A RISK ASSESSMENT FRAMEWORK FOR INCIDENTAL SEABIRD MORTALITY ASSOCIATED WITH NEW ZEALAND FISHERIES IN THE NZ-EEZ

Ben R. Sharp[1], Susan M Waugh[2], Nathan A. Walker[1]

1 New Zealand Ministry of Fisheries; 101-103 The Terrace; Wellington, New Zealand

2 Sextant Technology Ltd, 116 Wilton Road, Wellington 6012, New Zealand

Tel: +678 28078; Fax: +678 22518; E-mail: ben.sharp@vanuatu.com.vu

INTRODUCTION

This report outlines the risk assessment framework as developed at a workshop held on the 18th and 19th of February 2009 to support the revision of New Zealand's National Plan of Action - Seabirds. This report does not seek to specify the detail of the methods used to deliver results in subsequent risk assessment projects, but to detail the steps that make up the framework that should be used in those projects. Note that various methods could be used to deliver the framework.

Background of the National Plan Of Action - Seabirds

In the New Zealand area, a high number of seabirds occur, with around two thirds of the world's species of albatross and petrel known to occur within the New Zealand Exclusive Economic Zone. Of these, 77 species breed in New Zealand waters, and 39 are listed as threatened with extinction by the International Union for the Conservation of Nature (IUCN 2009). Of particular concern is the status of albatrosses, the world's most threatened family of birds, with 19 of the 22 species globally threatened with extinction (BirdLife International 2009). Eleven species of albatross nest in New Zealand.

As well as being a significant and unique part of the ecosystem, many species of albatross and petrel are considered to be *taonga* by tangata whenua and hold iconic status in the minds of the public of New Zealand.

Seabird species globally are facing a number of threats to their long term viability, both at the sites where they breed and while they are foraging at sea. One of the key threats at sea is the incidental catch of seabirds in the course of fishing activity. In longline fisheries, the baited hooks float on, or just below, the surface for a short time before they start sinking, thus they can be attacked by foraging seabirds that become hooked and drown. In trawl fisheries, contact with the warp cables causes significant levels of seabird mortality as seabirds forage on offal and discards from the vessel. Mortalities can also occur when birds dive into the trawl net as well as becoming entangled in the meshes when they are trying to seize fish.

Several population characteristics of albatrosses and petrels make them susceptible to long-term population decline from fishing-related mortalities. Albatrosses and petrels typically have an extended maturity time (3-15 years), low productivity (maximum of one nestling per year), and take a long time to form pairbonds if one partner is killed. If the death of a breeding individual occurs, the chick almost always dies and the remaining partner may take several years to start nesting again with a new partner.

The rate of population increase for these species is very low (around 1% per year), meaning that populations may not be able to compensate for fishing related removals fast enough to maintain healthy populations. As a result, decreases in population sizes and associated increasing seriousness of threat status are likely to occur.

There are two key pieces of legislation in New Zealand that are relevant to the impact of fishing activity on seabirds. These are the Wildlife Act 1953 and the Fisheries Act 1996. A number of international obligations are also relevant. The Fisheries Act requires the adverse effects of fishing on the aquatic environment to be avoided, remedied or mitigated. The Act also contains specific provisions relating to managing the effects of fishing on protected species. The Wildlife Act absolutely protects all but seven seabird species and partially protects two other species. However, the Act recognises and allows for the fact that fishing activity can result in the death of protected seabirds.

The principal international obligations stem from the Convention on Migratory Species (CMS), the Agreement for the Conservation of Albatrosses and Petrels (ACAP) and the FAO International Plan of Action for Reducing the Incidental Catch of Seabirds in Longline Fisheries (IPOA Seabirds). In addition, New Zealand has international obligations stemming from vessels fishing under the auspices of Regional Fishery Management Organisations (RFMOs) and the Antarctic Treaty system.

A key instrument to the management of protected species interactions with fisheries is the New Zealand National Plan of Action – Seabirds. This framework is currently being revised following the model of the International Plan of Action – Seabirds and associated Technical Guideline (FAO 1999, 2008) will provide for a reduction in risk to seabirds from fishing mortality associated with New Zealand fishing. Pivotal to an effective framework is the identification of which risks are most severe and urgent to address – where species are most under pressure from additional mortality above natural levels.

