Update on Seabird Ecological Risk Assessment

New Zealand – Compiled by N. Smith from material prepared by N. Walker, M. Cryer, B. Sharp and E. Abraham

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for ERSWG 10 Agenda Item 4.1.3

1. Executive Summary

Progress on a CCSBT area-wide seabird ecological risk assessment since ERSWG 10 has been limited. Several methodological developments have occurred in the related New Zealand seabird ecological risk assessment. In particular substantive progress has been made on approaches to estimate absolute risk (c.f. relative risk). Accordingly it is proposed that upcoming work on a southern hemisphere seabird ecological risk assessment will shift the CCSBT approach from an assessment of relative risk to absolute risk. It would be particularly useful if CCSBT members were able to identify data contributions to such an approach at the 2015 Ecologically Related Species Working Group (ERSWG 11) meeting.

2. Background

2.1 Hierarchical Structure of Risk Assessments (MPI 2014)

Hobday et al. (2007) described a hierarchical framework for ecological risk assessment in fisheries (see Figure 1). The hierarchy included three levels: Level 1 qualitative, expert-based assessments (often based on a Scale, Intensity, Consequence Analysis, SICA); Level 2 semi-quantitative analysis (often using some variant of Productivity Susceptibility Analysis, PSA); and Level 3 fully quantitative modelling including uncertainty analysis. The hierarchical structure is designed to “screen out” potential effects that pose little or low risk for the least investment in data collection and analysis, escalating to risk treatment or higher levels in the hierarchy only for those potential effects that pose non-negligible risk. This structure relies for its effectiveness on a low potential for false negatives at each stage, thereby identifying and screening out activities that are ‘low risk’ with high certainty. This focuses effort on remaining higher risk activities. In statistical terms, risk assessment tolerates Type I errors (false positives, i.e. not screening out activities that may actually present a low risk) in order to avoid Type II errors (false negatives, i.e. incorrectly screening out activities that actually constitute high risk), and it is important to distinguish this approach from normal estimation methods. Whereas normal estimation strives for a lack of bias and a balance of Type I and Type II errors, risk assessment is designed to answer the question “how bad could it be?” The divergence between the risk assessment approach and normal, unbiased estimation approaches should diminish at higher levels in the risk assessment hierarchy, where the assessment process should be informed by good data that support robust estimation.

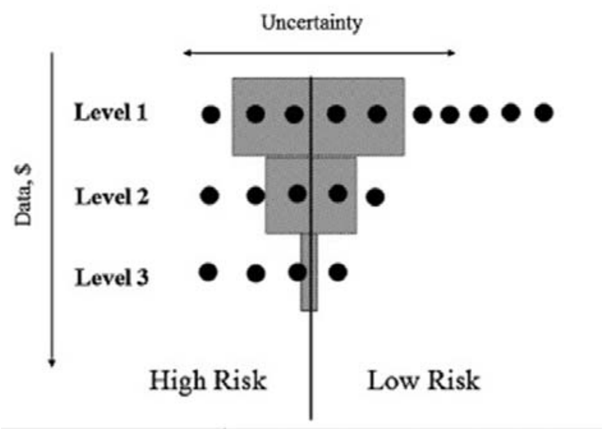
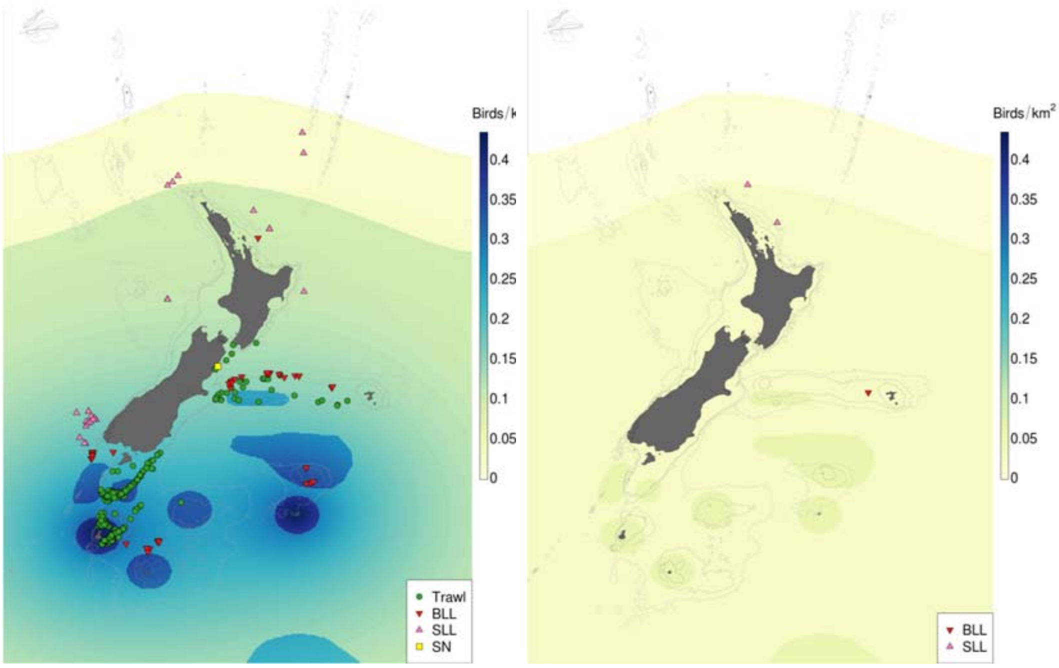


Figure 1: (*from* Hobday et al. 2007): Diagrammatic representation of the hierarchical risk assessment process where activities that present low risk are progressively screened out by assessments of increasingly high data content, sophistication, and cost.

