1,224-2,737) 0.05 (0.03-0.07) 263,270 (186,430-382,655) 0.01 (0.01-0.02) White-chinned petrel 3,562 (2,829-4,610) 4,663 (3,601-6,068) 263,870 (189,959-371,580) 0.02 (0.01-0.03) Spectacled petrel 132 (88-197) 8,140 (5,857-11,532) 157 (108-232) 8,153 (5,949-11,232) 0.04 (0.03-0.06) 0.02 (0.01-0.02)

Table 22: Estimated total deaths (D_s), maximum theoretical growth rate, ($r_s \cdot N_s$) and relative mortalities (deaths relative to $r_s \cdot N_s$) for: the selected 2025 risk assessment model; and, the 2024 risk assessment model. 95% CIs are provided in parentheses.

2025 risk assessment model 2024 risk assessment Relative Relative Total deaths mortalities Common name $r_c \cdot N_c$ Total deaths $r_c \cdot N_c$ mortalities Gibson's albatross 438 (305-646) 598 (450-800) 0.72 (0.48-1.14) 940 (701-1 265) 0.65 (0.43-0.97) 606 (444-827) Antipodean albatross 78 (53-115) 551 (407-782) 0.14 (0.10-0.21) 67 (48-96) 655 (499-861) 0.10 (0.07-0.15) Wandering albatross 265 (187-393) 1.879 (1.418-2.513) 0.14 (0.10-0.22) 253 (179-354) 1875 (1403-2594) 0.13 (0.09-0.19) 0.41 (0.28-0.62) Tristan albatross 166 (102-270) 459 (269-770) 0.36 (0.24-0.55) 188 (113-312) 455 (274-771) Amsterdam albatross 0.38 (0.25-0.60) 2 (2-4) 9 (7-13) 0.25 (0.17-0.38) 3(2-6)9 (7-13) 1.049 (685-1,810) Southern royal albatross 0.14 (0.09-0.21) 74 (53-103) 1 146 (712-1 900) 0.06 (0.04-0.11) 149 (93-251) Northern royal albatross 804 (524-1.383) 0.05 (0.04-0.07) 16 (9-26) 834 (567-1 367) 41 (24-70) 0.02 (0.01-0.03) Atlantic yellow-nosed albatross 1,071 (771-1,532) 5,130 (3,741-7,061) 0.21 (0.15-0.30) 91 (63-133) 5 304 (3 965-7 124) 0.02 (0.01-0.02) Indian vellow-nosed albatross* 1,299 (868-1,897) 6.476 (4.542-9.190) 0.20 (0.14-0.30) 943 (702-1.310) 13 901 (10 580-18 427) 0.07 (0.05-0.10) Black-browed albatross* 2.936 (2.275-3.861) 83.375 (65.879-106.184) 0.04 (0.03-0.05) 1,268 (926-1,769) 56 203 (44 501-70 437) 0.02 (0.02-0.03) Campbell black-browed albatross* 226 (158-338) 1,751 (1,309-2,439) 0.13 (0.09-0.20) 449 (332-626) 99 228 (71 446-138 500) 0.00 (0.00-0.01) Shy albatross 149 (103-226) 2,257 (1,547-3,327) 0.07 (0.04-0.10) 128 (84-198) 2 377 (1 656-3 475) 0.05 (0.03-0.08) New Zealand white-capped albatross* 3,445 (2,457-4,820) 14.038 (9.405-20.602) 0.24 (0.16-0.38) 2.158 (1.594-2.937) 28 743 (20 842-39 599) 0.07 (0.05-0.11) 3,929 (2,963-5,390) Salvin's albatross 260 (186-379) 0.07 (0.05-0.10) 127 (84-194) 6 885 (4 841-9 760) 0.02 (0.01-0.03) Chatham Island albatross 41 (29-62) 698 (572-868) 0.06 (0.04-0.09) 12 (8-18) 703 (568-894) 0.02 (0.01-0.03) Grey-headed albatross* 452 (263-949) 12,616 (7,323-27,748) 0.04 (0.03-0.05) 3,169 (2,409-4,250) 95 090 (76 764-118 084) 0.03 (0.02-0.05) Southern Buller's albatross* 344 (236-526) 1,541 (1,220-2,001) 0.23 (0.15-0.33) 2,110 (1,554-2,910) 23 601 (19 122-29 641) 0.09 (0.06-0.13) Northern Buller's albatross 227 (155-330) 2.273 (1.781-2.920) 0.10 (0.07-0.15) 99 (70-142) 2 260 (1 814-2 902) 0.04 (0.03-0.06) Sooty albatross 567 (402-825) 1,780 (1,326-2,453) 0.32 (0.20-0.49) 646 (475-857) 1677 (1193-2315) 0.39 (0.25-0.58) Light-mantled sooty albatross* 169 (99-287) 2,687 (1,782-4,198) 0.06 (0.04-0.11) 306 (220-426) 5 052 (3 505-7 424) 0.06 (0.04-0.09) Grey petrel* 141 (94-213) 17,819 (12,545-26,541) 0.01 (0.01-0.01) 458 (337-636) 35 025 (26 669-46 892) 0.01 (0.01-0.02) Black petrel 36 (25-52) 1,264 (1,073-1,538) 0.03 (0.02-0.04) 38 (26-54) 1 267 (1 069-1 520) 0.03 (0.02-0.04) Westland petrel* 82 (52-131) 1.782 (1.214-2.635) 0.05 (0.03-0.07) 117 (74-181) 1929 (1305-2896) 0.06 (0.04-0.09) White-chinned petrel* 3,562 (2,829-4,610) 263,270 (186,430-382,655) 0.01 (0.01-0.02) 3,167 (2,469-4,076) 148 436 (109 106-200 975) 0.02 (0.01-0.03) 132 (88-197) Spectacled petrel 8,140 (5,857-11,532) 0.02 (0.01-0.02) 374 (263-531) 26 760 (18 315-39 850) 0.01 (0.01-0.02)