Impact assessment, risk assessment, and ERA

Risk assessment can be described as a systematic framework for evaluating the consequences of decisions subject to considerable uncertainty. Risk assessment approaches are commonly applied to a diverse range of problems, from structural engineering to epidemiology to management of environmental pollution Understanding the likely impacts of human activities on wildlife populations is essential to managing conservation threats. Ecological Risk Assessment (ERA) is a framework of methods that has increasingly been used as a means of assessing the relative likelihood of effects on species and ecosystems of human activities (Hobday *et al.* 2006, Standards New Zealand and Standards Australia 2006, Link *et al.* 2002, Hayes and Landis 2004, United States Environmental Protection Agency 1992). Risk assessment in a natural resource management context arises from the need for managers to make difficult decisions despite incomplete – and at times completely inadequate – actual information upon which those decisions can be based. Properly applied, risk assessment bridges the gap between scientists, who operate in the realm of what is known or can be estimated from existing data and attempt to expand on that knowledge, and managers, who do not have the luxury of waiting for the knowledge base to grow.

Broadly speaking, the challenge of any risk assessment is to assemble whatever relevant knowledge is available – whether quantitative or qualitative, objective or subjective – and devise a means to utilise that knowledge in the most rigorous and objective way possible to examine the probable outcomes of various management options, while maintaining transparency about the requisite inputs and assumptions and clearly identifying associated uncertainties. Well designed risk assessments of this kind constitute powerful tools both for informing management decisions to reduce risk despite ongoing uncertainty, and for directing future research for maximum efficiency, i.e. to improve knowledge in areas where the associated uncertainties are most critical for future decision-making. Ideally risk assessment frameworks should also be designed to facilitate incremental improvement as knowledge increases, such that new data can be incorporated and the consequences for the risk assessment outputs re-calculated without revisiting the entire risk assessment process (e.g. see Sharp et al. in press).

Discussion of the actual mechanics of the risk assessment process is often fraught with confusion arising from vague terminology and the inconsistent use of language. The term ‘risk assessment’ is commonly applied to a whole range of only loosely related analytic approaches, and the words themselves employed in these analyses - i.e. ‘risk’, ‘probability’, ‘frequency’, ‘likelihood’, ‘consequence’, ‘impact’ ‘hazard’, ‘effect’, ‘event’, ‘exposure’, ‘uncertainty’ – are too often imprecisely defined and inconsistently applied (e.g. see Kaplan 1997, Beer 2006, Fox 2006, Kerns and Ager 2007). Kaplan (1997) writes,

> *“When our Society for Risk Analysis was brand new, one of the first things it did was establish a committee to define the word ‘risk’. This committee labored for 4 years and then gave up, saying in its final report, that maybe it’s better not to define risk. Let each author define it in his own way, only please each should explain clearly what that way is.”*

It is important then to distinguish clearly between different risk assessment approaches and to select the most appropriate approach for a particular management application, and then to be clear about that selection and its implications, taking special care to define the operative terms.

‘Ecological risk assessment’ (ERA) is a field of risk assessment devoted to analyzing uncertain future outcomes in natural biological systems. ERAs are generally focused on ecological responses to a particular human or natural influence, e.g. pollution, storm events, or the effects of fishing. However the actual mechanics of ERA analyses span the full range of risk assessment approaches. In the ‘likelihood-consequence’ approach, total risk is expressed as a product of the expected likelihood and expected consequence of an ‘event’ (i.e. occurrence of the influence in question), usually combined and assigned a subjective rating or numerical score in a ‘likelihood-consequence matrix’ (e.g. Australian/ New Zealand Standards 1999, Crawford 2003, Fletcher 2005). With its emphasis on discrete low-frequency events, the likelihood-consequence approach is not ideally suited for the assessment of risks arising from activities that are predictable, ongoing and cumulative, such as the environmental effects of fishing. In particular, both ‘likelihood’ and ‘consequence’ are unavoidably scale-dependent in both time and space, and there is often a mismatch between the scales at which individual fishing ‘events’ occur (e.g. hours-days and metres-kilometres) and the scales at which the ecological consequences become manifest in ways that are relevant for management (e.g. whole ecosystems and years-decades). At these longer and larger scales the assessed ‘events’ are certain and multiple, such that risk is a function not of their individual likelihood and consequence but of their cumulative impact, for which an alternate ERA approach is more ideally suited (below).

