Ministry for Primary Industries
Manatū Ahu Matua

Independent review of New Zealand's Spatially Explicit Fisheries Risk Assessment approach – 2017

New Zealand Fisheries Science Review 2017/2

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Preface

The Ministry for Primary Industries and its predecessor, the Ministry of Fisheries, have conducted fully-independent expert reviews of stock assessments, research methodologies and research programmes since 1998. We also run specialist technical review workshops to further advance fisheries and other marine science methodologies and techniques. These fully-independent reviews and technical workshops are separate from, but complementary to, the annual Science Working Group processes that are used to ensure the objectivity and reliability of most of our scientific research and analyses.

A new publication series, Fisheries Science Reviews, has been initiated in 2015 to ensure that reports from these reviews are readily accessible. The series will include all recent and new fully-independent reviews and technical workshop reports, and will also incorporate as many historical reports as possible, as time allows. In order to avoid confusion about when the reviews were actually conducted, all titles will include the year of the review. They may also include appendices containing the Terms of Reference, a list of participants, and a bibliography of supporting documents, where these have not previously been incorporated. Other than this, there will be no changes made to the original reports composed by the independent experts or workshop participants.

Fisheries Science Reviews (FSRs) contain a wealth of information that demonstrates the utility of the processes the Ministry uses to continually improve the scientific basis for managing New Zealand's fisheries.

Lonergan, M.E.; Phillips, R.A.; Thomson, R.B.; Zhou, S. (2017). Independent review of New Zealand's spatially explicit fisheries risk assessment approach – 2017. *New Zealand Fisheries Science Review 2017/2.* 36 p.

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EXECUTIVE SUMMARY

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Spatially Explicit Fisheries Risk Assessment (SEFRA) is a method that has been developed in New Zealand for determining whether current levels of fisheries bycatch impose unacceptable risks to seabird and mammal populations. It is designed to use sparse information and makes a clear distinction between the effects of uncertainty and management precaution. This document reports the conclusions of a review of SEFRA carried out in June 2017. The review process used draft documents describing the method and included two days of presentations from the main developers of SEFRA, given in an open meeting involving other interested parties. After deliberation, the Panel's draft findings were presented back to the open meeting for discussion; however, this report is solely the work of the four independent Panel members.

Our main conclusion is that SEFRA is a high quality method. It has been carefully thought out and implemented. We consider it to be a very useful tool, and hope it will become more widely known and used. We have, however, identified some areas that might benefit from further work.

The core of SEFRA is a detailed Bayesian model. The model is a good representation of the main features of the system but many of its prior distributions rely on sparse data or information elicited from experts. We feel that these should be re-examined to check their appropriateness and effects on the results. The treatment of overdispersion also needs to be standardised, and the assumptions around the linearity of effects and non-selectivity of bycatch considered.

All the models use a fixed value, of 0.2, for the coefficient of variability of abundances resulting from environmental variability. That parameter value is important to the conclusions of the models and requires further investigation. For seabirds the effects of unavailability while nesting, and the inclusion of populations that nest outside New Zealand need consideration. Some thought about the potential for interactions between species would be useful. We would also like to see further simulation and sensitivity testing, with particular emphasis on model misspecification and the characteristics of marine mammal populations.

The approach defines "Risk Ratio" as its main measure of how well bycaught species can be expected to do in the long term. At a simple level, low values are considered to be good, and high ones bad. The intuitive threshold for acceptability is 1, but some of the flexibility of the approach comes from the ability to choose other reference values. We feel that additional work is needed on the interpretation of differences from reference values (for example: how much worse, given a reference of 1, is 1.4 than 1.2?), and that alternative ways of communicating this information should be considered.

In general, we feel that the scientific component of SEFRA is well developed but may have got ahead of the wider management system it sits within. Even within the current implementations there are differences that suggest that communication between the developers has not always been perfect. It is important that both conservation and fisheries managers are included in future discussions to provide appropriate targets for species and ensure that the managers and other stakeholders understand the outputs and limitations of the method.

1 INTRODUCTION

Over the past several years New Zealand has adopted and developed a Spatially Explicit Fisheries Risk Assessment framework (SEFRA) to assess (and inform the management of) population-level risk to non-target species incidentally captured in commercial fisheries within a rigorous and transparent quantitative framework.

This review was commissioned to evaluate:

- the conceptual basis and mathematical formulation of the SEFRA method as it was originally conceived, with reference to particular implementations as illustrative examples only;
- two specific implementations of the SEFRA method, for New Zealand seabirds and New Zealand marine mammals;
- recent development of new tools to inform interrogation and evaluation of SEFRA outputs to inform risk management; and
- options for the application of the SEFRA framework to other New Zealand fisheries risk assessments, e.g. for non-target fish and benthic habitats.

The work presented for review is of high quality and has clearly been developed with considerable thought and attention to detail over a number of years. The Panel were presented with the SEFRA framework itself (MPI, 2016), and two applications of that framework (Abraham et al. 2017, Richard et al 2017) both of which involved a considerable amount of ancillary work, much of that done by Dragonfly Data Science, to estimate necessary inputs to the SEFRA (such as allometric modelling for seabirds, and elicitation of expert opinion for marine mammals). Where possible, outputs or components of the SEFRA implementations were compared with other, comparable, modelling work carried out in the past (such as the results from General Linear Mixed Models [GLMMs] to calculate observable seabird captures in New Zealand fisheries; Richard et al. 2017). The reviewers cannot claim to be in a position to provide detailed review of all components of the tremendous amount of work that has been done, but aim to provide feedback on the overall framework of the SEFRA and its application to marine mammals and seabirds.

We commend the developers of the SEFRA framework on a number of aspects of their work:

- The attempt to separate the component of the risk assessment that is the domain of science, from the component that is the purview of managers.
- The intention to clearly distinguish risk due to uncertain information from risk due to unsustainable catch.
- The clarity of the conceptual framework of the SEFRA, which is broken down into logical and understandable steps.
- The care with which the implementation has been thought through, and the comparisons with alternative approaches that have been applied outside of the SEFRA framework. Dragonfly Data Science is commended for their thoroughness and the quality of this work.

This approach to estimating total fishery bycatch for seabirds and marine mammals involves the modelling of observed bycatch rather than a scaling of observed bycatch based on the proportional coverage. It is well known that observed bycatch data is often of poor quality, seldom randomly sampled, often involves under-reporting, and is of a highly patchy nature in terms of spatial and temporal coverage (Bugoni et al. 2008, Phillips 2013). In addition, there is a major problem of unobserved (cryptic) mortalities in longline and, particularly, trawl fisheries (Brothers et al. 2010, Watkins et al. 2008). Particular difficulties can occur when observer sampling programmes are not appropriately designed or have insufficient coverage rates. Simply scaling observed catches per unit effort to total catch based on observer coverage is not to be recommended, and the authors of the SEFRA do not fall into that trap.

The method is consistent with existing fisheries risk assessment methodology (Hobday et al. 2011, Moore et al. 2013, Zhou et al. 2016). The concept of Potential Biological Removal (PBR) is used, to some degree, but with modifications in order to provide managers with more choices at the end of the process (rather than confining their involvement only to the simulation testing stage of the process that chooses the tuning parameter for the catch limit equation). This choice has been made deliberately, and with forethought. It carries the consequences that fisheries managers require knowledge of the whole SEFRA process in order to understand the outputs and convey the limitations of the method.

The primary output of the SEFRA is a Risk Ratio (RR), a measure that can be presented as a single number, or disaggregated by species, species group, fishery, fishing fleet, spatial area or even fishing event. A tuning parameter, ϕ, is chosen (by simulation testing) to meet pre-specified management criteria. In the seabird implementation, the reference standard 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 (MPI, 2016). This was interpreted as an environmental CV of 0.2, with additional demographic stochasticity. The simulations were based on a logistic biomass dynamics model that underlies the relationship between 0.5 r_{max} and K/2, the level at which maximum growth r_{max} occurs (Richard & Abraham 2013). Importantly, models that achieve r_{max} at higher levels than K/2 were included in the simulation work so that the results are robust to a range of maximum growth levels The simulation exercise also leads to the calculation of a correction factor for N for each species, which resulted from the assumption in the Population Sustainability Threshold (PST) analysis (but not the simulations) that adult survival applies to all ages (immatures and adults). The quality of this simulation work is critical in achieving Risk Ratios that, if kept below 1 (when all sources of mortality are considered) achieve the stated management reference criterion. The quality of this critically important simulation work appears to be high.

The SEFRA results can be disaggregated, down to the level of a species caught in a fishing operation at a point location, or to a grid of chosen resolution. The value of the tuning parameter, ϕ, is the same for all species (and the same across seabird and mammal applications) and therefore does not in any way affect the ranking of disaggregated Risk Ratio (RR) scores. If RR is disaggregated by fishery group, for example, then the relative contribution to overall RR by each fishing sector can be seen. These seem unlikely to relate linearly to the probability of meeting the management criteria, so should be interpreted with caution, and as a ranking system. Human minds tend to assume linearity, and coloured plots of RR by spatial block can be beguiling. Caution must be urged against the over-interpretation of disaggregated RR values. The primary use of the RR is only as a simulation tested tool for meeting management criteria (RR below 1) or not (RR above 1). Moreover, if the intention of the risk assessment is to identify priorities for research or management according to the rank of a particular species or fishery in a list (relative risk), not the absolute score, then the target reference point is less important. However, if the intention is, as currently stated, that the SEFRA "assigns risk to each species in an absolute sense", then the target reference does matter, because it determines the (supposed) absolute risk. However, as already stated, an absolute interpretation of this metric is problematic.

The development of the SEFRA framework has largely occurred within MPI (MPI 2016); however, its implementation to date has largely been contracted out (Abraham et al. 2017, Richard et al. 2017) while another implementation, as part of a presentation and disaggregation tool, is currently under development by another external contractor (D'Arcy Webber). There seems to be some divergence between these processes, and there is scope for inefficiency, because solutions to inherent problems are discovered independently (theoretical frameworks are always found to have unforeseen wrinkles during implementation). Definitions, in some cases, seem to differ between documents and presentations. It is urged that MPI, as the primary driver of this work, endeavour to maintain communication between all those working on the framework [RECOMMENDATION H1][1]. Careful, clear and internally consistent (error free) documentation of the process by someone who has actually implemented it (such as the work begun by D'Arcy Webber, see Appendix 5) is needed as a solution to this problem. A system for code-sharing between developers working on the method might also be useful.

[1] Recommendations in the text of this report are highlighted by upper case text in square brackets; e.g. [RECOMMENDATION H1], classified as high (H), medium (M) or low (L) priority and numbered. They are then compiled in section 4 of this report.

Ultimately, any modelling is limited by the quality of the input data. A great deal of development has gone into the formulation of the scientific component of the SEFRA approach and the method has reached a point where it would be useful to present it in a paper in a peer-reviewed scientific journal [RECOMMENDATION M1]. Focus could justifiably shift, now, to the collection of better input data and the management process that surrounds and utilises the science [RECOMMENDATION M2].

The reviewers' response to the Terms of References (Appendix 1) follows after the body of this report. The meeting agenda is included as Appendix 2, a participants list as Appendix 3, and a list of background documents as Appendix 4, along with biographies of the reviewers and their declarations of independence.

2 REVIEW BODY

2.1 SEFRA Conceptual Framework (TOR Q1)

2.1.1 Description

Spatially Explicit Fisheries Risk Assessment (SEFRA) is a method that has been developed in New Zealand with two aims:

1) to provide robust estimates of the direct (bycatch) impacts of fisheries on wildlife even when the available data are very limited, and
2) to clearly separate the scientific evidence base and analysis from the management decisions it informs.

One important driver for the development of the method seems to have been the conflation of uncertainty and risk in the PBR approach through the use of conservative abundance estimates in the setting of allowable takes (see Subsection 2.1.2 below). SEFRA is intended to instead provide the best available, unbiased, information, allowing managers to decide upon an appropriate level of precaution in its application.

The core of SEFRA is a Bayesian model. Use of the Bayesian statistical paradigm eases the combination of prior information and multiple sources of data, and the propagation of uncertainty. While broadly equivalent frequentist models could be constructed, in practice they are likely to be less complex than the SEFRA model. That is partly because in a frequentist approach it is more difficult to make use of the subjective information that informs many of the priors in a Bayesian model. While frequentist models can be interpreted as Bayesian models with flat priors, and *vice versa*, neither informative nor uninformative priors need be flat.