^{*} indicates species that had visible posterior updates to biological parameters in the 2024 risk assessment.

Figures

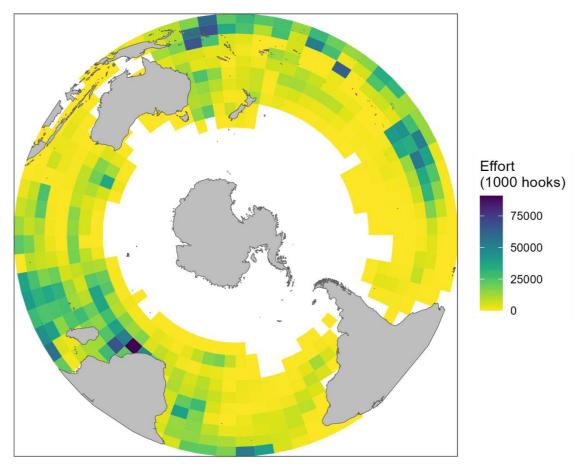
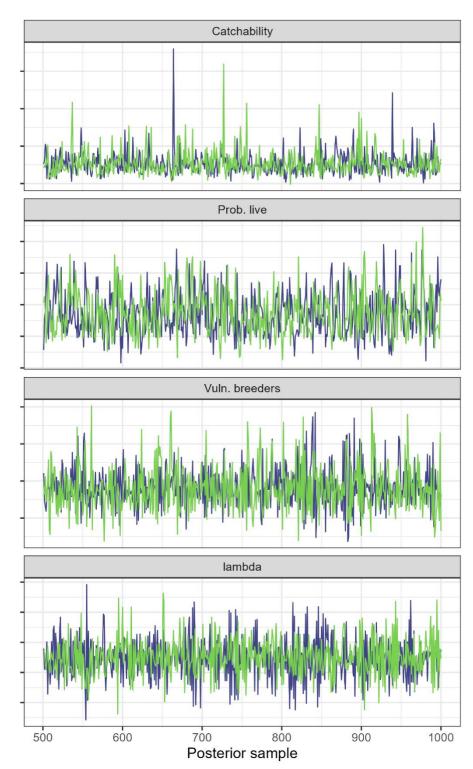


Figure 1: Map of total effort included in the risk assessment (in units of 1000 hooks).



Figure~2:~MCMC~trace~diagnostics~for~the~2025~risk~assessment~model~fit.~For~each~MCMC~chain,~the~Euclidean~norm~is~calculated~for~each~parameter~vector.

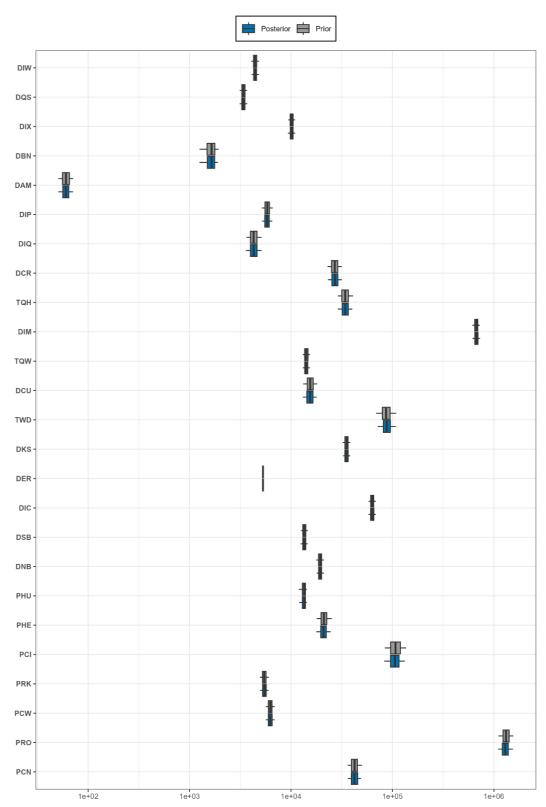


Figure 3: Prior and posterior distributions of the number of breeding pairs per species ($N_s^{\rm BP}$; log-10 transformed) from the selected 2025 risk assessment model.