The ‘exposure-effects’ approach to risk assessment is designed to address risks arising from cumulative exposure to influences that are measurable and ongoing (see US EPA, 1992, 1998). In the context of ERAs addressing the effects of ongoing activities such as fishing, ‘exposure’ refers to the total level of impact arising from the activity (e.g. numbers of bycatch species killed) and ‘effect’ refers to an ecological consequence of that exposure (e.g. population decline, disruption of ecosystem function). ‘Risk’ is then the sum of all such effects, or in a probabilistic sense the sum of all possible effects multiplied by their probability of occurrence (see Kaplan 1997). Where impact levels are not known or readily observable, this implies a two-stage process: first an impact assessment to estimate total cumulative impact, and second a risk assessment to evaluate the associated ecological risk (Sharp et al. in press). Note however that ecological systems are subject to negative and positive feedbacks (i.e. resilience, or disturbance thresholds) at different levels of impact, and to interactions with other variables acting at a range of spatial and temporal scales, such that the relationship between cumulative impact and total risk is rarely linear, and often unpredictably complex. The latter stage of ERA thus requires knowledge of the underlying ecology.

Goal and scope of the NPOA seabird fisheries risk assessment process

The New Zealand NPOA seabird fisheries risk assessment framework (hereafter RA framework) is an application of the ‘exposure-effects’ approach, above. See Appendix 1 for precise definitions of the terminology used in the RA framework.

The overall goal of the RA framework is *to evaluate the level of risk to New Zealand seabird populations arising from incidental mortality associated with New Zealand fisheries.* The risk assessment does not address possible indirect fisheries-related impacts, e.g. trophic effects.

The scope and nature of the RA framework was chosen with explicit reference to the information needs of fisheries managers charged with managing seabird impacts by New Zealand fisheries, and also with reference to the known availability of data and ecological knowledge for input into the risk assessment process. Risk assessment frameworks that carefully consider the management context in the design stage are likely to be more effective than generic templates applied universally for different kinds of threats and for a wide range of management applications (such as the templates described by Hobday et al. 2007).

Data availability and the needs of fisheries managers drove the following decisions:

- The fundamental unit at which risk is assessed is *per seabird species*. To meet their obligations under legislation fisheries managers must assess risk to seabirds with reference to units that are biologically meaningful. Only subsequently does it make sense to disaggregate and assign the risk to particular fisheries or areas. Assessment frameworks that assign risk on the basis of administrative categories but do not relate these to total risk at the species level (e.g. Campbell and Gallagher 2007) are inadequate for this purpose.

- The RA framework *can be applied to every species of New Zealand seabird.* Managers have an obligation to manage risks to all species, not only those for which data is readily available. The risk assessment process for rare and data-poor species will rely to a greater degree on expert knowledge and the strategic use of proxy data from comparable species (as in Smith et al. 2007), with associated increases in uncertainty, but potentially all species can be assessed. See Table 1. [Note hereafter ‘species’ refers to a reproductively and/or spatially distinct population of seabirds either breeding in, or resident for a substantial amount of time in, the New Zealand Exclusive Economic Zone (EEZ)]. Examples of spatially distinct populations include Northern and Southern Buller’s albatrosses which are classified taxonomically as one species but breed at distinct locations (BirdLife 2009). An example of a resident species is the wandering albatross which does not breed in New Zealand, but passes through the EEZ during its' annual migration.

- The assessment assigns risk to each species in an *absolute* sense, i.e. species are not merely ranked relative to one another (e.g. as in the PSA approach; Hobday et al. 2007, Waugh et al. 2008). An absolute- as opposed to relative-risk score is required by managers to track change over time. Managers will therefore be able to monitor the performance of different management options (e.g. mitigation) by specific fisheries. Change in risk scores can then be analysed in relation to the management actions implemented.

- Impact and risk is assessed with reference to the *New Zealand population* of the species in question. NZ fisheries managers are responsible for safeguarding the status of seabirds in New Zealand's EEZ. International Conventions such as the Agreement on the Conservation of Albatrosses and Petrels, or the Bonn Convention require New Zealand to mitigate risk to seabird species both within the EEZ, and for New Zealand fisheries outside the EEZ.