2.2 Semi-Quantitative (Level 2) Risk Assessment (MPI 2014)

The level 2 method developed by MPI is a generalisation of the spatial overlap approach described by Kirby & Hobday (2007) and arose initially from an expert workshop hosted by the then Ministry of Fisheries in 2008 and attended by experts with specialist knowledge of New Zealand fisheries, seabird-fishery interactions, seabird biology, population modelling, and ecological risk assessment. The overall framework is described in Sharp et al. (2011) and has been variously applied and improved in multiple iterations (Waugh et al. 2008a, b, developed further by Sharp 2009, Waugh & Filippi 2009, Filippi et al. 2010, Richard et al. 2011, Richard & Abraham 2013b). The method applies the “exposure-effects” approach where exposure refers to the number of fatalities arising from an activity and effect refers to the consequence of that exposure for the population. The relative encounter rate of each seabird taxon with each fishery group is estimated as a function of the spatial overlap between seabird distributions (e.g., Figure 2) and fishing effort distributions (e.g., Figure 3). These estimates are compared with observed captures in an integrated model including all seabird groups and fisheries to estimate vulnerability (capture rates per encounter) and total captures by taxon in each fishery group. All captures are assumed fatal because of the unknown survival rate of birds released alive. Potential fatality estimates also include scalars for cryptic mortality and are subsequently compared with population estimates and biological characteristics to yield estimates of population-level risk from fishing (see method diagram in Figure 4).



(a) Breeding distribution **(b) Non-breeding distribution**

Figure 2: (from Richard & Abraham 2014 supplementary material) Relative density of white-chinned petrel. The base map for the distribution was obtained from the NABIS database. The breeding season runs from October to May. Also shown are incidental captures recorded by observers between 2006–07 and 2012–13 in trawl, surface-longline (SLL), bottom-longline (BLL), and set-net (SN) fisheries.