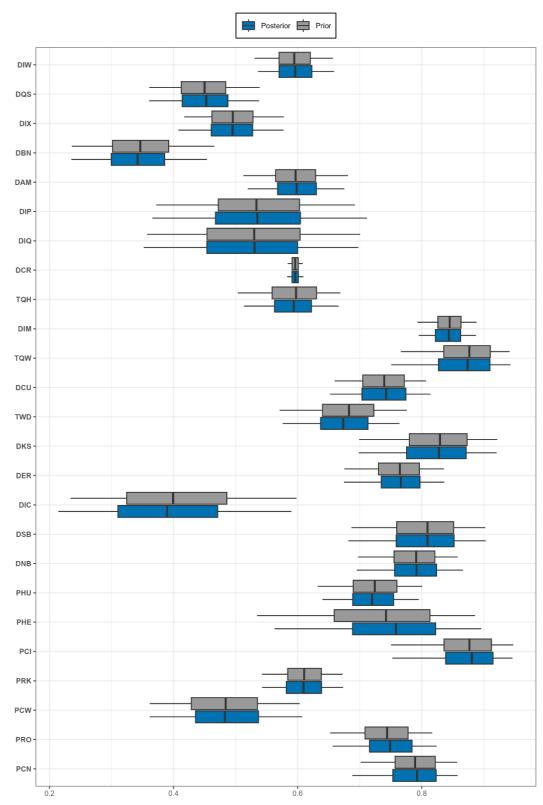


Figure 4: Prior and posterior distributions of the probability of breeding per species ($P_s^{\rm B}$) from the selected 2025 risk assessment model.

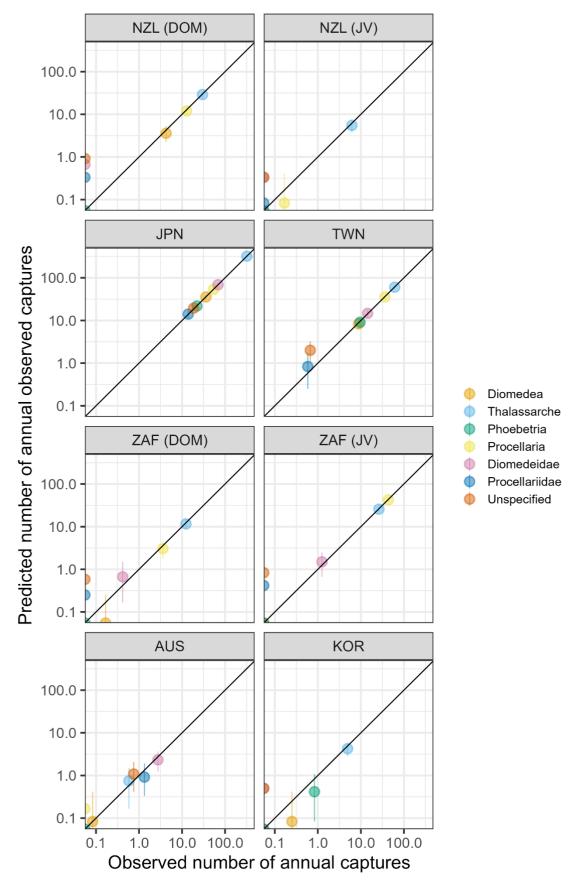


Figure 5: Fit to average annual empirical captures by capture code and fishery group (on the log-10 scale) from the selected 2025 risk assessment model. DOM denotes domestic fleet, and JV Joint Venture operations.

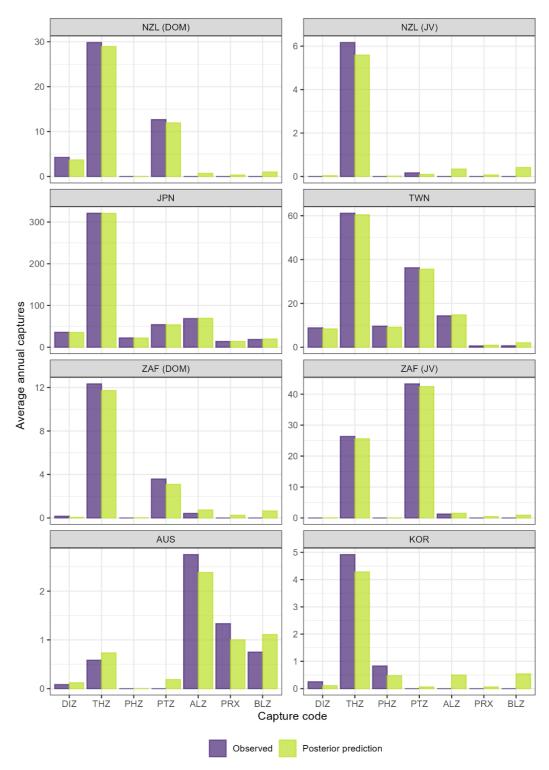
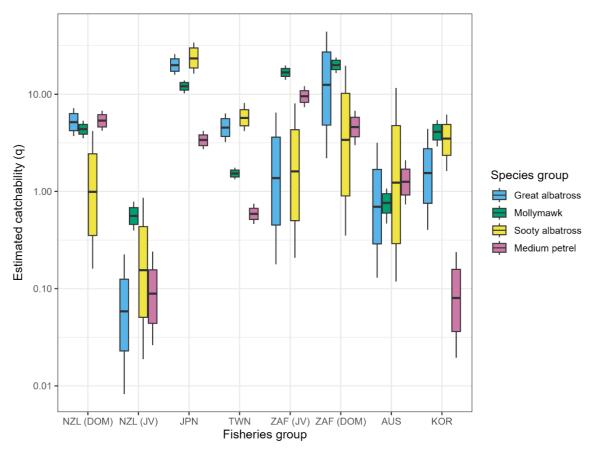