The SEFRA method has been implemented at least three times. We were impressed by the quality of all the work but concerned that some knowledge may have been lost during the process, and solutions reinvented. We also note the tension between theoretical purity and empirical knowledge in the differing implementations. The written description of the method in the AEBAR chapter (MPI, 2016) is very useful, but there is a need for clarity about which version it describes and the difference from previous implementations.

[RECOMMENDATION M3]: identification of how the methods used in the AEBAR chapter differ from the previous implementation and recording of the reasons for the differing decisions.

There has been an overall trend towards integration, and the inclusion of increasing amounts of associated data into a single fitting process. While this can improve consistency and the propagation of uncertainty, it does increase the complexity of the computations and reduce flexibility in the completion, maintenance, modification and development of the system.

[RECOMMENDATION M4]: identification of points at which the calculations within the SEFRA can be split into modules that are largely independent and, ideally, have efficient and meaningful inputs and outputs.

[RECOMMENDATION L1]: there should be some contingency planning for the possibility of non-completion of current implementation or unavailability of the current developer.

One major advantage of the new integrated implementation of SEFRA is its potential to allow managers to carry out scenario testing of changes to the model inputs or implementation that they might consider necessary, or potential changes in fishing patterns and practices.

While we recognise that the method is designed for data-poor situations, we are also slightly concerned that there seems to be no lower bound on the amount of data that is considered sufficient to provide meaningful results. When information is very scarce, not all sources of uncertainty may be adequately represented. We briefly discuss options for simulation testing of such limits (in Subsection 2.6.3 below), and also stress the importance of explicit acknowledgment of the existence of such limits.

[RECOMMENDATION H2]: results of calculations that depend on very sparse or indirect information be highlighted and treated with particular caution.

In the subsections below we touch on some key aspects of the design.

2.1.1.1 Population Sustainability Threshold (PST)

All the documentation uses the same equation:

$$PST = \tfrac{1}{2}\, \varphi\, r_{max}\, N$$

but there are differences in its interpretation. Webber (Appendix 5) simply mentions meeting of “the long-term goal”, usually of remaining above half carrying capacity; the AEBAR chapter interprets it as “analogous to PBR”, such that “impacts equal to PST (i.e., R = 1) correspond to a defined level of population stability or recovery”; whereas the current seabird implementation states:

> *PST is an index of the population productivity, adapted from the PBR. It is an estimate of the maximum number of human-caused mortalities that will allow populations to remain above half their carrying capacity after 200 years, with a 95% probability, when the number of annual potential fatalities equals the PST and when considering uncertainty and environmental stochasticity.*

Very similar wording to this is used in the description of the marine mammal implementation.

Two hundred years is long enough to be beyond simple extrapolation from current trends, but will not necessarily allow all these populations to return to close to equilibrium sizes. For faster growing populations, 200 years is also long enough that required annual growth rates need not exceed 1% even for very depleted populations, e.g., a population reduced to 11% of its carrying capacity could return to 50% in 200 years of logistic growth with a maximum annual growth rate around 1% p.a., but would be below 20% of carrying capacity for the first 60 years. Whether that is acceptable is not a scientific issue; however, the associated risks are, and it is important that such implications are not lost in the methodological details.

2.1.1.2 Variability of population size

The incorporation of the inter-annual variability of population size in the definition of PST requires the use of a value for the coefficient of variation. Necessarily, this will be hard to estimate for data-poor, depleted populations. The analyses supporting the use of a value of 0.2 for bird populations is very useful

(Richard et al. 2017), but whether that value is also appropriate for mammals, other taxa, and very small population is less clear.

[RECOMMENDATION M5]: that the effects and appropriate values of the CV used for inter-annual population variability be investigated, particularly for marine mammals.

2.1.1.3 Phi

The parameter φ is a key feature of the difference between the SEFRA method and PBR. When and by whom it is set is important to the separation of scientific information from management. This issue is discussed in Subsection 2.1.2 on PBR below.

2.1.1.4 Risk Ratio

The use of risk ratio is another distinctive feature of SEFRA, so its interpretation and interpretability will be very important to the use of the method. This is discussed in Subsection 2.5 below.

2.1.2 Risk assessment methods including fisheries stock assessment, SAFE, and PSA

Fisheries risk assessment methods can be grouped into four categories: qualitative, semi-quantitative, quantitative, and statistical models. Amongst these categories, semi-quantitative and quantitative methods are widely adopted in risk assessment of non-target species (Hobday et al. 2011). The Productivity and Susceptibility Assessment (PSA) is a typical example of the semi-qualitative methods, while the Potential Biological Removal (PBR) and the Sustainability Assessment for Fishing Effect (SAFE; Zhou & Griffiths 2008) are examples of quantitative methods (Zhou et al. 2011). Statistical models typically attempt to estimate both fishing impact and reference points simultaneously. A classic example is the quantitative fishery stock assessment.

These alternative approaches are all based on the same basic concept: to estimate fishing impact and the species' intrinsic capability to withstand such an impact, and then to derive a risk measure by comparing the two components. Hence, although different methods use different terms, they generally mean the same thing (e.g., susceptibility, fishing mortality, and fishery-related death all measure fishing impact; F_{msm}, MSY, PBR, and PST are all measures of productivity or production).

SEFRA adopts the basic conceptual framework of fishery risk assessment and falls into the quantitative category. It is well in line with international practice. SEFRA uses spatial overlap to estimate total fishing mortality, which is similar to the SAFE approach; it uses r_{max} to derive PST, which is similar to PBR and SAFE. As PBR and SAFE have been published and adopted in fisheries management, the framework of SEFRA is conceptually defensible.

2.1.2.1 Differences from other risk assessment methods

2.1.2.1.1 Estimating fishing mortality

Unlike most fish bycatch, SEFRA assumes that the observed catch and total abundance are available for seabirds and marine mammals. SEFRA has the advantage that it can make use of these data (which are seldom available for non-target fish bycatch).

SEFRA links the observed catch to abundance at each fishing site by a catchability parameter (a combination of vulnerability and cryptic interaction terms). The abundance at each fishing site is calculated from the known total abundance and density distribution. The catchability parameter is essentially a ratio estimator (observed catch divided by local abundance). In fishery observer programmes, the traditional approach of estimating total bycatch is to expand the observed catch by the observer coverage rate where fishing effort is the dominant variable. A question here is which variable, abundance or fishing effort, is a better choice to be used for this ratio estimator. To obtain an unbiased estimate, the sampling design requires that observers should be assigned randomly to fishing events. On the other hand, the accuracy of the estimated local abundance at fishing sites and the total abundance will have a direct impact on the estimation of the total catch.

An alternative is to use general linear, general additive or other non-Bayesian model-based approaches for estimating total catch. In these a model is built by linking observed catch to known covariates, which in the seabird literature may include latitude, longitude, depth, temperature, time of day, moon phase, fishing gear type, etc. (e.g. Jiménez et al. 2014). Prediction of total catch can be made from such a model to all fishing events that have these covariates (but not if they are unavailable, which limits the application of this approach to many fisheries). Although analyses of this type are reported in the current seabird implementation (Richard et al. 2017), it would be useful to explore whether density could be included as another covariate to improve the model performance, and determine the extent to which density influences bycatch rate and the shape of the relationship in different fisheries [RECOMMENDATION L2].

In risk assessment of fish species, overlap is the ratio between area-fished and species distribution area, or relative density (if known), where area-fished (= gear-affected area) is a linear function of fishing effort. For fishing methods that use baits to attract fish (e.g. trap and longline), gear affected area is a function of bait odour dispersion range and fish swimming speed toward the bait. Applying a similar idea to seabirds, the overlap would consider gear operation duration and species' attraction distance (which may vary between species) as well as seabird density. It would be useful to check whether, when seabirds are attracted to a point (the fishing boat) within their detection distance, the linear relationship between overlap (impact) and fishing effort (e.g. number of hooks or hauls, kilometers of net) still holds [RECOMMENDATION M6].

2.1.2.1.2 Reference points

SEFRA modifies PBR and suggests PST as a primary reference point for quantifying fishing risk for marine mammals and seabirds. The population outcome associated with the definition of PST chosen in these two risk assessment implementations is to "recover or stabilise to an equilibrium population at or above 50% of K, with 95% probability, including effects of environmental stochasticity". The formulation is similar to MSY for target fish species and MSM (maximum sustainable mortality) for non-target fish bycatch. There are three potential issues: (1) It assumes that the logistic model is applicable to seabirds and marine mammals. Is there sufficient evidence that this model is a valid assumption for these taxa? (2) There are limited options for taking the time dimension into consideration, which depends on the productivity of the species and the current population status relative to the status prior to the onset of fishing. This difficulty should be recognised. (3) How does PST include environmental stochasticity? Environmental variability can affect r_{max} and K, which is difficult to incorporate into PST.

We recommend further work be carried out that:

(1) Considers whether observed catch can be expanded by total fishing effort to derive total catch, or expand existing model-based approaches by incorporating density, in addition to other covariates, as a predictor [RECOMMENDATION L2].

(2) Examines whether the population dynamics of seabirds and mammals can be described by the classic logistic model [RECOMMENDATION L3].

(3) Examines the method for deriving overlap and the assumption of a linear relationship between bycatch and fishing effort [RECOMMENDATION M6].

(4) Considers ways similar to those used in SAFE (Zhou & Griffiths 2008, Zhou et al. 2011) to display and communicate the outputs of risk assessment [RECOMMENDATION H3].

2.1.2.2 Fisheries stock assessment

Fisheries stock assessment is the most rigorous risk assessment in fisheries. Stock assessment also involves estimating fishing impact and the corresponding reference points. Stock assessment models require a relatively large amount of available information about the stock structure and life history of the species under examination as well as data from the fishery that captures it (e.g. Quinn & Deriso 1999). Model complexity varies from simple Surplus Production models that lump the individuals of a population into a single biomass value, with no age structure, to statistical catch at age models that consider separate age, and perhaps also length, classes and incorporate individual somatic growth rates. These models estimate the productivity of populations, as well as their current size relative to carrying capacity.

2.1.2.3 PBR

The starting point for PST is the concept of Potential Biological Removals, PBR (Wade 1998). Like PST, PBR is just a reference point—one of two components in fisheries risk assessment (the other component is estimating fishing impact). PBR is sometimes considered to be just the formula itself (PBR = N_{min} 0.5 R_{max} F_R) with appropriate values chosen for the components. It is essentially the MSY concept when F_R = 1 and $M_{min} = M_{mean}$. Alternatively, PBR could be viewed as a framework (arguably an MSE framework) in which a catch limit equation is used in the context of considerable simulation work that demonstrates which values of F_R (and default values for R_{max}), and the definition of N_{min}, together with an 8 year survey cycle, will meet pre-specified management goals. Such goals might be that population size not drop below 0.5K in 95% of cases, under a range of scenarios that specify uncertainty, and plausible bias in parameter values and for a range of (actually two) shapes for the curve that gives productivity as a function of population size relative to carrying capacity. In addition, PBR allows for more rapid recovery rates for very depleted populations by specifying the use of a lower value of F_R. The likelihood that a population with a take limited by a value set using the PBR equation will indeed meet the management objective inherent in the method is only as good as the match between that the reality for that population and the simulation tests performed by Wade (1998). Any application of PBR should consider this match carefully and, if necessary, new simulation tests should be performed specifically for that application.

PST is not PBR. The PBR definition of N_{min} (which was selected using simulation testing) was deliberately changed in the SEFRA to N, an estimate of average current population size (in the most recent 3 years). This change was intended to shift the point at which management goals are implemented away from the simulation testing step to a later step in the process. However, simulation testing was used (Richard & Abraham 2013) to select a value for a tuning parameter akin to the F_R used in the PBR calculation, although here that parameter also assumes the role taken in PBR by a definition of N that was deliberately on the low (precautionary) side. PST is designed in SEFRA to be used in the calculation of Risk Ratio, RR, where an estimate of the total numbers killed is divided by an estimate of PST. In the presentations at the meeting, the importance of using the same definitions in both numerator and denominator was stressed. However, it could be argued that formulations in which N_{min} was used in the numerator and N in the denominator are also valid because ultimately the denominator is the maximum number of animals that are permitted (in some sense) to be killed and therefore is comparable with the numerator no matter how that value was determined.