- The assessment deals primarily with risks *from New Zealand fisheries*. Risks from other sources, e.g. non-New Zealand fisheries mortality outside the EEZ, or non-fishery threats, are beyond the mandate of NZ fisheries managers, and are considered separately (see 'absolute risk' vs. 'complete risk' in Appendix 1).

- All fishing effort is classified into one of sixteen distinct *fishery groups* within which gear configuration and fishing strategy is assumed to be sufficiently consistent that impact estimates can be applied uniformly to all effort in that group. See Table 2. Impacts are thus calculated separately *per species* fishery group*, then summed across fishery groups to yield total impact *per species*.

- The fundamental metric by which impact is expressed is 'mortality', i.e. *seabirds killed by New Zealand fishing effort annually, expressed a proportion of the New Zealand population*. It is important that impact be expressed in a metric that is *quantitative, measurable, and objectively scalable between fisheries* using different methods, so that risk at a species level can be disaggregated and assigned to different fisheries or areas based on proportional impact. This allows managers to identify trouble spots and target management interventions effectively, to track location- or fishery-specific change over time, and to fairly assign responsibility for required changes to fishing practices. It also provides tangible incentive for the adoption of mitigation to reduce impact on a location- or fishery-specific basis.

- The impact assessment stage *does not rely on the existence of universal or representative fisheries observer data* to estimate seabird mortality. Although fisheries data within New Zealand is generally of a high quality, information on non-fish captures are generally insufficient for the purpose of defining species level impacts, except in particular areas or fisheries where they have been relatively well observed. The framework is flexible in order to allow inclusion of accurate estimates of incidental catch for particular species and fisheries where these are available.

- The risk assessment stage *does not rely heavily on NZ-specific population models or ecological studies* which are currently unavailable for the great majority of species.

- Species-level risk is defined as a function of total impact relative to an established population parameter (R_{max}) with reference to an existing assessment tool. An approached analogous to that developed in the Potential Biological Removals (PBR) models is used (Wade 1998; Niel and Lebreton 2005).

- *The [risk score] calculation for each species is guided by a transparent algorithm* without resort to subjective interpretation, *and is quantitative*, enabling managers to utilise a consistent decision framework under which risk scores are generated. This also allows managers to track changing risk over time.

- The RA framework is designed to *readily incorporate new information.* Assumptions in the impact assessment stage are transparent and testable; as new data becomes available the consequences for the subsequent impact and risk calculations arise logically from the rules of the framework *without the need to revisit other assumptions or repeat the entire risk assessment process*, which would otherwise constitute a major and cost-prohibitive institutional burden to managers.

METHODS

The RA framework was initially devised by means of an expert workshop attended by experts with specialist knowledge of New Zealand fisheries, seabird-fishery interactions, seabird biology, population modelling, and ecological risk assessment. These experts included fishery observers and fisheries managers as well as scientists. The framework was further developed by New Zealand government scientists and fisheries managers in iterative consultation with stakeholder representatives. A diagrammatic representation of this framework is set out in Figure 1.

Risk assessments under the RA framework are implemented via a series of sequential and nested processes as described below, numbered in relation to sub-sections of the diagram in Figure 1.

The **impact assessment (1)** is the process by which total mortality is estimated for each species. It relies on both direct estimation of impacts, e.g. using fisheries observer data, and the spatial overlap approach, below, to estimate total mortality at a species level. See Figure 1.

The **spatial overlap approach (2)** is a critical component of the impact assessment, devised to overcome the lack of adequate incidental catch data in many fisheries and for many bird species. Mortality is estimated as a function of the spatial overlap between the bird species distribution and the distribution of fishing effort.

The **vulnerability calibration (3)** is a process by which the results of the spatial overlap and estimates of capture from fisheries observers are used to model the rate of capture per unit fishing effort for different seabird species and different fishing methods as a function of the density of birds, expressed by the species *vulnerability* term in Figure 1. See Figure 2.

The **risk assessment (4)** incorporates population parameters for the bird species in question, and generates two outputs for managers: an *absolute risk* score representing the extent to which the New Zealand fisheries impact calculated by the impact assessment stage constitutes a risk to the bird species, and a *complete risk* score incorporating information indicative of threats other than New Zealand fishing effort, e.g. terrestrial threats or out-of-zone fisheries mortality. Management objectives are included in the calculation of both the *absolute risk* score and *complete risk* score. See Figure 1.