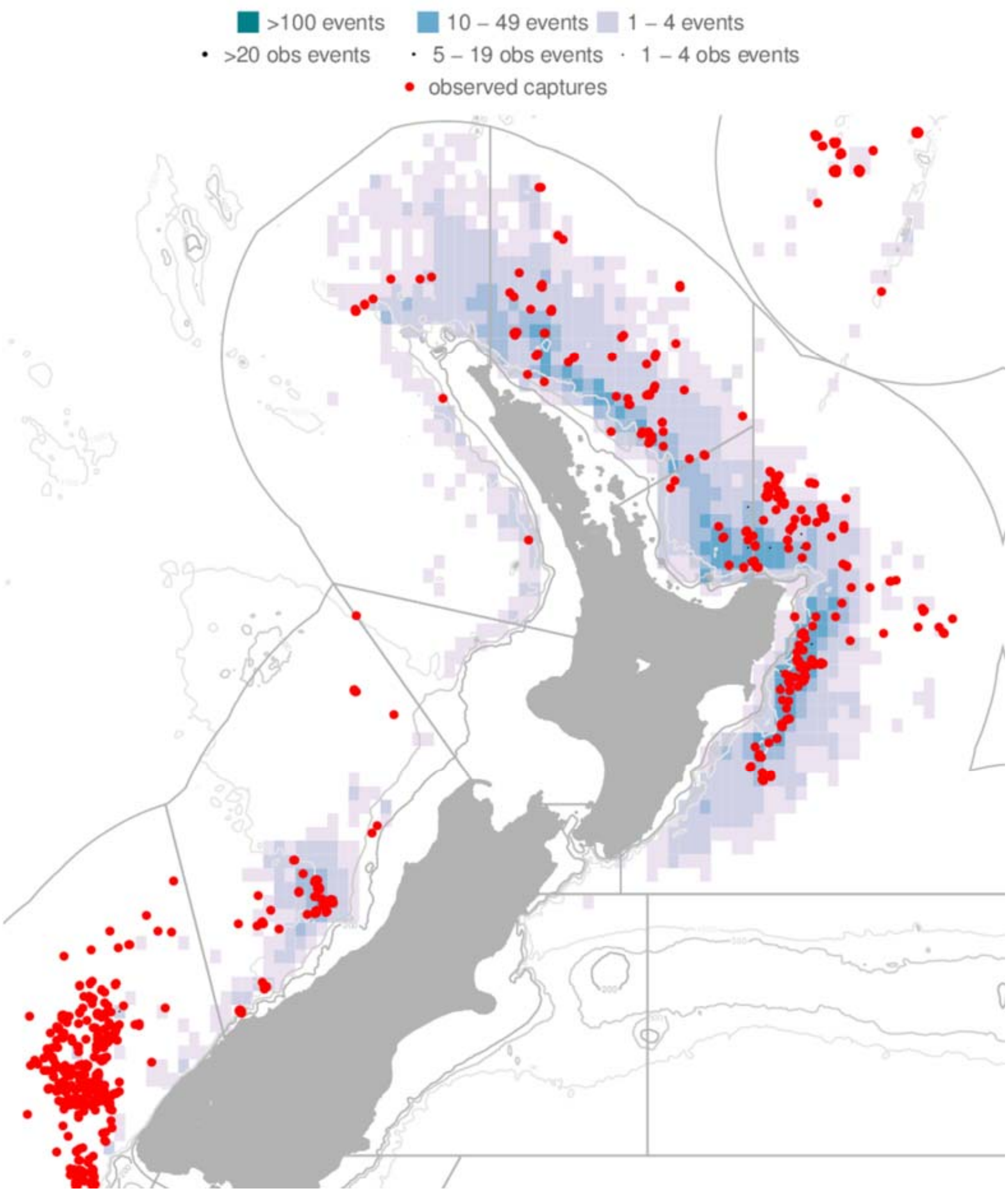


Figure 3: (from Abraham et al. 2013) Map of surface longline fishing effort and all observed seabird captures by surface longlines, October 2002 to September 2013. Fishing effort is mapped into 0.2-degree cells, with the colour of each cell being related to the amount of effort (events). Observed fishing events are indicated by black dots, and observed captures are indicated by red dots. Fishing is shown only if the effort could be assigned a latitude and longitude, and if there were three or more vessels fishing within a cell (here, 89.4% of effort is displayed).

For each taxon, the risk was assessed by dividing the estimated number of annual potential fatalities (APF) by an estimate of Potential Biological Removals (PBR, after Wade 1998). This index represents the amount of human-induced mortality a population can sustain without compromising its ability to achieve and maintain a population size above its maximum net productivity (MNPL) or to achieve rapid recovery from a depleted state. In the risk assessment, PBR was estimated from the best available information on the demography of each taxon, including the seasonality of the distribution of various species where applicable (Figure 2). Because estimates of seabirds' demographic parameters and of fisheries related mortality are imprecise, the uncertainty around the demographic and mortality estimates was propagated through the analysis. This allowed uncertainty in the resulting risk to be calculated, and also allowed the identification of parameters where improved precision would reduce overly large uncertainties. However, not all sources of uncertainty could be included, and the results are best used as a guide in the setting of management and research priorities. In general, seabird demographics, the distribution of seabirds within New Zealand waters, and sources of cryptic mortality were poorly known.

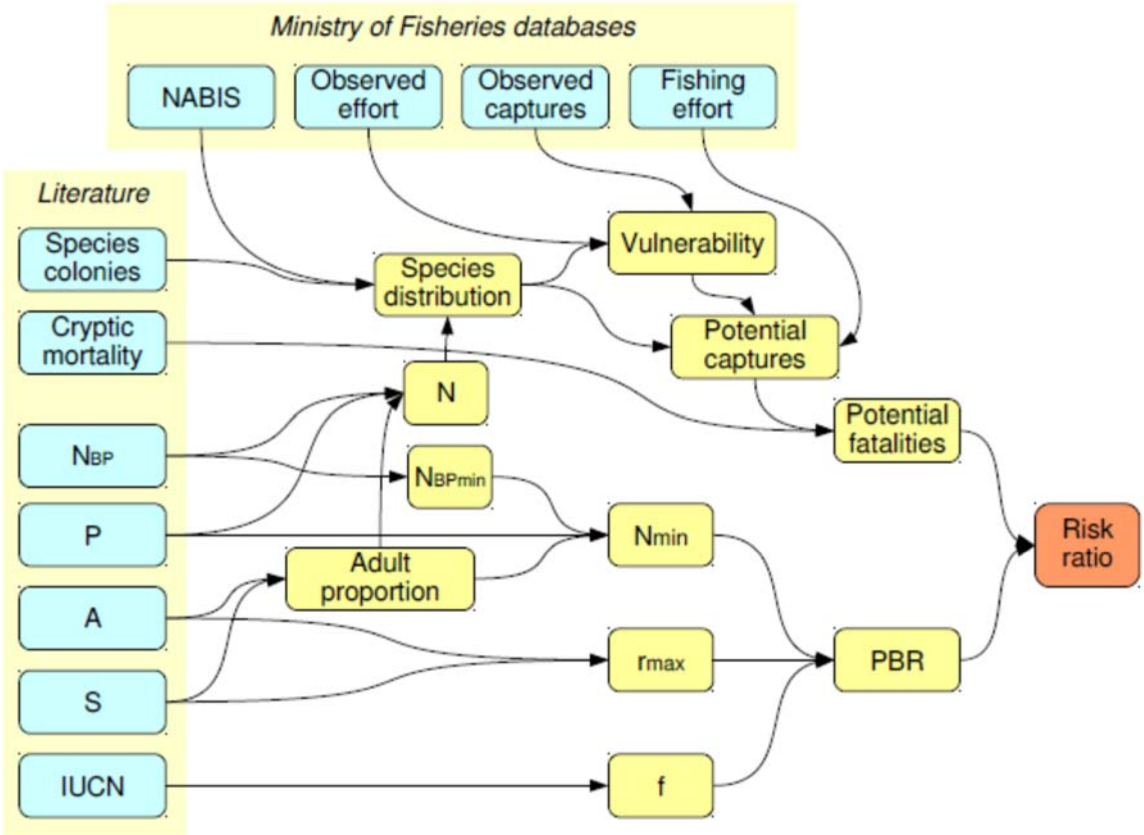


Figure 4: (reproduced from Richard et al. 2011): Diagram of the modelling approach to calculate the risk index for each taxon. NBP, number of annual breeding pairs; N, total number of birds over one year old; NBPmin, lower 25% of the distribution of NBP; Nmin, lower 25% of the distribution of the total number of birds over one year old; rmax, maximum population growth rate; f, recovery factor; PBR, Potential Biological Removal (set to 1.0 by Richard & Abraham 2013b); P, proportion of adults breeding in a given year; A, age at first reproduction; S, annual adult survival rate.