The value chosen for the tuning parameter (φ in PST terminology) is crucially important in ensuring that PST can be expected to meet pre-specified management goals. This value must be based on simulation tests that span the range of scenarios, including likely model misspecification and bias in parameter estimates that pertain to any particular application of the SEFRA. Such simulation testing was done for the seabird application (although further expansion to model misspecification and possible bias scenarios might be useful [RECOMMENDATION M7] and should either be repeated for the marine mammal application, or a stronger argument be made that the simulation work for seabirds is equally applicable [RECOMMENDATION M7].

Note that by using a definition of N_{min} that is the 20^{th} percentile of available estimates, PBR incorporates an inbuilt incentive to reduce uncertainty in estimated population size. The SEFRA framework can accomplish the same goal by considering the full range of probable values for RR. It is therefore recommended that statistics such as the probability that RR is below 1, be used in framing advice to managers, rather than any single value (e.g. mean or median) of RR [RECOMMENDATION H3].

In developing the PBR, Wade and collaborators deliberately avoiding using trend in population size because, given the typically large CV in population estimates, small populations could be extinct before declines could be detected with statistical significance. However, breeding populations of many (but not all) seabird populations can be counted with high accuracy, so it is recommended that trends in any available time series be considered. Such consideration could range from examination of a time series plot when making decisions about management actions on the basis of the SEFRA, to developing a set of (simulation tested) rules regarding how to incorporate available trend information into management action [RECOMMENDATION L4]. Trends could also be used, in comparison with RR, to detect whether additional sources of mortality (other than fishing within the New Zealand EEZ) are likely to be important. A declining trend in association with a low RR would suggest such a situation.

It is important to note that a criterion that populations be maintained above a specified level, with 95% certainty is likely to result in at least 3 out of the 71 populations to which it was applied, not meeting the management criteria. Use of PBR involves setting a much lower take (bycatch) limit for populations that are thought to be at very low levels (e.g. Endangered or Critical according to IUCN Red List criteria) to allow for faster recovery. This is so-called biomass-based reference point in fisheries when time series of abundance data are available. For populations (perhaps such as Maui dolphin) that are at such low levels that they are at risk of extinction, any bycatch mortality above zero might not be appropriate. It is recommended that SEFRA introduce specific rules or clear guidance for managers that recognise that such populations require special treatment [RECOMMENDATION H2].

2.1.3 Management Strategy Evaluation

Management Strategy Evaluation (MSE) is now commonly used in fisheries management (Punt et al. 2016) and has evolved from the development of the Revised Management Procedure (RMP) by the International Whaling Commission (IWC). The essence of the approach is the development of a rule (or set of rules), that takes available data (and any future data collections of the specified form) and translates these into management action. The rule(s) is thoroughly simulation-tested in a range of possible real-world scenarios that include all foreseen combinations of variability, uncertainty, incorrect model specification, and bias, and which must meet pre-specified goals chosen by managers, with a probability that they also decide. Once the simulation testing is complete, managers lock in those rules for a specified period of time (that would also have been included in the simulation tests) and undertake to slavishly enact the actions prescribed by the rule(s) until the end of the pre-specified period, when the MSE is to be repeated in the light of any new information or change of policy. Some MSEs include a ‘special circumstances’ rule that allows management to abandon the rule(s) if certain pre-specified situations arise that are outside of the scenarios that were tested (such as a population estimate that is outside a particular range). PBR includes most of the elements of MSE; however, it differs in that PBR lacks rules stating exactly how managers should reduce mortality if the PBR level is exceeded.

SEFRA avoids directly locking managers in to management action based on the PST results and the RR. It attempts to clearly distinguish value judgements from the scientific information they are based on.

2.1.4 Soundness of SEFRA conceptual framework

Along with other methods that consider productivity and population growth rates, and aim for a proportion of carrying capacities, management though SEFRA will be vulnerable to changes in environmental and other conditions. Carrying capacity may be reduced by loss of habitat or regime shifts. In principle this could lead to extinction of a population or species without either being considered (by the method) to be in unfavourable conditions.

[RECOMMENDATION H2]: The targets and limitations of the approach should be made explicit to managers and stakeholders.

The method does not consider the New Zealand Threat Classification System (NZTCS) or international (IUCN) Red List category, nor the previous population history of each species (e.g. whether the population has been in steep decline and currently is severely depleted). This is reasonable within the current modelling framework, but these factors, along with threats from other fisheries outside of New Zealand waters, or on land, must be taken into consideration when determining the management response to the outputs of the SEFRA [RECOMMENDATION L5].

Unlike the marine mammal SEFRA, which includes migrants, the seabird SEFRA is currently restricted only to populations of seabirds that breed within New Zealand. However, New Zealand has an obligation under international treaties such as ACAP (Agreement on the Conservation of Albatrosses and Petrels; www.acap.aq) to also understand and mitigate threats to albatrosses and petrels listed under this agreement that use New Zealand waters, but do not necessarily breed in the region. For this reason, the seabird SEFRA should be extended as far as practical (given data limitations) to other species listed by ACAP (which are all considered to be at potentially high risk from fisheries) that use New Zealand waters [RECOMMENDATION L5]. This may not add many species, but would include wandering albatrosses from breeding populations in the Atlantic and Indian oceans, and black-browed albatrosses from Islas Diego Ramirez, for example, which migrate to the New Zealand region ACAP 2009, ACAP 2010).

2.2 The selection and preparation of input data (TOR Q 4)

2.2.1 Input data in the Seabird Risk Assessment

The methods used to generate demographic parameters and explore issues related to PBR for seabirds (Richard & Abraham 2013, Richard et al. 2017) rely on a number of assumptions. Allometric relationships are used, which are based on measured values for species or populations that may be biased low due to emigration, biennial and deferred breeding, and other factors that affect recapture rates of banded individuals. The simulations could usefully have included an annual-breeding albatross (the two albatrosses that were included are both biennial breeders, and the giant petrel has been included with a mean age of first breeding that is several years younger than that expected for an annual-breeding albatross). It is not always clear within the framework when survival rate and age at first breeding are observed (used as *current* in the SEFRA), substituted from other species, or optimal (based on allometric relationships), nor the extent to which the observed data has been filtered according to quality before inclusion. Furthermore, the terminology needs to be clearer, as it seems that the allometric analyses were based on observed survival rates (which include natural and anthropogenic mortality; note that the latter would not be allometric), but the derived/estimated survival is then used as if it were optimal.

It would therefore be useful to quality-control the input data in the allometric analyses, and the current survival estimates used in the SEFRA, excluding those from populations likely to have been impacted by fisheries, or which appear to be anomalously low for methodological reasons [RECOMMENDATION L6].

The next iteration of the allometric analysis should also include mean survival rate and age at first breeding from outside the New Zealand area, particularly for bird families that are poorly represented within the current analysis [RECOMMENDATION L6].

The use of phylogenetically independent contrasts may be a better way to account for the apparently contrasting allometric relationships with survival that are apparent between seabird families or orders [RECOMMENDATION L7].

2.2.2 In the Marine Mammal Risk Assessment

2.2.2.1 Expert elicitation

None of the Panel members are experienced in the elicitation of knowledge through Delphi processes. However, as this approach provides many of the life-history parameters and a large part of the distributional information used for the less-well studied marine mammals, we will make some brief, general, comments on the issue.

The questions asked of the marine mammal experts seem particularly difficult. Few of the experts are likely to have had previous experience of being asked to provide bounds on 95% CIs. These are more usually associated with the presentation of data in published papers or models. Extrapolating from published CIs to minimal, and indirect, data requires consideration both of how uncertainties scale with sample size and the similarity between studied and unstudied situations. Estimates of patterns over space and time will often require extrapolation from occasions where individuals were seen in areas that have not been visited or where search effort was much lower, and coping with the bias that an observation tends to be more memorable than an absence (i.e., most people are much better at recalling having seen a species than the often more numerous occasions when an area was visited but that species was not seen). Estimation of r_{max} will depend on assessment of the similarities of different populations and species, and balancing of all the different ways in which the target species might differ from potential comparators, without much solid data on which to base the comparison.

The attempted elicitation of spatial information is a particularly interesting exercise, but it is unsurprising that it appeared to meet with limited success. The best way to ask such questions is not obvious, though one possibility might be to ask about the plausibility of a series of candidate maps rather than directly request that a new one be drawn.

The limited numbers of experts asked, and that responded, to the questions is grounds for concern. Low response rates raise questions about the wider representativeness of surveys. In this case there is a risk that only those most confident in their knowledge will have replied. There is some evidence that the amount of confidence experts have in the accuracy of their estimates increases faster than that accuracy (McBride et al. 2012).

It might be possible to increase the amount of information obtained from each expert by following the request for 95% CI, with a further request for the mean (or median) values; however, the small number of responses, especially to the second round of the process, does not suggest there was much enthusiasm, or confidence in the utility of the exercise, among the subject experts.

The cautious use of the elicited values by effectively combining them into a mixture distribution seems sensible, especially given that the provision of information to the experts may have compromised their independence. It may therefore be worth considering using Wade's default r_{max} values as the basis for an additional, virtual, expert, although the distribution that should be allocated is not immediately obvious. It would be useful to predefine both what would be considered an adequate pattern of responses that provides useable information, and the next course of action when that is not available. In addition, the effect of the inclusion of very large values in the default diffuse priors for the species where no experts estimated abundance may need further consideration.

Wherever possible, it is preferable to derive values through a documented process based on available data. The allometric modelling that was done for seabirds (Richard et al. 2017) is a good example of such a process. Rather than asking experts for a total abundance in New Zealand, abundance could be derived by deciding what proportion of a global estimate of abundance for a species is thought to occur in New Zealand.

[RECOMMENDATION M8]: Elicited priors should be avoided as much as possible.

[RECOMMENDATION M8]: If elicited priors are to be used, they should be diluted with a less informative distribution to reduce the risk of underestimating the uncertainty.

[RECOMMENDATION M8]: If expert elicitation is to remain an important part of the SEFRA process, then suitable advice should be sought and some validation work would be required.

2.2.2.2 *Uncertainty in spatial distributions*

Uncertainty in population sizes is much easier to capture and represent than uncertainty in locations. Methods, such as generalised additive modelling, which are based on splines, share the problem of more direct approaches of adding uncertainty at each location, in that this uncertainty acts in the wrong dimension i.e., changing densities at points rather than the locations of high density areas. Simply rescaling the surface to preserve the total volume will introduce distortions across the wider area. Kriging is a geospatial technique that might be less vulnerable to the problem. However, the best solution might be to generate the surface as a knot-based spline and to randomly displace the knot locations in the horizontal plane. Alternatively, a more straightforward approach might be to generate a suite of approximate density maps, by random resampling of location data or otherwise.

[RECOMMENDATION M2]: Given the limited precision of the available data on distributions of most species, exploration of the uncertainty may be less of a priority than filling the gaps in the input data.

2.3 Recent implementations of the framework; (TOR Q2, Q5 & Q6)

2.3.1 Model structure

The current categorisation of groups is quite rigid, with species and fisheries being considered either entirely separately or as being identical to all other members of their group. An alternative or additional approach that may be worth consideration would be to define values for some species or fisheries as being intermediate between the characteristics of two others.

2.3.2 Priors

Uniform priors are now not considered uninformative, as they tend to pull posteriors away from the ends of their range. Beta(½, ½) priors may be more appropriate where very little is known about proportions. Unfortunately, they will generally give rise to wider confidence intervals than uniform priors.

The conservatism that SEFRA is intended to avoid, can creep in through the specification of priors. Two examples in the Marine Mammal Risk Assessment (Abraham et al. 2017) are the prior on the observability of bycatch which (p19) is given as U(0.5,1) based on “one of two Hectors dolphins in a net not being seen by crew”. It seems surprising that that is not symmetric about 0.5. Similarly the Beta(3,1) for the prior on the probability of live release is based on only four animals and appears to be quite informative, but could be made less so while maintaining the same mean (Figure 1).