Figure 6: Fit to average annual empirical captures by capture code and fishery group from the selected 2025 risk assessment model. DOM denotes domestic fleet, and JV Joint Venture operations.



Figure~7:~Estimated~catchabilities~per~species~group~and~fishery~group~(on~the~log-10~scale)~from~the~selected~2025~risk~assessment~model.~DOM~denotes~domestic~fleet,~and~JV~Joint~Venture~operations.

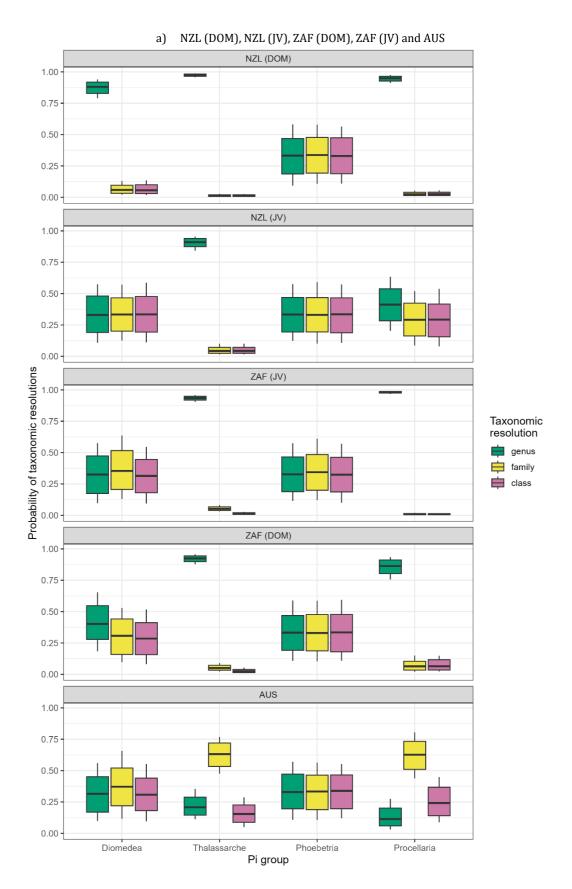


Figure 8: Estimated π per genus and fishery group from the selected 2025 risk assessment model for a) NZL (DOM), NZL (JV), ZAF (DOM), ZAF (JV) and AUS), and b) (continued on next page) JPN, TWN and KOR. DOM denotes domestic fleet, and JV Joint Venture operations.

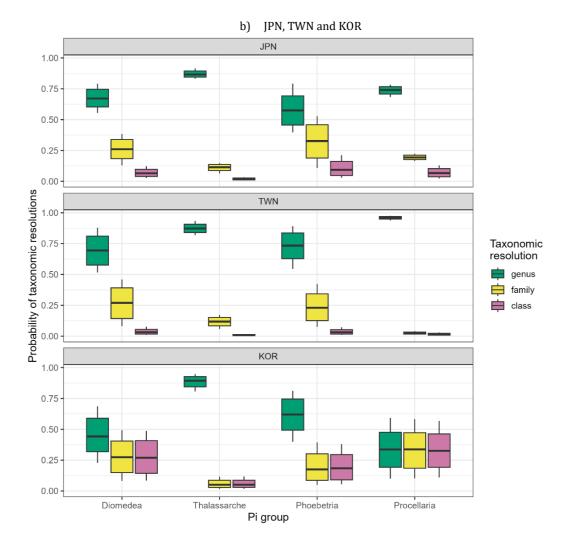


Figure 8 continued.