Integral to Richard & Abraham's (2013b) update of the semi-quantitative risk assessment was a simulation study (Richard & Abraham 2013a) to assess the accuracy of the approximations used in PBR calculations used by Richard et al. (2011) for seabird demographics. They showed that the PBR is typically overestimated, largely because rmax is overestimated by Niel & Lebreton's (2005) approximation. Richard & Abraham (2013a) therefore recommended that an additional calibration factor be included in the calculation of the PBR to correct the approximation. The calibration factor varied between 0.17 and 0.61, depending on the seabird type; in general, the calibration factor was smaller for species with slower population growth rates, such as albatrosses, and higher for species with higher growth rates, such as shags and penguins. Previous estimates of the PBR using Niel & Lebreton's (2005) approximation for seabird populations that did not include this calibration factor are likely to have overestimated the human caused mortalities that the populations could support (Richard & Abraham 2013a).

The management criterion used for developing the seabird risk assessment was that seabird populations should have a 95% probability of being above half the carrying capacity after 200 years, in the presence of ongoing human-caused mortalities, and environmental and demographic stochasticity (Richard & Abraham 2013b). By simulating seabird populations, the factor ρ was calculated so that this criterion would be satisfied, provided human caused mortalities were less than the base PBR (the PBR with a recovery factor, f, of 1 and using the population size rather than a minimum population estimate). In calculation of the PBR during the simulations, the Neil & Lebreton (2005) method was used for estimating rmax, and the Gilbert (2009) method was used for estimating total population size. The simulations did not allow for any bias in the input parameters for individual populations.

Calculation of the PBR for a seabird species requires specification of the recovery factor, f. This factor is typically set between 0.1 and 0.5 and can be used for several purposes (e.g. Lonergan 2011). It can be used to “protect” against errors in the input data used to calculate the PBR for individual populations, to provide for faster recovery rates, and to reflect general risk aversion (especially for endangered species). For the 2013 update to the risk assessment, Richard & Abraham (2013a b, 2014) set the recovery factor to f = 1 and suggested that appropriate values for each species should be determined at a later stage.

Following the completion of the 2013 iteration of the level two seabird risk assessment (Richard & Abraham 2013b), the Ministry for Primary Industries convened an expert workshop in November 2013 to review the level two seabird risk assessment inputs and results (Walker et al. 2015). That workshop systematically reviewed input data and other available information for the 26 seabird taxa with the highest risk ratios as assessed by the level two risk assessment. In summary, the results of the workshop were that:

- risk appeared to be overestimated for fourteen taxa, including black petrel
- risk appeared to be reasonably estimated for nine taxa, and
- risk appeared to be underestimated for three taxa: New Zealand king shag and Gibson’s and Antipodean albatrosses.

A general preponderance of overestimated risk is acceptable in a risk assessment framework so long as results are used carefully. Risk assessments are generally designed to be conservative in order to highlight gaps in information to direct future research accordingly. In contrast, any persistent significant underestimation of risk across many species is more problematic as a species may then not be subject to the additional research or management intervention required. Note however that the spatially explicit risk assessment framework is used not only to identify which species are potentially at risk, but also to inform choices about the likely effectiveness of various management options to reduce that risk, and to prioritise further research. In this context over-estimated risk scores for a particular species, fishery group, or area may lead to sub-optimal prioritization, and ultimately delay risk reduction interventions for those species genuinely at risk. For this reason, modification to improve the level two risk assessment consistent with the recommendations of this workshop was considered a high priority for all at-risk species, regardless of whether those modifications are expected to produce a decrease or an increase in overall species-level risk.

Where current risk estimates were thought to be biased in either direction, this workshop did not seek to replace or modify the existing risk estimates for each taxon, but rather gave advice on how to improve the risk assessment at the next iteration under the existing framework, and made some recommendations for further research.

3. Implementation

3.1 New Zealand in 2014 (MPI 2014)

The 2014 iteration of the level two seabird risk assessment for New Zealand (as described by Richard & Abraham 2014) estimated the risk posed to each of 70 seabird taxa by trawl, longline and set net fisheries within New Zealand's TS and EEZ. Substantial modifications to the 2013 iteration of the risk assessment (Richard & Abraham 2013b) were made in the 2014 iteration following the recommendations of the review workshop (Walker et al. 2015), these included:

- re-attribution of species captures data to more likely taxa based on location and season information
- updates to population size, taxa specific parameters, at-sea distributions and timing of the breeding season in the next risk assessment based on new sources of information and recent studies
- increased partitioning/disaggregation of taxa for estimation of vulnerability based on improving observer data availability
- increased partitioning/disaggregation of fisheries to improve resolution of result for management purposes and better reflect at sea practices, and
- the introduction of a parameter to describe the proportion of birds that remain in New Zealand waters during the non-breeding season, instead of treating birds as absent during the non-breeding season.