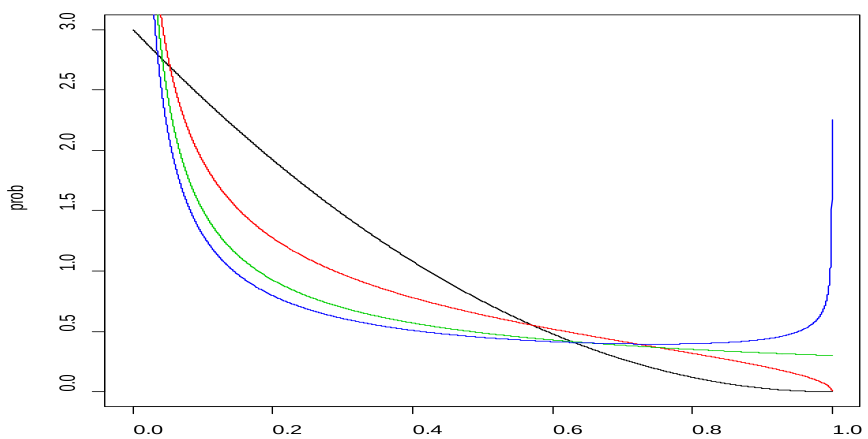

Figure 1: Prior on probability of live release of animals (red); the other lines are distributions with the same mean but higher variances.

[RECOMMENDATION H4]: All the priors intended to be uninformative should be re-examined, and the effect of increasing their variances should be investigated.

2.3.3 Over-dispersion

Over-dispersion is a serious problem when modelling bycatch data. This has been recognised and accounted for in modelling work done by Dragonfly to estimate captures, e.g., the hurdle model used for dolphins where the probability of an encounter between a fishing vessel and a species is modelled separately from the number of animals caught if an encounter occurs (Berkenbusch & Abraham 2017), and the choice of statistical distribution used in the regression models of seabird bycatch (Richard & Abraham 2013, Richard et al. 2017). More consistency should be applied to dealing with over-dispersion in the SEFRA framework, with consistent and documented choices being made between negative binomial and quasi-Poisson distributions [RECOMMENDATION M9].

2.3.4 Spatially-explicit density estimation and overlap calculations

The seabird implementation of the SEFRA uses a continuous surface to describe the density distribution of seabirds. Fishing shots are points on this surface. Catch is a function of density at the location of the fishing shot, multiplied by vulnerability parameters whose value can exceed 1. The interpretation of values greater than 1 would be that seabirds are attracted from the surrounding area (which they have to be, as the fishing location is a point estimate). Only the density at the point where fishing occurs is taken into account, not the slope of the density map around that point, from which birds are being attracted. It would be useful to set a reasonable radius of attraction for each species group (similar to estimating gear-affected area for fish species, Zhou et al. 2014), and to integrate the density distribution around the point estimate given that attraction radius (perhaps using a bivariate normal distribution, or at least a uniform circle) [RECOMMENDATION L2]. This would have the advantage that vulnerability values greater than 1 would be unrealistic, and could flag problems (such as a poor density distribution, too low an abundance, or a radius of attraction that is too small). Note that there are several recent tracking papers which report the distances at which individual birds respond to the presence of a fishing vessel (Collet et al. 2015, 2017).

For the Seabird Risk Assessment, the vulnerability value for one seabird group (white-chinned petrels) and deepwater trawl fisheries, large bottom-longline fisheries, large surface-longline fisheries and set nets had to be set equal to 1 to allow estimation of all vulnerabilities. The units of vulnerability must therefore also reflect these choices in that they are expressed relative to these groups. This model estimated vulnerabilities not only for the species groups and fishing fleets, but also for their interaction. Therefore, for identifiability, it would have been unavoidable to set, rather than estimate, some parameters. Note that if no interactions are estimated, then the vulnerability parameters for a SEFRA implementation form a matrix of number of species groups (n_s) by number of fishing fleets (n_g). Observations of captures provide information on those $n_s \times n_g$ parameters. If observations are available for all cells in this matrix, then there are as many equations as there are unknowns. If any cells lack observations, then there will be more unknowns than equations and some vulnerability values will have to be assigned values. For this reason implementations of SEFRA should err on the side of lumping rather than splitting species and fishery groups, and tables reporting the number of observations available in each species group by fishery group cell of the matrix should be presented [RECOMMENDATION L8].

The availability of birds across the seabird distributions should take account of breeding birds (about 50% of the total) that are on land during the incubation and brood-guard periods, i.e., the overlap calculations for those periods should include a factor of 0.5 [RECOMMENDATION H5].

2.3.5 Demographic parameters

There is an assumption in the SEFRA model that all birds killed are adults. However, it may be difficult to distinguish older immatures/pre-breeders (which may also have well-developed gonads) from adults using necroscopy. In addition, there may be a degree of spatial segregation between immatures (pre-breeders) and breeders, which can affect bycatch rates (Gianuca et al. 2017), and could reduce the representation of immatures among the birds returned for necropsy if there is unbalanced sampling across fisheries and regions. For this reason, it would seem reasonable to allow that some of the estimated bycatch includes immatures [RECOMMENDATION H5]. Also, note that “juvenile” is usually used to describe a bird still in its juvenile plumage, and so the term “immature” or “pre-breeder” is preferable to describe individuals that have never bred.

For the Seabird Risk Assessment, available counts of breeding pairs were converted to total population size (of birds aged 1 and older, 1+) using an estimate of current adult survival rate (S_{curr}) and the assumption that the survival rate of juveniles was equal to that of adults (animals in their first year of life, i.e., the 12 months after egg-laying is not directly calculated but is effectively assumed to be whatever value ensures that the population remains in stable equilibrium). The same assumption about juvenile vs adult survival rate is made when S_{curr} is used as an upper limit for total mortality. If, as is likely, the mean mortality rate of younger birds is higher than that calculated for breeding birds, then total (age 1+) population size will be underestimated. This will in turn lead to overestimation of vulnerability parameter values and underestimation of PST. As the vulnerability parameters are scaling factors, and may be expressed relative to fixed values for some groups, this can largely be expected to ‘come out in the wash’ except that the contrast between adult and juvenile survival rates will be greater for some species than others so that the grouping of these species could to some extent negate this effect; however, the problem is likely to be relatively small. The estimation of S_{curr} is challenging enough without also adding the need to estimate current juvenile survival. Underestimation of PST is also not a concern because the denominator of the risk ratio equation would be equally underestimated (N appears in both the numerator and denominator of this equation and cancels out).

In the allometric modelling for seabirds, the slope but not intercept parameters were allowed to vary by taxonomic grouping. There is some indication that the slopes for some taxonomic groups, in particular the gulls, differ which seems to justify freeing the slope parameter [see RECOMMENDATION L7 above].

A prior should not be placed on the survival value that is used to constrain the number of birds killed (i.e., total deaths $\leq (1\text{-}S_{curr})$) [RECOMMENDATION H6].

Neither should an estimate of natural mortality (or indeed any other additional sources of mortality) be used to tighten this constraint, unless investigators are very sure of the accuracy of their estimates of natural mortality (or additional mortality), S_{curr} and of the assumption that juvenile survival is equal to S_{curr} [RECOMMENDATION H6].

Ideally, the constraint value should be at least equal to the number of deaths, or greater (i.e., investigators should err on the side of making the constraint value too large, never too small, because constraints that are too tight may rule out the correct answer). If a suitable distribution of values is available for the constraint (such that it could be used for a prior) then instead of using the distribution itself, a single value from the high end of the distribution should be used for all iterations of the model [RECOMMENDATION H6].

Constraints should be limiting, but should not be informative. The model fit for any species for which this constraint strongly informs the posterior should be regarded with suspicion [RECOMMENDATION H6]. This is clearly illustrated for black petrels for which high bycatch by one vessel in a poorly observed fishery seems to have inflated the vulnerability parameter value to a point where a relatively large number of model runs estimate total deaths that are unrealistically large i.e., close to or above the mortality constraint.

2.3.6 Impact scalars (vulnerability, catchability, susceptibility)

Although it seems reasonable to assume that relative species vulnerabilities are similar across similar fisheries within New Zealand waters, as in the current SEFRA, the same assumption cannot be made in other regions (see below). There appears to be an inherent assumption in the SEFRA model that any relationship that exists between density overlap and the number of captures/deaths will be linear; however, the relationship may be asymptotic if a limited number of individual birds can be present in the area behind vessels in which baited hooks are accessible in longline fisheries, or around trawl warps etc. In addition, over shelf waters, thousands of birds may feed behind demersal trawl vessels which are generating large amounts of discards, whereas only a few hundred birds may be attracted by surface longline vessels which produce fewer discards. Unless limited by some mechanism other than current mortality rate (which only affects the total number of birds that can be killed), the assumption of a linear relationship could lead to considerable overestimation of numbers of birds killed at a local scale in high density areas. This importance of this issue should be explored [RECOMMENDATION M6].

SEFRA currently assumes that bycatch affects both sexes equally. However, this is often not the case for seabirds (Gianuca et al. 2017). It is recommended that, where possible, the sex ratio of captures be reported for consideration alongside risk ratio. If a highly skewed sex ratio is observed, then it is necessary to take into consideration the number of animals of the sex that is captured most frequently when quantifying impacts at the population-level [RECOMMENDATION H5].

It is difficult to compare vulnerabilities to different fisheries in the current SEFRA implementations. Consideration should be given to the possibility of rescaling the parameters to simplify these comparisons [RECOMMENDATION M10].

2.4 Application of SEFRA framework to other fisheries risk assessments (TOR Q7)

The SEFRA framework; i.e., estimating fishing impact based on overlap, deriving a reference point based on life-history traits, and quantifying the risk by evaluating the two components, is essentially the same as other ecological risk assessment (ERA) methods that have been applied to benthic habitats and non-target fish species, and considered for target fish stocks for which little information is available. However, the detailed method may not be transferable to other taxa simply because the required data are not available. For example, for fish bycatch and benthic invertebrates, observed catch, natural mortality and current mortality are not available and there is no estimate of total abundance for these taxonomic groups. Alternative techniques must be developed to quantify fishing impact. This has been done in other regions, but the methods can be adapted and improved to suit the situation in New Zealand waters. Indeed, the presentations to the Panel on the ERAs in development for other taxa and habitats in New Zealand demonstrated that impressive progress had been made.

[RECOMMENDATION L5]: continue improving quantitative risk assessment methods for other taxonomic groups to accommodate the unique information available in different situations.

2.5 Diagnostics, display and communication of outputs (TOR Q3, Q8)

While it is important to diagnose and assess the goodness of fit of the model, it is necessary to be realistic about the limitations of the input data. Where the data are very limited, tests will have little power; where they are abundant, testing may highlight trivial errors.

It is important to examine the correlation matrix for estimated parameters as an indication of how much power the data have to discriminate between parameters. If two parameters are highly correlated (e.g. > 0.8) then the data cannot support both in the model and they should be merged or one of their values fixed and sensitivity tests performed to alternative values [RECOMMENDATION H7].

One potential approach, that seems to be being considered within the new integrated implementation, is the use of residual plots to examine distribution maps. If only observed hauls, with and without relevant bycatch are considered, then the difficult problem of distributing zeros, which affects presence-only data such as that from animal tracking, is avoided. In such situations testing for pattern with logistic generalised additive models might be appropriate. However, there seems to be very limited data that can be used to test the fit of the modelled spatial patterns.

If it is considered necessary to carry out that sort of investigation, it would seem sensible to use simple "back-of-the-envelope" calculations based on the scaling of standard errors with sample size. However, the least problematic of such methods might be to use upper bounds on estimated effect sizes to determine minimum sample sizes that could have a chance of detecting an effect, and to use these to avoid committing resources to carrying out data collection that lacks the potential to provide useful results.

[RECOMMENDATION M11]: detailed diagnosis of goodness of fit seems unlikely to be very informative in spatial models based on sparse data.

We recommend that, for clarity, the units of all parameters and variables should be reported (this includes the vulnerability parameters) [RECOMMENDATION H8]. Furthermore, in any documentation of the SEFRA, in both wording and mathematical notation, the distinction between S_{curr}, the current survival rate, and S_{opt}, and the survival rate that relates to maximum population growth (r_{max}) should always be clear [RECOMMENDATION H9].