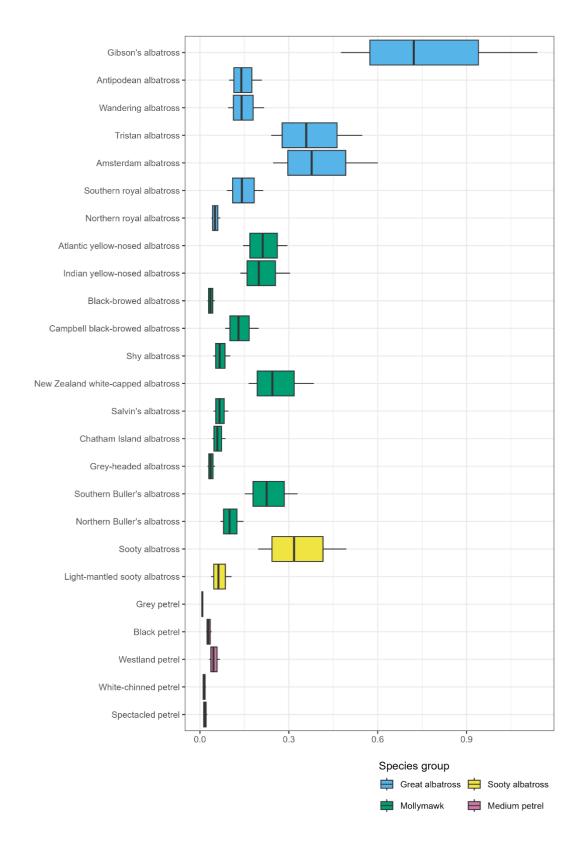


Figure 9: Estimated relative mortality rates per species, i.e., total deaths relative to $r_s \cdot N_s$), from the selected 2025 risk assessment model.

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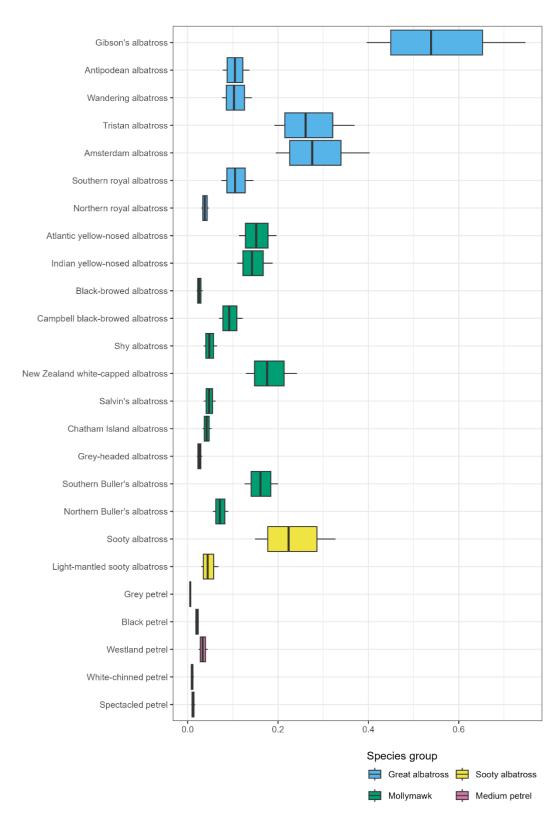


Figure 10: Estimated relative mortality rates per species from observable deaths (i.e., with no cryptic mortalities), relative to $r_s \cdot N_s$, from the selected 2025 risk assessment model.



Figure 11: The spatial distribution of the estimated observed density overlap per species group and fishery group, for a) NZL (DOM), NZL (JV), ZAF (DOM), ZAF (JV) and AUS), and b) (continued on next page) JPN, TWN and KOR. The maps provide the proportion of total estimated observed density overlap per species group by 5°cell per fishery group.

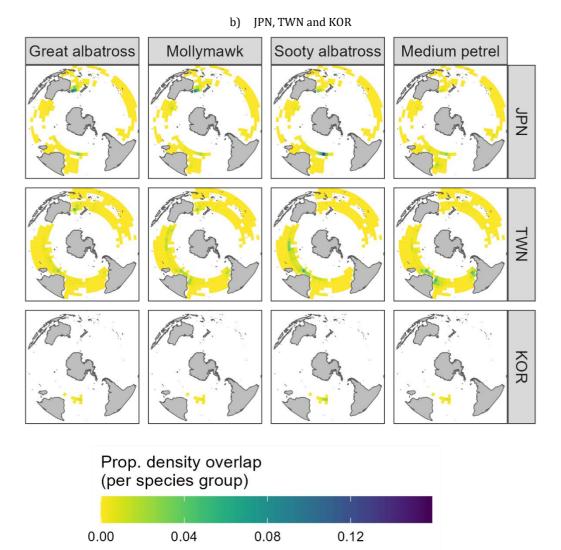


Figure 11 continued.



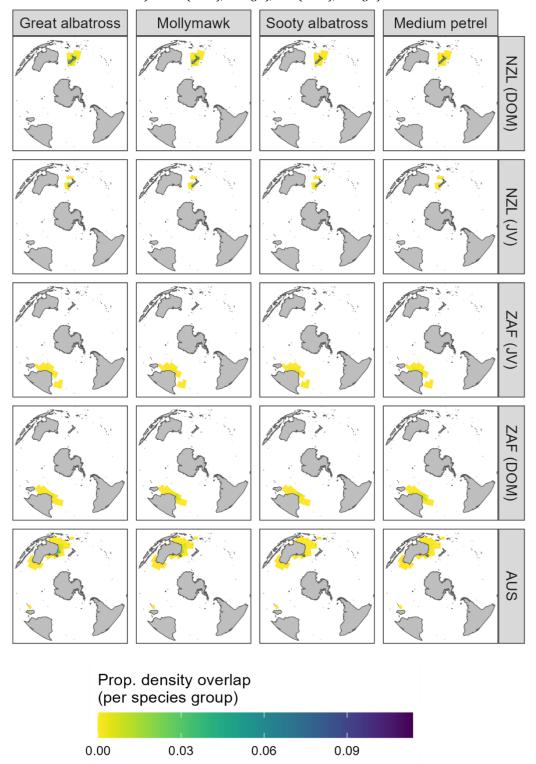


Figure 12: The spatial distribution of estimated total density overlap per species group and fishery group (expressed as the proportion of the total density overlap per species group), for a) NZL (DOM), NZL (JV), ZAF (DOM), ZAF (JV) and AUS), and b) (continued on next page) JPN, TWN and KOR. The maps provide the proportion of total density overlap per species group by 5°cell per fishery group.