Other changes included in the 2014 iteration of the risk assessment (Richard & Abraham 2014) were that the data on fishing effort and observed captures included two more years, and vulnerability was estimated using data between the 2006–07 and 2012–13 fishing years. Also, the APFs were estimated on data between 2010–11 and 2012–13 fishing years to reflect the current level and spatial distribution of fishing effort.

While re-running the risk assessment, errors were found in the calculations used in the 2013 iteration of the risk assessment (Richard & Abraham 2013b). These affected both the calculation of the PBR1, and the estimates of APF. An error during data preparation led to over-counting the effort observed in the poorly observed inshore trawl fishery. Also PBR1 was not calculated using the lower quartile of the distribution of the number of annual breeding pairs as documented by Richard & Abraham (2013b).

In order to be confident in the integrity of the risk assessment, the PBR calculations were independently checked. A parser was written to read the input parameters, independently repeat the PBR calculations, and then confirm that the PBR1 values could be reproduced. In repeating the calculation, the mean value of PBR1 was calculated, by repeatedly drawing sets of 4000 samples from each of the distributions to derive a distribution of mean PBR1 values. For each species, it was confirmed that the mean value lay within the 95% confidence interval of the resulting distribution (Richard & Abraham 2014). In addition, the code used for the calculations was reviewed and the PBR calculations were independently checked; no further errors were found (Webber 2014).

Richard & Abraham (2014) repeated the 2013 risk assessment after correcting these errors, and following the recommendations from the review workshop (Walker et al. 2015) to the risk assessment structure and inputs. The overall changes between the corrected 2013 risk assessment and the 2014 iteration can be seen in Figure 5. The progressive change in risk ratio to each successive change in the risk assessment structure and input parameters is given in Figure 6.