The choice to use effort units of shots for longline fishing in the seabird implementation but hooks in the marine mammal implementation is inconsistent. Internationally, values for seabird captures per unit effort are almost always provided as birds per 1000 hooks, which enables easy comparisons between studies. There is debate as to whether the use of hours trawled (to better reflect warp strikes) or trawl shots (to better reflect capture during hauling and setting) is to be preferred. A sensitivity test that compares alternative effort formulations would be desirable [RECOMMENDATION H10].

The data inputs for the marine mammal implementation that derive from the Delphi expert opinion process could be seriously in error, including credibility intervals that are much too narrow. It is important that the influence of these 'data' on the results be highlighted [RECOMMENDATION M8]. One way to do this could be to present results where these inputs are replaced by very broad priors.

The PST attempts to step away from the PBR method of binding the process of meeting management goals into the specification of the PBR limit, via the selection of the conservative definition of N_{min} and the choice of F_R. However, the choice of ϕ is made at this step. Thus the SEFRA presents managers with a Risk Ratio based on the probability of meeting an abundance level. The uncertainty in the Risk Ratio is also provided. However, because it is not immediately obvious how Risk Ratios should be interpreted (beyond comparing their values to 1) it is even less clear how uncertain values ought to be interpreted. For example, which is better: RiskRatio1 = 1.1 (95% CI: 0.9,1.2) or RiskRatio2 = 1.1 (95% CI: 0.8,1.3)?

Risk results from the combination of estimated fishing impact and the specific reference point adopted. Both components are biologically meaningful, may vary between species and fisheries, and both involve uncertainty. Using a single risk ratio and presenting it as shown in Figure 3.2 of Ministry for Primary Industries (2016) annual review report appears to be clear and neat. However, this presentation has four major drawbacks: (1) it does not reveal the resilience/productivity of each species and how it compares with other taxa; (2) it does not reveal the magnitude of the fishing impact; (3) it does not reveal which components contribute to the uncertainty; and (4) it does not suggest on which component attention should be focused to provide maximum efficiency in reducing uncertainty. We suggest that Figure 2, as used in SAFE reporting (Zhou et al. 2011), be considered in future SEFRA reporting [RECOMMENDATION H3].

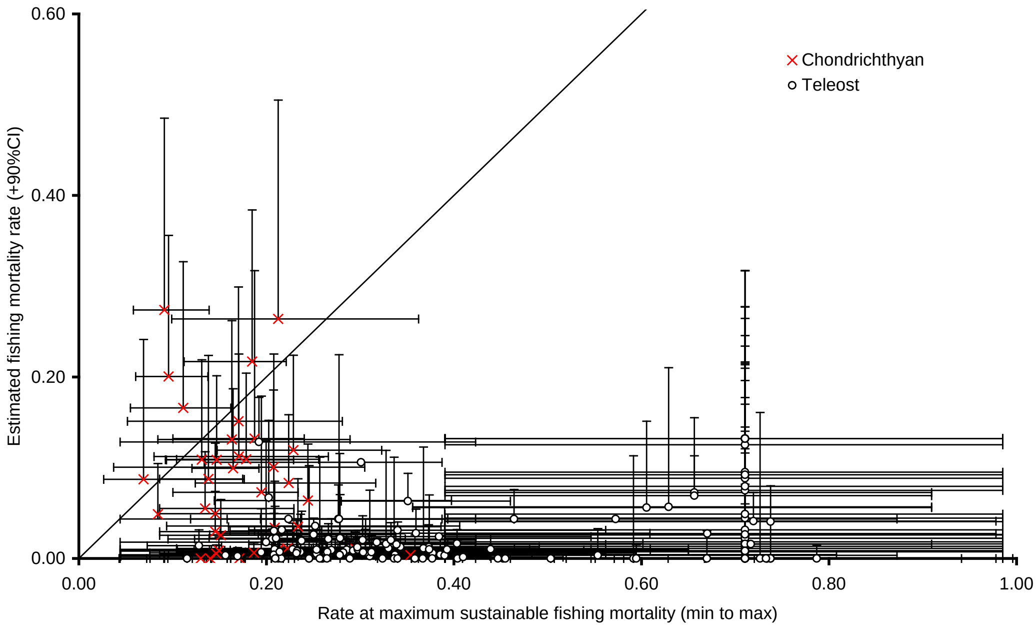

Figure 2: An example of quantifying risk in SAFE reports (Zhou et al. 2011). The diagonal line is where F = F_{msm}.

Another approach might be to focus on the probability of the Risk Ratio exceeding 1, and so heading towards a bad outcome, instead of the Risk Ratio itself. The uncertainty might then be considered in terms of how that probability would be changed by the simultaneous occurrence of other anthropogenic mortalities. The priority during the definition of measures of risk needs to be on their accessibility and interpretability. Complex and unintuitive measures are more likely to be ignored or misinterpreted. Straightforward representations of potential harms and their probabilities may be less susceptible to this than other formulations. In this light, as an alternative to, or alongside the violin plots presented in the SEFRA applications, it is recommended that the approach shown below be considered (Figure and Figure 4) [RECOMMENDATION H3].

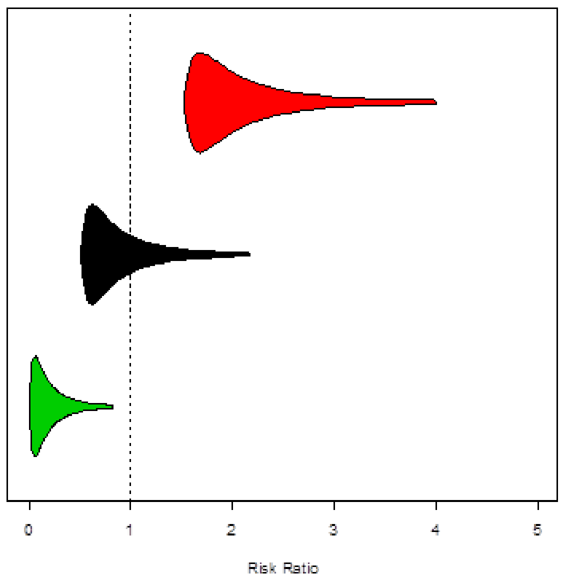

Figure 3: Plot of Risk Ratio for three notional species: Green - fishing impacts sustainable; Red - deleterious impacts; black – species probably, but not definitely, secure. In this example the management target is taken to be achieved at a Risk Ratio of 1; choice of a different target would shift the dotted line.

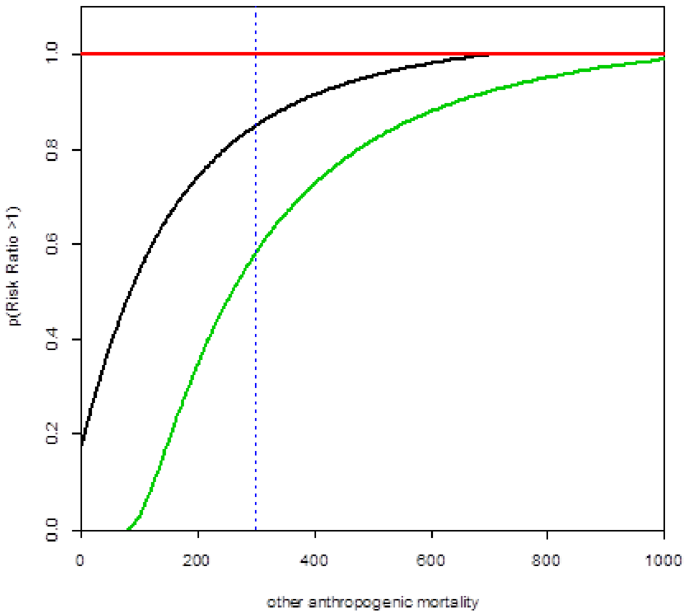

Figure 4: Probabilities of Relative Risk becoming greater than 1 (i.e., species being in trouble) for various levels of additional (non-bycatch) anthropogenic mortality. Each line corresponds to the same colour in Fig 1. The values on the x axis in this plot are notional, and in reality would depend on the particular population size and characteristics. The risk can be determined from the intercepts with the y-axis. So red indicates the species is clearly in trouble, green indicate that the species can sustain the current level of fishery bycatch, and black indicates that the species currently has a c. 80% chance of coping with the current situation.

The intersection with the x-axis shows how much other human impact each species could cope with alongside the effects of this fishery. So green would still be safe with around 80 other losses. The blue broken line shows the situation if there were another estimated 300 mortalities. It suggests that would give nearly a 60% chance of green being in trouble and over an 80% chance of black being in a bad way. In practice, the blue line is more likely to be a blue box because other mortalities are likely to be very poorly understood. This type of plot might provide a more accessible way of viewing the output from the model and be useful to people who are less comfortable with mathematical abstractions.

Figure 4, as drawn, considers only a threshold at RR=1, but can be considered as a slice through a three dimensional graph where the Risk Ratio threshold forms the third axis.

The cryptic mortality parameter is estimated outside the model using whatever information is available from other studies. To bring that estimation inside the main risk model would be difficult, and because it uses data that are not used elsewhere, may be of limited benefit (although in theory it would carry the associated uncertainty through to the final calculation of Risk). The more straightforward solution would be to include the estimate with an associated prior.

2.6 Priorities for future work (TOR Q9)

2.6.1 Population models

For some species for which there is concern (this might arise from a high risk ratio, RR, or a known declining trend even if RR is low) and for which sufficient information is available, population dynamics models should be developed [RECOMMENDATION L3]. Francis & Sagar (2012) provide a suitable framework for such a model and, since this was developed in New Zealand, there is likely to be expertise available. An alternative is the model developed in Australia (e.g. Tuck et al. 2015, Thomson et al.

2015). These models include fishing effort from all fleets that might be overlapping with the bird distributions, including those outside of New Zealand, and can incorporate other sources of mortality including from the environment. They can be used, in forward projections, to quantify the effects of future changes in bycatch mitigation (both spatially and by particular fishing fleets, and targeting specific life-history stages) or shifts in fishing effort. Sources of information (such as time series of population counts, survival estimates, breeding success etc. at the colony, shifts in bird distribution over time, sex and age composition of catches) can be incorporated into such model frameworks.

2.6.2 Sensitivity testing

A range of issues should be investigated in sensitivity analyses to determine the effects on the SEFRA outputs. These include: (i) use of number of hours trawled (which may better reflect the number of birds likely to be killed in collision with warp cables) vs number of hauls (this may be more closely-correlated than hours trawled with the number of birds recovered in nets; (ii) the relative advantages of: a) incorporating seasonal changes in bird distributions, and those created using more accurate tracking or other distribution data vs b) the default of a uniform distribution within the range, and symmetric exponential decay in numbers of breeders around the colony), which is used for species with no other distribution information; (iii) whether the distributions or population size information from the Delphi consultation for marine mammals are improvements on the assumption of uniform distributions; (iv) the effect of setting or limiting the multiplier for cryptic mortality to determine if this is a major determinant of the overall uncertainty (which is not provided by the current set of diagnostics); (v) comparison of results for data-rich species run with complete data, against those for the same species if included as data-poor, i.e., for which one or more aspects of the input data (spatial distribution, tightness of population or survival priors) are the robustness of the results degraded [RECOMMENDATION H10].

It would be desirable to also test the sensitivity of the model to (vi) the assumed distribution functions. One suggested method for doing that has been to base the model on a grid instead of using continuous distributions and then allocate a CV to each cell of the spatial grid from which densities are drawn (here, density means proportion of the population, not number of animals, in a cell). However, because the density has to integrate to 1 over the whole domain, this would be problematic. An easier way might to pre-specify a set of alternative maps (say 20 of these) and randomly draw from these for model iteration [RECOMMENDATION H10].

We note that Dragonfly performed a sensitivity test on the effects of not using a distribution (i.e., assuming a uniform distribution across the spatial domain) in the Marine Mammal Risk Assessment (Abraham et al. 2017) during the review process. This analysis involved a species-group by fishery fixed effect, and a species by area random effect. The resulting estimates of mammals killed and Risk Ratio had long tails, so that many species are potentially at risk, and the CVs were high. Bycatch mortality, and consequently the risk, were broadly similar for the species with many observed captures (e.g. New Zealand fur seal, common dolphin, New Zealand sealion, and Hectors dolphin). It may therefore be possible to develop a plausible model without distributions that has fewer degrees of freedom (similar high tails were seen in some versions of the SEFRA model, depending on the assumed structure of the vulnerability).