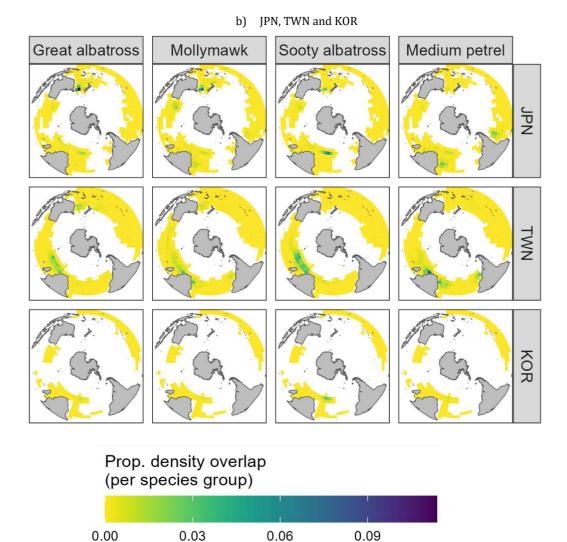


Figure 12 continued.

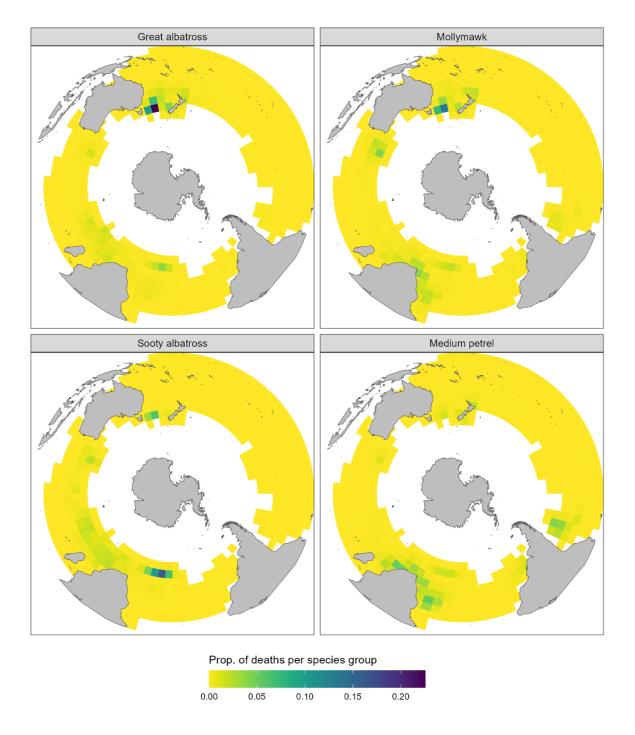


Figure 13: The spatial distribution of total estimated annual deaths per species group from the selected 2025 risk assessment model, provided as the proportion of total annual deaths of each species group by 5°cell.

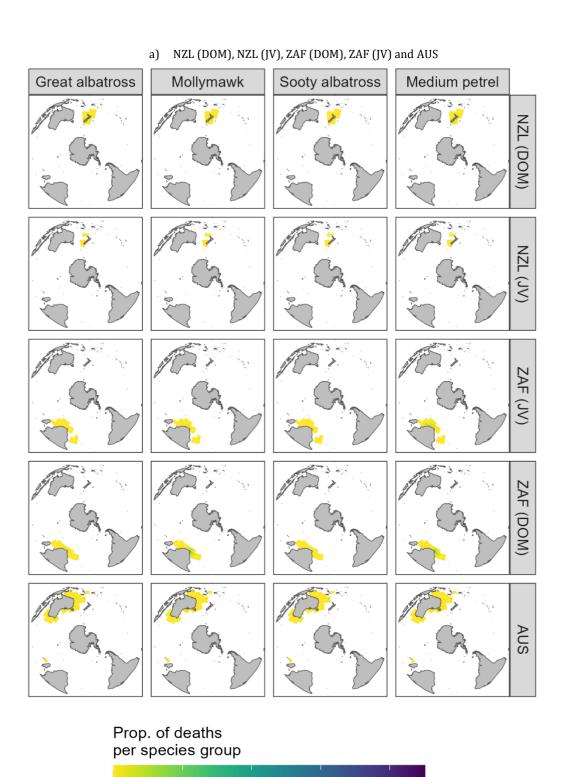


Figure 14: Maps of total estimated annual deaths per species group and fishery group from the selected 2025 risk assessment model, for a) NZL (DOM), NZL (JV), ZAF (DOM), ZAF (JV) and AUS), and b) (continued on next page) JPN, TWN and KOR. The maps provide the proportion of total estimated annual deaths per species group accounted for by each 5°cell per fishery group.