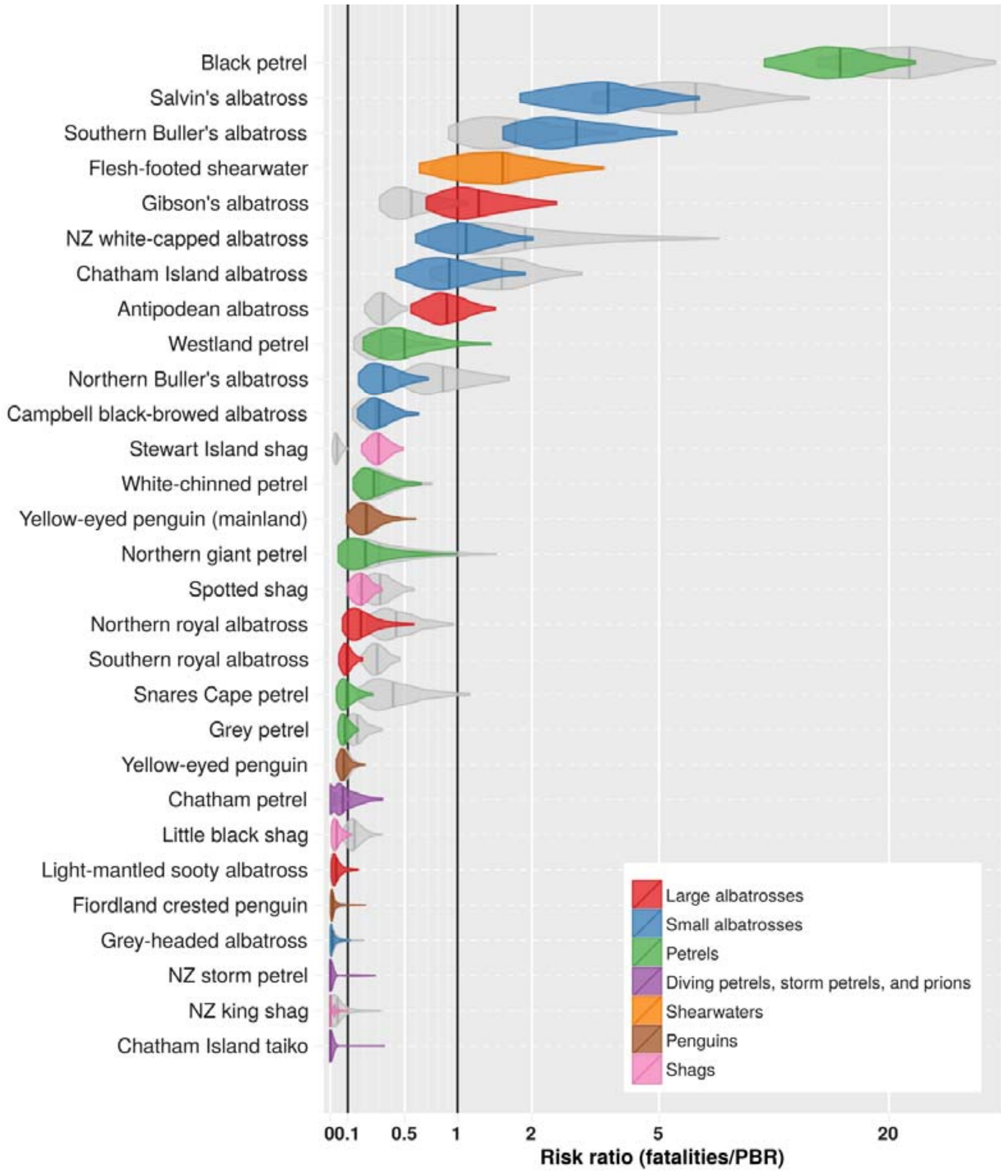


Figure 5: (reproduced from Richard & Abraham 2014) Risk ratio, updated to include data from the 2011–12 and 2012–13 fishing years. The risk ratio is displayed on a logarithmic scale, with the threshold of the number of potential bird fatalities equalling the PBR with f = 0:1 and f = 1 indicated by the two vertical black lines, and the distribution of the risk ratios within their 95% confidence interval indicated by the coloured shapes, including the median risk ratio (vertical line). Seabird species are listed in decreasing order of the median risk ratio. Species with a risk ratio of almost zero were not included (95% upper limit with f = 1 less than 0.1). The risk ratio of yellow-eyed penguin refers to the mainland population only, based on the assumption that all estimated fatalities were of the mainland population, and the number of annual breeding pairs was between 600 and 800. The grey shapes indicate the risk ratios from the previous assessment (Richard & Abraham 2013b), corrected for errors, to show the change in risk since the 2010–11 fishing year.

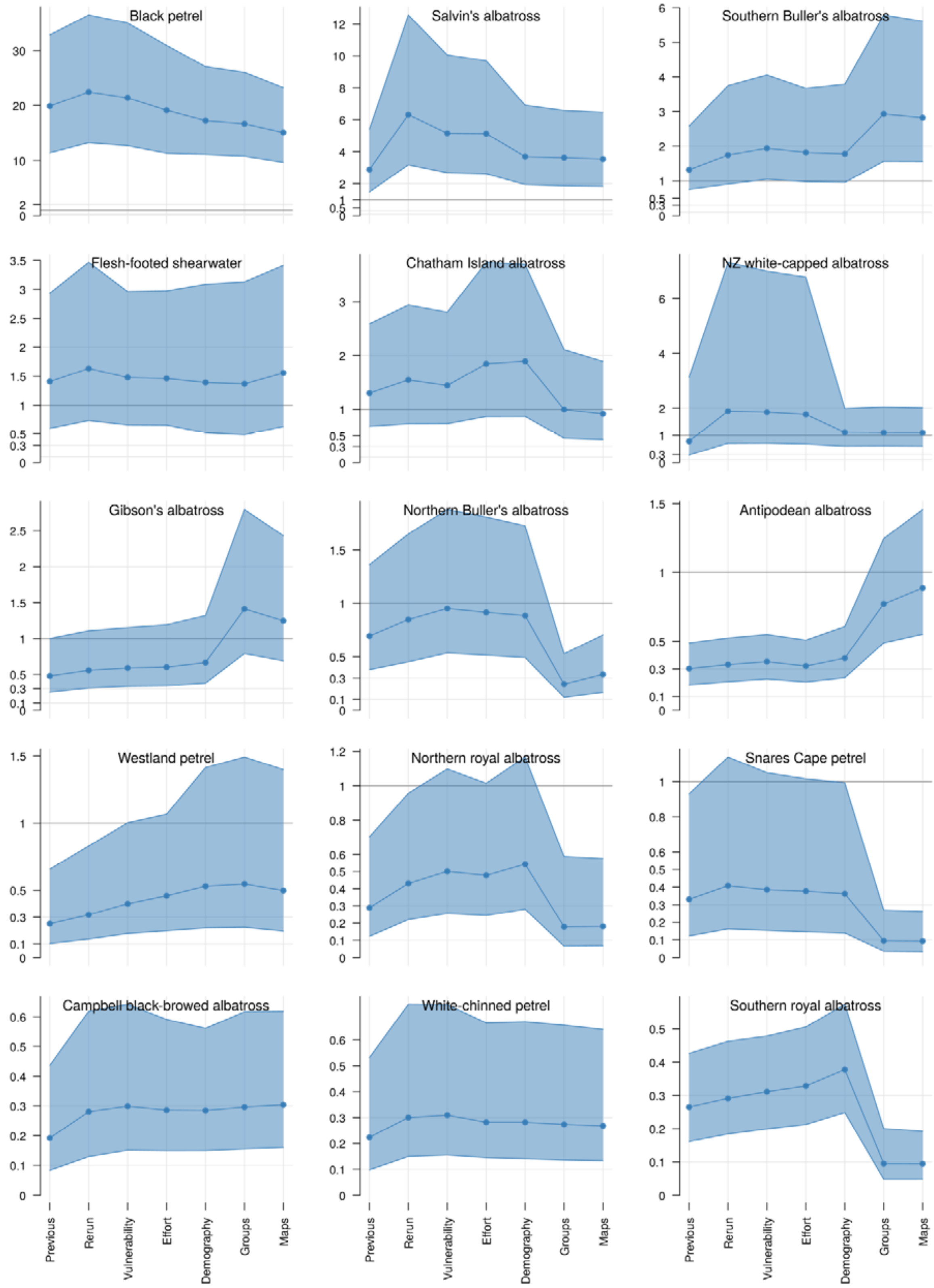


Figure 6: (reproduced from Richard & Abraham 2014) Progressive changes in the risk ratio. Previous: previous assessment (2013); Rerun: same years as 2013; Vulnerability: new fishing data, vulnerability estimated on 7 years, APFs on same 5 years as in previous assessment; Effort: effect of change in effort, vulnerability estimated on 7 years, APFs on last 3 years; Demography: updated demographic parameters; Groups: updated species and fishery groups; Maps: updated distribution maps. For the 15 species the most at risk.