2.6.3 Simulation testing

The correct interpretation of Risk Ratio values not equal to one is not obvious. It would be useful to carry out simulations to determine at what abundances populations with different combinations of demographic parameters and Risk Ratios could be expected to settle [RECOMMENDATION M12]. Unless there is a clear and comprehensible pattern, it will be difficult to provide useful information on the interpretation and use of the absolute Risk Ratio. A simple transform may suffice if the patterns are non-linear, but more complex patterns may be more problematic.

2.6.4 Further risk assessment framework development

Although relative species vulnerabilities are potentially consistent across similar fisheries within New Zealand waters, the same assumption cannot be made elsewhere if the framework is extended to the entire Pacific and/or Southern Oceans. If present, diving seabirds (shearwaters or *Procellaria* petrels) return baited hooks back to the surface, particularly those on long leaders in surface longline fisheries; these hooks are then more accessible to albatrosses, which are poor divers, increasing their bycatch risk (Jiménez et al. 2012). In addition, bycatch rates of species such as grey-headed albatrosses tend to be low when more aggressive species, such as black-browed albatrosses are present in large numbers (often the case in shelf waters), but much higher in surface longline fisheries in oceanic waters when they may be the most common *Thalassarche* species, and therefore much more likely to scavenge behind vessels. For these reasons, regional changes in species assemblages associated with fishing vessels needs to be taken into consideration [RECOMMENDATION L9].

In some fisheries, observers collect counts, by species, of birds present around fishing vessels during routine observation, which could be of value for improving bird distribution maps (although with caveats, because there will be no data in unfished areas) [RECOMMENDATION L10].

In addition, the SEFRA, like many models, assumes a linear relationship between fishing effort and birds captured. However, saturation is likely to occur at some point e.g. the total number of birds that can be caught cannot exceed the number of birds present at the fishing vessel, and the shape of this relationship could potentially be explored using the vessel-based counts [RECOMMENDATION M6].

It would be worth considering whether any information on marine mammal strandings could be used to improve distribution models for these species [RECOMMENDATION L10].

2.7 Other

2.7.1 Clear model specification

It would be desirable to move towards a single internally consistent mathematical formulation of the equations that are used to calculate RR that would be used by all users of the method [RECOMMENDATION M3].

The document provided by Webber (Appendix 5) was useful. It is clearly an early draft, and some suggestions are given, such as several symbols that need to be added to its table 1 (e.g. cryptic mortality, probability densities p). One concern is how much uncertainty a full Bayesian model can accommodate. Trying to estimate a range of parameters simultaneously can result in MCMC chains not converging or variance too large to be meaningful. The description of the model includes two α parameters and a β parameter (not estimated) that specify how much of an animal's time is spent in New Zealand waters during the breeding and non-breeding seasons. These are necessary, but could easily lead to misunderstanding or even error as they inform the definition of N; this could be the global population size (all α and β values would be < 1), the size of the population in New Zealand waters at some reference time period (one of the α and β values would be $= 1$), or some less easily understood definition if all of these parameters exceeded 1. Those working with the model should be very clear regarding their definition of N. The values of α and β should be fixed based on other available information (such as tracking data) and should not be included as estimable parameters in the SEFRA because this is bound to lead to non-identifiability of confounded parameters. As a minimum, absolute N within the spatial domain of the model has to be known so that other parameters such as vulnerability can be estimated [RECOMMENDATION H7].

The correction factors for N calculated by Richard & Abraham (2017) should be incorporated in Webber's description. Also, the a-dash (a') variables in equations 21 and 22 should surely be just "a", not only the observed a-dash. The summation over "i,s in z" in the same equations would be more clearly expressed as two separate summation terms, one over "all i" and the other over "s in z".

3 RESPONSE TO TERMS OF REFERENCE

While largely following the structure of the questions we were asked, we found it easier to arrange and group the issues in a slightly different way. Table 1 shows where we have addressed each of the original questions, and which recommendations relate to each one.

Table 1: Coverage of the questions within the terms of reference by the subsections of section 2, and the recommendations associated with each question.

Question within the Terms of Reference	Subsection addressing question	Recommendation numbers
1. Is the conceptual framework sound, and how does it compare to international best practice?	2.1	H2; H3; M3–M7; L1–L5
2. Can the analytical methods be improved? Are there alternative analytical methods that should be considered?	2.3	M6; L2
3. Can the Panel recommend particular diagnostics that can be used to validate or improve model structural assumptions and input parameterisations?	2.5	M8; M11
4. Can the Panel recommend best practice in the selection and preparation of input data?	2.2	M2; M8; L6; L7
5. Can the Panel provide specific recommendations to improve the most recent implementations of the framework; i.e., the New Zealand seabird risk assessment and the New Zealand marine mammal risk assessment?	2.3	M9; M10; L7; L8
6. Can the assumptions used in these implementations of the framework be improved?	2.3	H4–H6
7. Can the Panel provide guidance on the application of this framework to other fisheries risk assessments (e.g. impacts on benthic habitats, non-target fish species and low information stocks)?	2.4	L5
8. Are there better ways to display or communicate the outputs of risk assessments?	2.5	H3; H7–H10
9. What are the comparative pros and cons of collecting better input data versus further risk assessment framework development?	2.6	H10; M6; M12; L3; L9; L10

Recommendations H1 and M1 concern wider issues around the method rather than directly answering the specific questions asked.

4 RECOMMENDATIONS

The members of the Panel identified 62 points on which they wanted to comment in the text of this report. These are highlighted in Section 2 of this report. The list contains some duplication and has been collapsed into 32 recommendations, which are classified as high (H), medium (M) or low (L) priority and tabulated below (Table 2) along with the numbers of the section they appear in.

Table 2: The Recommendations of the Panel and the section of the main text each one appears in. The recommendations are classified as high (H), medium (M) or low (L) priority and numbered.

Recommendation number	Short description	Appears in Section…
H1	MPI, as the primary driver of this work, endeavour to maintain communication between all those working on the framework.	1
H2	There should be explicit discussion with managers of the targets, assumptions, and limitations of the method. This should give particular consideration to situations where the Risk Ratio is above 1 or the available data are very sparse.	2.1.1; 2.1.2.3; 2.1.4
H3	Other ways of displaying the outputs (including the information in Risk Ratios) be used alongside the violin plots.	2.1.2.1.2; 2.1.2.3; 2.5
H4	All priors intended to be uninformative should be re-examined, and the effect of increasing their variances investigated.	2.3.2
H5	The effects of selective (by sex or age-class) bycatch and the unavailability of birds sitting on nests be investigated.	2.3.4; 2.3.5; 2.3.6
H6	Model fits where constraints on priors strongly inform posterior distributions should be treated with caution and the survivorship constraint should remain at its current extreme value.	2.3.5
H7	Whether all the parameters in the model can really be estimated or some collinear or unidentifiable terms need to be combined.	2.5; 2.7.1
H8	The units of all parameters and variables should be reported.	2.5
H9	The distinction between S_{curr}, the current survival rate, and S_{opt}, and the survival rate that relates to maximum population growth (r_{max}) should always be clear.	2.5
H10	Additional sensitivity testing be carried out.	2.5; 2.6.2
M1	The method be submitted for publication in a peer-reviewed scientific journal.	1
M2	Emphasis shift from the model and towards the collection of better input data and development of the management process that surrounds and utilises the science.	1; 2.2.2.2
M3	That a consistent reference formulation of the equations for the models be decided upon and clearly documented. Alternative implementations also need to document both their differences and the reasons for these differences.	2.1.1; 2.7.1
M4	The potential for modularising the code implementing the method be examined.	2.1.1
M5	The effect of the CV used for inter-annual population variability be investigated, particularly for marine mammals.	2.1.1.2
M6	Assumptions of linear relationships within the models be identified, re-examined, and tested where possible.	2.1.2.1.1; 2.1.2.1.2; 2.3.6; 2.6.4
M7	Further simulation testing be carried out (with emphasis on model misspecification and bias, and the characteristics of marine mammal populations).	2.1.2.3
M8	The use of elicited priors be reconsidered, their effects examined by simulation, and additional advice sought on the robustness of the elicitation methodology.	2.2.2.1; 2.5

M9	Consistency should be applied to dealing with over-dispersion in the SEFRA framework, with consistent and documented choices being made between negative binomial and quasi-Poisson distributions.	2.3.3
M10	Consideration should be given to rescaling vulnerability parameters to simplify comparisons between different fisheries.	2.3.6
M11	Detailed diagnosis of goodness of fit seems unlikely to be very informative in spatial models based on sparse data.	2.5
M12	Investigation of the implications, for abundance, of Risk Ratio values not equal to one.	2.6.3
L1	There is contingency planning for risks of non-completion of the current implementation or the unavailability of the current developer.	2.1.1
L2	Consideration of the effects of the local density of species on bycatch rates.	2.1.2.1.1; 2.1.2.1.2; 2.3.4
L3	Population dynamics models should be developed for species where sufficient information is available.	2.1.2.1.2; 2.6.1
L4	Trends in any available time series be considered.	2.1.2.3
L5	Consideration of how to incorporate other information on the history, status and other threats to, particularly seabird, species including those breeding outside New Zealand.	2.1.4; 2.4;
L6	The allometric analysis should also include data from outside the New Zealand area, particularly for bird families that are poorly represented within the current analysis, and exclude populations likely to have been impacted by fisheries, or which appear to be anomalously low for methodological reasons.	2.2.1
L7	The effects of fitting separate slopes for different taxonomic groups be investigated and the use of phylogenetically independent contrasts should be considered as a way to account for the apparently contrasting allometric relationships with survival that are apparent between seabird families or orders.	2.2.1; 2.3.5
L8	Implementations of SEFRA should favour lumping rather than splitting species and fishery groups, and tables reporting the number of observations available in each species group by fishery group cell of the matrix should be presented.	2.3.4
L9	Consideration of the effects of interactions between bycatch species.	2.6.4
L10	The potential utility of other observations (including from ships and strandings) be re-examined.	2.6.4

5 ACKNOWLEDGMENTS

All participants listed in Appendix 3 are thanked for their contributions to presenting and discussing the work reviewed here. The Ministry for Primary Industries is thanked for funding the review.

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APPENDIX 1. TERMS OF REFERENCE

Ministry for Primary Industries Terms of Reference for: An independent review of New Zealand's spatially explicit fisheries risk assessment approach – 2017

1. Background

Over the past several years New Zealand has developed and adopted a Spatially Explicit Fisheries Risk Assessment framework (SEFRA) to assess (and inform the management of) population-level risk to non-target species incidentally captured in commercial fisheries within a rigorous and transparent quantitative framework.

The SEFRA method combines a spatially explicit impact assessment with a biological assessment of the associated population-level effect. The impact assessment works by estimating the rate at which animals encounter fishing effort as a function of the spatial and temporal overlap between animal distributions and fishing effort distributions. Captures and mortalities per encounter are estimated using fisheries observer data.

In the most advanced implementations of the SEFRA framework, captures, mortalities, and population level risk are estimated for multiple species simultaneously using fully integrated Bayesian models, with explicit consideration of uncertainty from input parameters through to output estimates of mortalities and population level risk.

SEFRA outputs do not only estimate impact and risk at a population level; instead, because all such estimates are explicit in space and time, SEFRA outputs can be disaggregated and interrogated at any scale, for example to compare risk arising from different fisheries, methods, locations, or time periods.

Because risk estimates arise from transparent mathematical formulations, it is also possible to adjust these inputs to evaluate alternate management scenarios, including for example mitigation, spatial management options, or investments in higher levels of fisheries observer coverage. In this way SEFRA outputs can inform risk management, not just risk assessment.

This review will consider:

- the conceptual basis and mathematical formulation of the SEFRA method as it was originally conceived, with reference to particular implementations as illustrative examples only;
- two specific implementations of the SEFRA method, for New Zealand seabirds and New Zealand marine mammals;
- recent development of new tools to inform interrogation and evaluation of SEFRA outputs to inform risk management; and
- options for the application of the SEFRA framework to other New Zealand fisheries risk assessments, e.g. for non-target fish and benthic habitats.