0.15

0.20

0.10

0.00

0.05

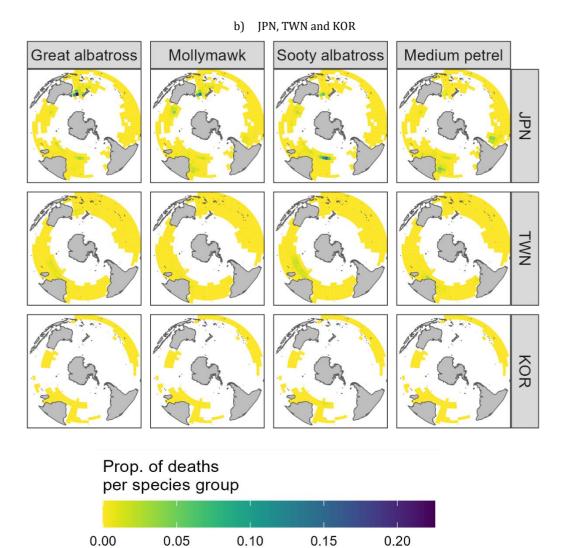
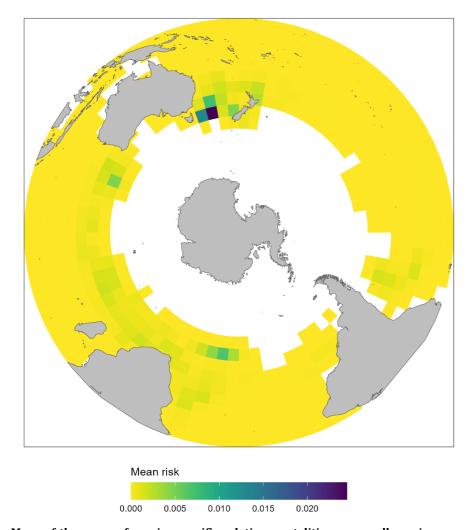


Figure 14 continued.



Figure~15:~Maps~of~the~mean~of~species-specific~relative~mortalities~across~all~species~groups~from~the~selected~2025~risk~assessment~model.

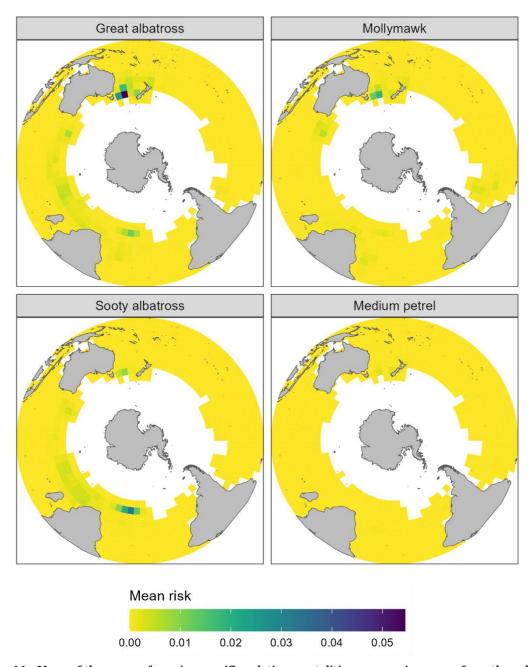


Figure 16: Maps of the mean of species-specific relative mortalities per species group from the selected 2025 risk assessment model.

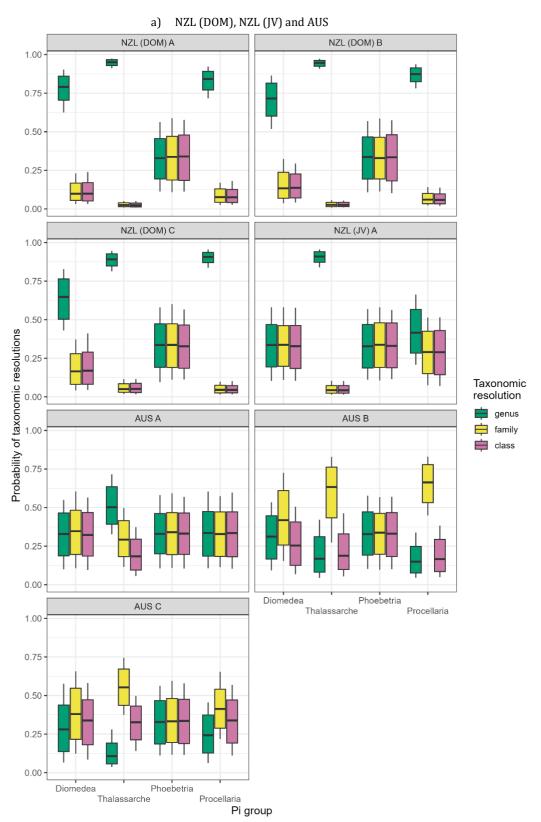


Figure 17: Estimated π for the model with temporal variation in catchabilities and π , for a) NZL (DOM), NZL (JV) and AUS, (continued on next page) b) JPN, TWN and KOR, and c) and (continued on next page) ZAF (DOM) and ZAF (JV). DOM denotes domestic fleet, and JV Joint Venture operations. The suffixes A B and C refer to time periods: (A) 2012 to 2016, (B) 2017 to 2019, and (C) 2020 to 2023.