The method described by Richard et al. (2011) and Richard & Abraham (2013b, 2014) offers the following advantages that make it particularly suitable for assessing risk to multiple seabird populations from multiple fisheries:

- risk is assessed separately for each seabird taxon;
- fisheries managers must assess risk to seabirds with reference to units that are biologically meaningful;
- the method does not rely on the existence of universal or representative fisheries observer data to estimate seabird mortality (fisheries observer coverage is generally too low and/or too spatially unrepresentative to allow direct impact estimation at the species or subspecies level);
- the method can be applied to any fishery for which at least some observer data exists;
- the method does not rely on detailed population models (the necessary data for which are unavailable for the great majority of taxa) because risk is estimated as a function of population-level potential fatalities and biological parameters that are generally available from published sources;
- the method assigns risk to each taxon in an absolute sense, i.e. taxa are not merely ranked relative to one another; this allows the definition of biologically meaningful performance standards and ability to track changes in performance over time and in relation to risk management interventions;
- risk scores are quantitative and objectively scalable between fisheries or areas, so that risk at a population level can be disaggregated and assigned to different fisheries or areas based on their proportional contribution to total impact to inform risk management prioritisation;
- the method allows explicit statistical treatment of uncertainty, and does not conflate uncertainty with risk; numerical inputs include error distributions and it is possible to track the propagation of uncertainty from inputs to estimates of risk; and
- the method readily incorporates new information; assumptions in the assessment are transparent and testable and, as new data becomes available, the consequences for the subsequent impact and risk calculations arise logically without the need to revisit other assumptions or repeat the entire risk assessment process.

The key disadvantages of the method of Richard et al. (2011), many of which were addressed by subsequent iterations (Richard & Abraham 2013b, 2014), were that:

- fishing methods for which no observer information on seabird interactions is available cannot be easily included in the analysis;
- the assumption that the vulnerabilities of particular seabirds to capture in different fisheries are independent does not allow “sharing” of scarce observer information between fisheries within the risk assessment (addressed in subsequent iterations);
- the spatial overlap method relies on appropriate spatial and temporal scales for the distributions of birds and fishing effort being used; use of inappropriate scales can lead to misleading results (partially addressed in 2013 revision);
- strong assumptions have to be made about the distribution and productivity of some taxa, the relative vulnerability of different taxa to capture by particular fisheries, cryptic mortality associated with different fishing methods, and the applicability of the

allometric method of estimating Potential Biological Removals (partially addressed in 2013 revision).

Most of these limitations are a result of the scarcity of relevant data on seabird populations and fisheries impacts and can be addressed only through the collection of more information or, in some cases, sensitivity testing. Further refinement of this method would be possible if:

- estimates of PBR could be compared with total annual human caused mortality rather than mortality from commercial fishing within the New Zealand region
- little is known about the impact of New Zealand recreational fishing on seabirds or fatalities in overseas fisheries of seabirds that forage beyond New Zealand's waters
- better information on cryptic mortality was available - studies on cryptic mortality are extremely limited, and
- further observer coverage was targeted at fisheries where substantial reductions in the uncertainty about potential fatalities would result (most such fisheries are poorly observed).

It should be noted that the level two risk assessments conducted thus far (Richard et al. 2011, Richard & Abraham 2013b, 2014) includes APFs in commercial fisheries within New Zealand's EEZ but excludes non-commercial impacts, fatalities on the High Seas and in other jurisdictions, and all other anthropogenic sources of mortality. Because of this focus and the definition of PBR as a level of mortality that can support all anthropogenic sources of mortality and still lead to good population outcomes, the risk ratios estimated by Richard & Abraham (2014) will be underestimates of the total risk faced by each taxon and interpretation should be in this context. Many of the other anthropogenic sources of mortality excluded from the risk assessment are poorly understood.

3.2 The 2013 CCSBT risk assessment

The 2013 CCSBT risk assessment (CCSBT-ERS/1308/18) used improved spatial seabird distribution data layers utilising all available satellite tracking data for key species. The risk scores were a combination of productivity and susceptibility. Productivity is a function of the seabird's biology and breeding patterns. The susceptibility index was the product of the fishing distribution and the species distribution (spatial overlap) multiplied by the vulnerability of the species to the longline fishing gear. The results indicated that species at highest risk are primarily large albatrosses at temperate latitudes, followed by smaller albatrosses. Geographical areas of highest risk include the Tasman Sea and the area around New Zealand, primarily in the austral autumn and winter.

The challenge was to progress from the 2013 assessment to a risk assessment which provided absolute levels of risk for taxa, incorporated improved catch effort information so as to better identify higher risk areas more precisely, and to incorporate up to date observer data to support the definition of area and fishery specific vulnerability estimates for each seabird species or guild. As described in section 3.1 above, several methodological developments have occurred in the related New Zealand seabird ecological risk assessment. In particular substantive progress has been made on approaches to estimating absolute risk (c.f. relative risk).

4. Continuing seabird ecological risk assessment research

To progress a CCSBT area-wide seabird ecological risk assessment is challenging. The work completed to date (CCSBT-ERS/1308/18, described in section 3.1 above) has shown that is practical, at least for the determination of relevant levels of risk between taxa. This section is focussed on work underway and next steps. It provides updates on progress with issues identified in the 2014 New Zealand assessment and by ERSWG10.