2. Terms of Reference

- An independent Panel comprising Drs Shijie Zhou, Mike Lonergan, Robin Thomson and Richard Phillips, will be convened. All the members of the Panel have extensive scientific expertise in risk assessment methodology, seabirds, marine mammals or more than one of these disciplines.

- Panel members must declare any actual or possible conflicts of interest that might affect their ability to come to an objective view of the risk assessment approach.

- The review will be chaired by Pamela Mace, the Principal Advisor Fisheries Science. Ben Sharp, the primary developer of the risk assessment framework, and other presenters including Ed Abraham and Marie-Julie Roux will be available continuously to the Panel. The Chairs of the Aquatic Environment Working Group will also be available if required.

- Reviewers will be asked to address the following questions:

  1. Is the conceptual framework sound, and how does it compare to international best practice?

  2. Can the analytical methods be improved? Are there alternative analytical methods that should be considered?

  3. Can the Panel recommend particular diagnostics that can be used to validate or improve model structural assumptions and input parameterisations?

  4. Can the Panel recommend best practice in the selection and preparation of input data?

  5. Can the Panel provide specific recommendations to improve the most recent implementations of the framework; i.e., the New Zealand seabird risk assessment and the New Zealand marine mammal risk assessment?

  6. Can the assumptions used in these implementations of the framework be improved?

  7. Can the Panel provide guidance on the application of this framework to other fisheries risk assessments (e.g. impacts on benthic habitats, non-target fish species and low information stocks)?

  8. Are there better ways to display or communicate the outputs of risk assessments?

  9. What are the comparative pros and cons of collecting better input data versus further risk assessment framework development?

- Questions 2, 4 and 5 above should be considered in terms of spatially-explicit density estimation and overlap calculations; impact scalars (vulnerability, catchability, susceptibility); population size proxies; and sustainability thresholds definition).

- The expert Panel will summarise their findings and any recommendations in a report to the Principal Advisor Fisheries Science, Ministry for Primary Industries. Where consensus cannot be reached by the external reviewers, any differences of opinion should be recorded.

3. Rules of participation

Relevant aspects of the “Membership and Protocols for all Science Working Groups in 2017” will be followed for this review. In particular (adapted from paragraphs 4, 10 and 11 of that document):

Participants must commit to:

- participating appropriately in discussions;
- maintaining confidentiality of presentations, discussions and deliberations;
- adopting a constructive approach;
- avoiding repetition of earlier deliberations;
- facilitating an atmosphere of honesty, openness and trust;
- respecting the role of the Chair and the Panel; and
- listening to the views of others, and treating them with respect

It is extremely important to the proper conduct of these reviews that all contact with the Panel reviewers is through the Chairs of the Aquatic Environment Working Group or the Principal Advisor Fisheries Science.

Under no circumstances should participants approach the Panel reviewers directly until after the final report of the review has been published.

4. Out of scope

The review will focus on the risk assessment methodologies, not the risk assessment results.

Management implications of the risk assessments and the setting of population targets will not be considered as science is only one input to such decisions. However, input on better ways to develop or communicate the science will be welcomed.

5. Background documents

The following documents will be provided:

- The risk assessment chapter of the Aquatic Environment and Biodiversity Annual Review (AEBAR).
- The latest seabird risk assessment Aquatic Environment and Biodiversity Report
- The marine mammal Aquatic Environment and Biodiversity Report
- Illustrative outputs of a spatially explicit risk assessment disaggregation and query tool (new contract in progress)

6. Format for review

The format for the review will be a workshop involving the independent external reviewers, key players and other interested parties in Wellington, New Zealand. The review will start with a number of presentations to ensure a common understanding of the work (about 1.5 days), and be followed by a period of contemplation, focused discussions with the lead researcher or other parties (at the Panel's discretion), and drafting of a report containing conclusions and recommendations (2–3 days). The review Panel will present a draft version of their findings to interested parties on the last day to receive feedback and suggested corrections on matters of fact. The review Panel may, at their discretion, reflect such feedback in their report. The aim would be to have a near-final version of the report by the end of the week.

7. Timetable

The workshop is set down for 12 - 16 June, 2017 and will be held in the Allen Board Room, National Institute for Water and Atmospheric Research (NIWA), Greta Point, Wellington, New Zealand. Pamela Mace, MPI, will chair the open sessions.

Monday 12 June	Presentations	Open session
Tuesday 13 June a.m.	Presentations conclude	Open session
Tuesday 13 June p.m.	Panel confers with individuals and writes review	Panel's discretion
Wednesday 14 June	Panel confers with individuals and writes review	Panel's discretion
Thursday 15 June	Panel confers with individuals and writes review	Panel's discretion
Friday 16 June a.m.	Panel presents draft findings	Open session
Friday 16 June p.m.	Panel concludes review	Closed session

It is anticipated that the review can be concluded by 5 pm on Friday 16 June, although final drafting of the report may take place over subsequent days.

APPENDIX 2. MEETING AGENDA

Ministry for Primary Industries Agenda for an independent review of New Zealand's spatially explicit fisheries risk assessment approach – 2017

NIWA, Greta Point, Evans Bay, Wellington   Chair: Pamela Mace

Monday 12 June

1. 9:30 am – Welcome and introductions
2. 9:45 am – Terms of reference (including out of scope): Chair
3. 10:00 am – Management context: Tiffany Bock and Erin Breen

10:30 Morning tea

4. 11:00 am – New Zealand's risk assessment framework: Ben Sharp

1:00 pm Lunch

5. 1:45 pm – Seabird data and risk analysis: Ed Abraham

3:15 pm – Afternoon tea

6. 3:45 pm – Marine mammal data and risk analysis: Ed Abraham

5:00 pm Adjourn

Tuesday 13 June

7. 9:30 am – Outputs of customised risk assessment disaggregation and query tool: Ben Sharp

11:00 am – Morning tea

8. 11:30 am – Progress on other SEFRA implementations (e.g. benthic risk analysis): Marie-Julie Roux

1:00 pm Lunch

9. 1:45 pm – Further discussion, if needed: led by Expert Panel

Tuesday pm, Wednesday 14 June, Thursday 15 June

Panel sequestered – no open sessions

Friday 16 June

9:30 am – Panel presents draft conclusions and recommendations to open session; participants offer comment

12:30 onwards – Panel finalises report: no open sessions

Note: Extended discussions around the presentations filled the whole of both Monday and Tuesday up to 5pm each day

APPENDIX 3. LIST OF PARTICIPANTS AND PANEL MEMBERS

COMBINED ATTENDANCE for the open sessions

Pamela Mace (MPI - Chair) Rich Ford (MPI), Tiff Bock (MPI), Nathan Walker (MPI), Mary Livingston (MPI), Ben Sharp (MPI), John Annala (MPI), Erin Breen (MPI), Conor Neilson (MPI), Kevin Sullivan (Independent consultant for MPI), Marie-Julie Roux (NIWA), Charles Edwards (NIWA), Jim Roberts (NIWA), Edward Abraham (Dragonfly Data Science), Yvan Richard (Dragonfly Data Science), Amanda Leathers (WWF), Anton van Helden (Forest and Bird), Barry Weeber (ECO), Igor Debski (DOC), Geoff Tingley (Independent Consultant for Deepwater Group).

Panel: Shijie Zhou (CSIRO), Robin Thomson (CSIRO), Mike Lonergan (University of Dundee), Richard Phillips (British Antarctic Survey).

BIOGRAPHIES AND INDEPENDENCE:

Mike Lonergan (University of Dundee):
Mike is a statistician with experience of both marine mammal and medical data. He was a biometrician for the Sea Mammal Research Unit, at the University of St Andrews, for ten years. His work at that time focussed on estimating the abundances and trajectories of harbour and grey seals around the UK. He has also been involved in estimating the environmental impacts of tidal turbines, studies of bycatch mitigation for harbour porpoise in the UK, and evaluating the justifications of conservation targets. He was a member of two previous Panels that reviewed methods for quantifying threats to New Zealand sea lions, but has no other interests that could conflict with involvement in this review.

Richard Phillips (British Antarctic Survey):
Professor Richard A. Phillips is a seabird ecologist, appointed at the British Antarctic Survey (BAS), Cambridge, in 2000, and currently leader of the Higher Predators and Conservation group (15 staff) in the Ecosystems programme. His work focuses on the ecology and conservation of seabirds. He has 245 peer-reviewed papers in press or published, cited > 9,200 times since 1996, and has an h-index of 50 (Google Scholar accessed 1 Jun., 2017). He is closely involved in international initiatives to improve the conservation of seabirds, including as convenor of the ACAP Populations and Conservation working group, and was closely involved with the Seabird Risk Assessment of the Ecosystems sub-committee of the International Commission for the Conservation of Atlantic Tuna (ICCAT) in the late 2000s. He has no competing interests with regard to this review.

Robin Thomson (CSIRO):
Robin works in the field of stock assessment and population dynamics modelling. Together with Geoff Tuck (CSIRO) she has developed a population dynamics modelling framework for seabirds that incorporates both natural mortality and incidental capture in fishing operations as well as environmental influence on chick survival. She has applied the model to several albatross populations. Robin has also developed models for investigating the link between krill abundance and predator survival for a number of Antarctic seabird and pinniped species using data collected by the CCAMLR CEMP program. Robin has no competing interests with regard to this review.

Shijie Zhou (CSIRO):
Shijie Zhou is a Senior Principal Research Scientist at CSIRO’s Oceans and Atmosphere, Australia. In recent years he undertook research on fishery and ecosystem dynamics modelling. He led research in developing methods in population dynamics and ecological risk assessment, including the SAFE method (Sustainability Assessment for Fishing Effect). His current research interests involve fisheries stock assessment, bycatch and discards, methods for data poor species, Bayesian modelling, and fisheries management. He has research experience in Asia and North America and is an Editor for the ICES Journal of Marine Science. Shijie has no competing interests with regard to this review.

APPENDIX 4. LIST OF BACKGROUND DOCUMENTS PROVIDED TO THE PANEL

Abraham, E.R.; Neubauer, P.; Berkenbusch, K.; Richard, Y. (2017). Assessment of the risk to New Zealand marine mammals from commercial fisheries. New Zealand Aquatic Environment and Biodiversity Report DRAFT.

Berkenbusch, K.; Abraham, E.R. (2017). Estimated captures of New Zealand fur seal, New Zealand sea lion, common dolphin, and turtles in New Zealand trawl and longline fisheries, 1995–96 to 2014–15. New Zealand Aquatic Environment and Biodiversity Report DRAFT.

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Spatially Explicit Risk Assessment Tool: Seabird Implementation

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Still missing all of the math for years and averaging across years. I will get to this some other time.

Table 1: Notation used in discussing and defining the model.

Symbol	Support	Description
N_s	≥ 0	Total population size
N_s^{BP}	≥ 0	Number of breeding pairs
N_s^{B}	≥ 0	Number of breeders
N_s^{NB}	≥ 0	Number of non-breeders
N_s^{adults}	≥ 0	Number of adults
$N_s^{\text{juveniles}}$	≥ 0	Number of juveniles
P_s^B	$0 \leq P_s^B \leq 1$	Proportion breeding
S_s^{curr}	≥ 0	Adult survival
S_s^{opt}	≥ 0	Adult survival
A_s^{curr}	≥ 0	Age at first reproduction
A_s^{opt}	≥ 0	Age at first reproduction
α_s^B	$0 \leq \alpha_s^B \leq 1$	Proportion of breeding individuals remaining in spatial domain during breeding season
α_s^{NB}	$0 \leq \alpha_s^{NB} \leq 1$	Proportion of non-breeding individuals remaining in spatial domain during breeding season
β_s	$0 \leq \beta_s \leq 1$	Proportion of individuals remaining in spatial domain during non-breeding season
Ψ_{sg}	≥ 0	Probability of an individuals being alive given that it was caught
ω_{sg}	≥ 0	Probability that a live released inidividual will survive
V_{sg}	≥ 0	Vulnerability (combining species vulnerability and fishing group vulnerabilty terms)

1 Risk Ratio

We will start at the end and define the risk ratio R_{sgi} for each species s, fishing group g, and fishing event i as

$$R_{sgi} = \frac{D_{sgi}}{PST_s} \tag{1}$$

where D_{sgi} is the number of fishery related deaths and PST_s is the population sustainability threshold (PST).