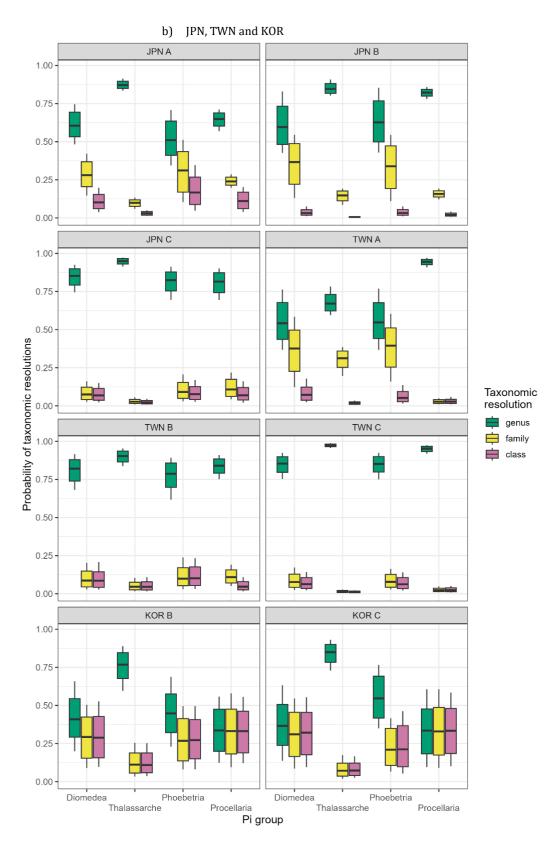


Figure 17 continued.

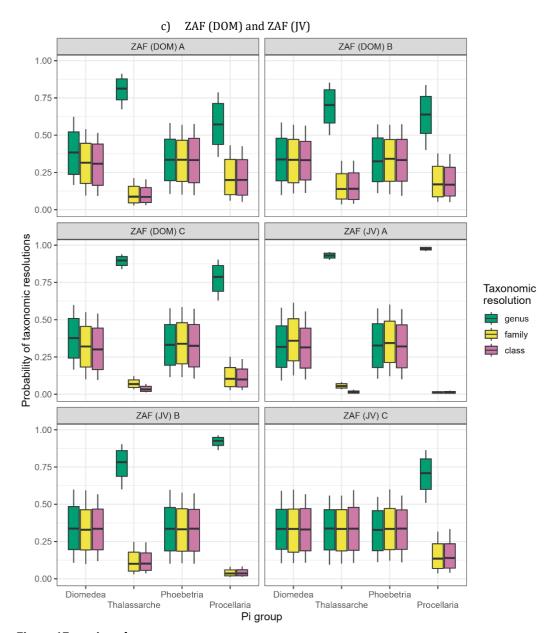


Figure 17 continued.

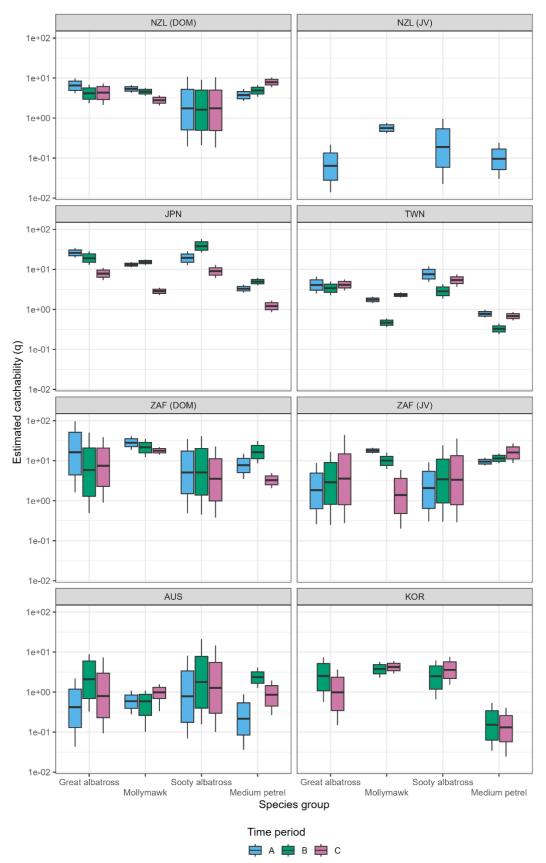


Figure 18: Estimated catchabilities (on the log-scale) for the model with temporal variation in catchabilities and π . DOM denotes domestic fleet, and JV Joint Venture operations. Time periods are: (A) 2012 to 2016, (B) 2017 to 2019, and (C) 2020 to 2023.

Appendix A Report describing updates to density maps.

Appendix B Biological inputs to the seabird risk assessment model