4.1 Scarcity of information on cryptic mortality (MPI 2014)

Cryptic mortality is particularly poorly understood but has substantial influence on the results of the risk assessment. Richard et al. (2011) provided a description of the method used to incorporate cryptic mortality into their estimates of potential fatalities in the level-2 risk assessment (their appendix B authored by B. Sharp, MPI). This method builds on the published information from Brothers et al. (2010) for longline fisheries and Watkins et al. (2008) and Abraham (2010) for trawl fisheries. Brothers et al. (2010) observed almost 6000 seabirds attempting to take longline baits during line setting, of which 176 (3% of attempts) were seen to be caught. Of these, only 85 (48%) were retrieved during line hauling. They concluded that using only observed captures to estimate seabird fatalities grossly underestimates actual levels in pelagic longline fishing.

Given the relatively small sample sizes there is substantial (estimatable) uncertainty in the estimates from the trial and additional (non-estimatable) uncertainty related to the extent to which the trial is representative of all fishing of a given type. The binomial 95% confidence range (calculated using the Clopper-Pearson “exact” method) for the ratio of total fatalities to observed captures in the Brothers et al. (2010) longline trial is 1.8–2.5 (mean 2.1 fatalities per observed capture). Some of this uncertainty is included and propagated in the most recent New Zealand risk assessment (Richard & Abraham 2013b).

A review of available information on cryptic mortality has been commissioned in New Zealand under CSP project INT2013-05 and supported by MPI project PRO2012-17. The final report was not available to be included in this report.

4.2 Seabird population range based ecological risk assessments (MPI 2014)

Robertson et al. (2003) mapped the distribution of the 25 breeding (mainly endemic) New Zealand seabird taxa they considered most at risk outside New Zealand waters. These ranged widely: 4 used the South Atlantic; 4 the Indian Ocean; 22 Australian waters and the Tasman Sea; 15 used the South Pacific Ocean as far afield as Chile and Peru; and 6 used the North Pacific Ocean as far north as the Bering Sea. These taxa therefore use the national waters of at least 18 countries. For example, the level-2 risk assessment described by Richard et al. (2011) includes only that part of the range of each taxon contained within New Zealand waters, but many including commonly-caught seabirds like white-capped albatross and white-chinned petrel range much further and are vulnerable to fisheries in other parts of the world. For instance, fatalities of white-capped albatross outside the New Zealand EEZ greatly exceed fatalities within the zone (Baker et al. 2007, Francis 2012), and more than 10 000 white-chinned petrel have been estimated as killed off South America each year (Phillips et al. 2006), noting that reliable records are not available for most of the fisheries involved. Also note that white chinned petrels also breed on Prince Edward and Falkland Islands, South Georgia, Iles Crozet, and the Kerguelen group, so South American captures may be from other populations other than New Zealand’s. Based on similar analyses, Moore & Zydelis (2008) concluded that a population-based, multi-gear and multinational framework is required to identify the most significant threats to wide-ranging seabird populations and to prioritize mitigation efforts in the most problematic areas.

The seabird tracking data available from Birdlife International provide a substantive and robust datasource for a broad range of seabirds (as used in CCSBT-ERS/1308/18). An updated output from Birdlife International will be essential for future CCSBT area-wide seabird ecological risk assessment.

4.3 Further development of the risk assessment framework (MPI 2014)

New Zealand has committed to undertake a further two iterations of the New Zealand seabird level two risk assessment in 2015 and 2016. That work will include another review workshop such as undertaken in 2013 to assess the risk assessment structure and input parameters. Additional analyses for the 2015 iteration of the risk assessment may include, but are not limited to:

- improving the distribution of uncertainty around NBPmin
- investigating the ability of the risk assessment to detect changes in vulnerability over time and determining the required level of coverage and observed captures to allow changes in vulnerability to be calculated
- simulations to test the ability of the risk assessment to detect/predict capture levels
- further consideration of the ρ factor, including the application of ρ where different information on abundance and overlap is used and determining whether and to what extent the use of NBPmin leads to a bias in PBR1.

The analyses will also include each new year of catch effort and observer data, and any newly published information on populations and their distributions.

5. Next steps for CCSBT ERS seabird ecological risk assessment research

Considerable progress has been made with method development and review of approaches to seabird risk assessment. As a result, New Zealand is currently in the process of commissioning a “global” seabird risk assessment to include at least the commercial fishing components of within zone and high seas fisheries across the Southern Hemisphere. The coverage of fisheries and taxa is still being finalised and it is unlikely that it will be truly global in the first iteration. However, it is highly likely that surface longline fisheries and CCSBT fishing will be part of the first iteration. The project remains open to collaboration from other researchers.

To address the issues identified at ERSWG10 the approach will provide absolute levels of risk for taxa, incorporate improved catch effort information so as to better identify higher risk areas more precisely, and incorporate up to date observer data to support the definition of area and fishery specific vulnerability estimates for each seabird species or guild.

It would be particularly useful if CCSBT members were able to identify data contributions to such an approach at the 2015 Ecologically Related Species Working Group (ERSWG 11) meeting. It would be particularly beneficial if the approaches to access and use such data in a CCSBT ERS seabird ecological risk assessment were identified.

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