2 Population Sustainability Threshold (PST)

The PST for each species is defined as

$$PST_s = \frac{1}{2}\phi_s r_s^{\max} N_s, \tag{2}$$

where ϕ_s is a correction factor that allows for the calibration of the PST to achieve particular management goals (e.g. $\phi_s = 0.5 \forall s$ may be used so that populations meet the long-term goal of remaining above half their carrying capacity), $r_s^{\max}$ is the maximum population growth rate under optimal conditions, and N_s is the total population size.

$$\lambda_s^{\max} = \exp\left[\left(A_s^{\text{opt}} + \frac{S_s^{\text{opt}}}{\lambda_s^{\max} - S_s^{\text{opt}}}\right)^{-1}\right] \tag{3}$$

$$r_s^{\max} = \lambda_s^{\max} - 1 \tag{4}$$

where A_s^{opt} is the age at first reproduction. Currently $A_s^{\text{opt}} = A_s^{\text{curr}}$, I wonder if they were to differ that $A_s^{\text{opt}} \leq A_s^{\text{curr}}$?

3 Population Size

The total population size is

$$N_s = \frac{2N_s^{BP}}{P_s^B} \left(S_s^{\text{curr}}\right)^{1-A_s^{\text{curr}}} \tag{5}$$

The number of adults is

$$N_s^{\text{adults}} = \frac{2N_s^{BP}}{P_s^B} \tag{6}$$

Therefore it follows that the number of juveniles is

$$\begin{aligned} N_s^{\text{juveniles}} &= N_s - N_s^{\text{adults}} \\ &= \frac{2N_s^{BP}}{P_s^B} \left(\left(S_s^{\text{curr}}\right)^{1-A_s^{\text{curr}}} - 1\right) \end{aligned} \tag{7}$$

The number of breeders is

$$N_s^B = 2N_s^{BP} \qquad \text{where} \qquad 0 \leq N_s^{BP} \leq N_s^B \leq N_s \tag{8}$$

The number of non-breeders is

$$\begin{aligned} N_s^{NB} &= N_s - N_s^B \\ &= N_s - 2N_s^{BP} \qquad \text{where} \qquad 0 \leq N_s^{BP} \leq N_s \end{aligned} \tag{9}$$

4 Overlap

Overlap is

$$O_{sgi} = \begin{cases} a_{gi}\left(p_{si}^B + p_{si}^{NB}\right) & i \in \text{breeding season} \\ a_{gi} p_{si}^{NB} & i \in \text{non-breeding season} \end{cases} \tag{10}$$

where a_{gi} is fishing intensity, p_{si}^B is the probability density of breeders, p_{si}^{NB} is the probability density of non-breeders.

Density overlap is

$$\mathbb{O}_{sgi} = \begin{cases} a_{gi}\left(p_{si}^B N_s^B \alpha_s^B + p_{si}^{NB} N_s^{NB} \alpha_s^{NB}\right) & i \in \text{breeding season} \\ a_{gi} p_{si}^{NB} N_s \beta_s & i \in \text{non-breeding season} \end{cases} \tag{11}$$

where α_s^B is the proportion of the breeding population in NZ during the breeding season, α_s^{NB} is the proportion of the non-breeding population in NZ during the breeding season, β_s is the proportion of the total population in NZ outside the breeding season.

5 Captures

Total captures are calculated as

$$C_{sgi} = \mathbb{O}_{sgi} V_{sg} \frac{1}{k_{sg}} \qquad \text{where} \qquad C_{sgi} \geq 0, V_{sg} \geq 0 \tag{12}$$

where V_{sg} is the vulnerability, k_{sg} is the cryptic interaction term, and total captures includes both live and dead captures $C_{sgi} = C_{sgi}^{\text{live}} + C_{sgi}^{\text{dead}}$. The cryptic interaction term could be written as the probability that an event is observable instead

$$p^{\text{observable}} = \frac{1}{k_{sg}} \tag{13}$$

Note that we could combine vulnerability and the cryptic interaction term to form the catchability q_{sg}

$$q_{sg} = V_{sg} \frac{1}{k_{sg}} \tag{14}$$

but I keep the terms separate to remind me about the components of captures, deaths and risk.

$$C_{sgi}^{\text{live}} = \mathbb{O}_{sgi} V_{sg} \frac{1}{k_{sg}} \Psi_{sg} \tag{15}$$

$$C_{sgi}^{\text{dead}} = \mathbb{O}_{sgi} V_{sg} \frac{1}{k_{sg}} \left(1 - \Psi_{sg}\right) \tag{16}$$

where $\Psi_{sg} = \psi_s \psi_g$ and represents the probability of an individuals being alive given that it was caught.

6 Deaths

$$\begin{aligned} D_{sgi} &= C_{sgi}^{\text{live}} \left(1 - \omega_{sg}\right) + C_{sgi}^{\text{dead}} \\ &= \mathbb{O}_{sgi} V_{sg} \frac{1}{k_{sg}} \Psi_{sg} \left(1 - \omega_{sg}\right) + \mathbb{O}_{sgi} V_{sg} \frac{1}{k_{sg}} \left(1 - \Psi_{sg}\right) \\ &= \mathbb{O}_{sgi} V_{sg} \frac{1}{k_{sg}} \left(1 - \Psi_{sg} \omega_{sg}\right) \end{aligned} \tag{17}$$

where ω_{sg} is the live release survival rate.

7 Mortality Constraint

$$U_s = \frac{\sum_{gi} D_{sgi}}{N_s^{\text{adults}}} \qquad \text{where} \qquad U_s \leq 1 - S_s^{\text{curr}} \tag{18}$$

This assumes that all bird caught in fisheries were adults. I need to think about this more, if we expand deaths and overlap then we may be able to reorganise this. Natural mortality and potentially non-fishery threats could be added to this as well...

8 Bayesian Inference

We are interested in estimating the parameters of our model but must do so using only observed data. We denote variables as being observed using the $'$ (prime) symbol (e.g. if all effort is a_{gi} then observed effort is a'_{gi}). In general, observed effort can be thought of as a subset of the total effort, thus we could write

$$a'_{gi} \subset a_{gi} \tag{19}$$

which means that a_{gi} contains the observed fishing events a'_{gi} and the unobserved fishing events.

The density overlap calculation in equation 11 is computationally expensive and should not be attempted inside the model. Instead we can write

$$\mathbb{O}_{sg} = \begin{cases} o^B_{sg} N^B_s + o^{NB}_{sg} N^{NB}_s & \text{breeding season} \\ o_{sg} N_s & \text{non-breeding season} \end{cases} \tag{20}$$

where

$$\begin{aligned} o^B_{sg} &= \alpha^B_s \sum_i a'_{gi} p^B_{si} && i \in \text{breeding season} \\ o^{NB}_{sg} &= \alpha^{NB}_s \sum_i a'_{gi} p^{NB}_{si} && i \in \text{breeding season} \\ o_{sg} &= \beta_s \sum_i a'_{gi} p^{NB}_{si} && i \in \text{non-breeding season} \end{aligned} \tag{21}$$

Equation 20 can be evaluated easily within the model.

By species group we might write

$$\begin{aligned} o^B_{zg} &= \alpha^B_s \sum_{is \in z} a'_{gi} p^B_{si} && i \in \text{breeding season} \\ o^{NB}_{zg} &= \alpha^{NB}_s \sum_{is \in z} a'_{gi} p^{NB}_{si} && i \in \text{breeding season} \\ o_{zg} &= \beta_s \sum_{is \in z} a'_{gi} p^{NB}_{si} && i \in \text{non-breeding season} \end{aligned} \tag{22}$$

What about for black petrel?

The data: $\boldsymbol{y} = \left\{ \left(C^{\text{live}}_{sg}\right)', \left(C^{\text{dead}}_{sg}\right)' \right\}$

The covariates: $\boldsymbol{z} = \left\{ o^B_{sg}, o^{NB}_{sg}, o_{sg} \right\}$

The unknown parameters: $\boldsymbol{\theta} = \left\{ v_s, v_g, k_{sg}, \psi_s, \psi_g, \omega_{sg}, \tau, \varepsilon_{sg}, N^{BP}_s, P^B_s, S^{\text{curr}}_s, A^{\text{curr}}_s \right\}$

Using Bayes theorem, the posterior distribution of the model parameters ($\boldsymbol{\theta}$), given the data ($\boldsymbol{y}$) and covariates ($\boldsymbol{z}$) is

$$\pi\left(\boldsymbol{\theta}|\boldsymbol{y}, \boldsymbol{z}\right) \propto \pi\left(\boldsymbol{\theta}\right) \pi\left(\boldsymbol{\theta}|\boldsymbol{y}, \boldsymbol{z}\right) \tag{23}$$

where the prior is

$$\begin{aligned} \pi\left(\boldsymbol{\theta}\right) &= \pi\left(v_s, v_g, \psi_s, \psi_g, k_{sg}, \omega_{sg}, \tau, \varepsilon_{sg}, N^{BP}_s, P^B_s, S^{\text{curr}}_s, A^{\text{curr}}_s\right) \\ &= \pi\left(v_s\right) \pi\left(v_g\right) \pi\left(\psi_s\right) \pi\left(\psi_g\right) \pi\left(k_{sg}\right) \pi\left(\omega_{sg}\right) \pi\left(\tau\right) \pi\left(\varepsilon_{sg}|\tau\right) \pi\left(N^{BP}_s\right) \pi\left(P^B_s\right) \pi\left(S^{\text{curr}}_s\right) \pi\left(A^{\text{curr}}_s\right) \end{aligned} \tag{24}$$

and the likelihood is

$$\begin{aligned}
\pi\left(\boldsymbol{y}|\boldsymbol{\theta},\boldsymbol{z}\right) &= \prod_s \prod_g \pi\left(\left(C_{sg}^{\text{live}}\right)', \left(C_{sg}^{\text{dead}}\right)' | v_s, v_g, k_{sg}, \psi_s, \psi_g, \varepsilon_{sg}, N_s^{BP}, P_s^B, S_s^{\text{curr}}, A_s^{\text{curr}}, o_{sg}^B, o_{sg}^{NB}, o_{sg}\right) \\
&= \prod_s \prod_g \pi\left(\left(C_{sg}^{\text{live}}\right)' | v_s, v_g, k_{sg}, \psi_s, \psi_g, \varepsilon_{sg}, N_s^{BP}, P_s^B, S_s^{\text{curr}}, A_s^{\text{curr}}, o_{sg}^B, o_{sg}^{NB}, o_{sg}\right) \\
&\times \prod_s \prod_g \pi\left(\left(C_{sg}^{\text{dead}}\right)' | v_s, v_g, k_{sg}, \psi_s, \psi_g, \varepsilon_{sg}, N_s^{BP}, P_s^B, S_s^{\text{curr}}, A_s^{\text{curr}}, o_{sg}^B, o_{sg}^{NB}, o_{sg}\right)
\end{aligned} \tag{25}$$

notice that ω_{sg} does not appear above and τ is in the prior.

The likelihood of this model is made up of two main components: the likelihood of live captures and dead captures.

$$\begin{aligned}
\left(C_{sg}^{\text{live}}\right)' &\sim \mathcal{P}\left(\mu_{sg}^{\text{live}}\right) \\
\left(C_{sg}^{\text{dead}}\right)' &\sim \mathcal{P}\left(\mu_{sg}^{\text{dead}}\right)
\end{aligned} \tag{26}$$

$$\tau \sim \mathcal{G}\left(0.001, 0.001\right) \tag{27}$$

$$\varepsilon \sim \log \mathcal{N}\left(0, \frac{1}{\sqrt{\tau}}\right) \tag{28}$$

is what has been used in the past but Gelman recommends against priors of this form. Instead use

$$\sigma \sim \mathcal{U}\left(0, 100\right) \tag{29}$$

$$\varepsilon \sim \log \mathcal{N}\left(0, \sigma^2\right) \tag{30}$$

8.1 Posterior Predictive Distributions

Talk about this for captures, deaths, risk.