Detailed descriptions of data sources for each input and of the technical assumptions implicit in the design of the risk assessment adopted by the RA workshop can be found in the **Annex,** below.

1) Impact Assessment

The primary challenge of a New Zealand seabird risk assessment is to generate credible estimates of impact – i.e. numbers of seabirds killed by NZ fisheries – despite a relative paucity of data. Data available for this task mainly derives from fisheries observers. However due to the sheer size of the New Zealand EEZ and the high abundance and diversity of seabirds occurring there, observer data for individual bird species are generally inadequate for one or more of the following reasons:

i) *insufficient data* (i.e. not enough observer coverage);
ii) *unrepresentative data* (i.e. highly concentrated in particular areas/fisheries that may not be indicative of impacts on a larger scale);
iii) *incomplete data* (i.e. some areas/fisheries are not observed at all); or
iv) potentially *inaccurate* data (e.g. because the more rarely caught bird species may not be correctly identified).

This means that observer data cannot be used in isolation or in a straightforward way to generate impact estimates for any bird species at the scale of the whole New Zealand EEZ for all New Zealand fisheries. The main task of the risk assessment is thus to find ways to utilise the observer data in areas or fisheries where it is thought to be adequate, and to estimate impacts using other means in areas or fisheries where observer data is inadequate. The NZ NPOA seabird fisheries risk assessment workshop devised the following sequence to accomplish this task.

For each combination of *species*fishery group:*

1.1 Identify areas or fisheries for which observer coverage is adequate.

Some such areas were identified by the workshop, and additional areas were identified by examination of the data and application of simple decision rules. For example at the workshop the level of observer coverage of 5-10% was discussed as a threshold for which data might be considered appropriate for estimating captures of single, but abundant species, to generate appropriate estimates of captures in a particular fishery or area. Representativeness of observer coverage was noted as an important requirement. Expert knowledge was therefore considered necessary in order to define where such datasets can be used.

The capture estimate input can also be taken from independent sources, such as via estimation of incidental mortality (by either model-based or ratio-estimation procedures). Alternatively they can be generated from within the impact assessment.

1.2 Where observer coverage is considered adequate, use observer data to estimate impacts directly

Note however that at this stage the estimates are of *observable captures*, not actual kills. Bird kills may also arise from interactions with fisheries that are not observable, termed *cryptic kill* (see below). Observable captures also includes birds that are captured and released alive, although the fate of these birds in terms of their long-term survival is poorly understood.

1.3 Where observer coverage is inadequate, derive impact estimates by application of the ***spatial overlap*** *approach and* ***vulnerability calibration*** *(processes 2 and 3, below).*

This approach involves estimating the probability of a bird of a particular species encountering fishing effort of a particular fishery group (i.e. *spatial overlap*) multiplied by the likelihood of that bird being killed in an encounter (i.e. *vulnerability*). *Spatial overlap* is determined by overlaying fishing effort and seabird distributions, whereas *vulnerability* is estimated by a calibration exercise comparing *observed captures* and *bird density* where observer data is considered to be adequate for this purpose See **2) spatial overlap** and **3) vulnerability calibration**, below.

1.4 For species for which both the direct observation and spatial overlap approaches have been used, combine the outputs of steps 1.2 and 2.4.

This yields total *estimated observable captures* for each *species* fishery group.* Incorporating the *population size* estimate is necessary at this stage because the units of step 1.2 and steps 2.1-2.4 and 1.5 need to be comparable (i.e. estimated captures expressed as a proportion of the total population).

The relationship between observable captures (C), spatial overlap (S), vulnerability (V), and population size (N) can be expressed:

$$C = SVN$$

Note that it is also possible to estimate *observable captures* using the spatial overlap approach alone, such that estimates of population size are unnecessary except for use in the calibration (step 3.1). Where population size estimates are unavailable, the *spatial overlap* term for a particular bird species represents a probability distribution of the location of any individual bird over its entire range (rather than a density distribution of all birds) multiplied by the density of fishing effort. Multiplying by the *vulnerability* term V then yields probability of capture per bird (C/N) without the need for a population estimate. Algebraically,

$$C/N = SV$$

(Note that vulnerability (V) is estimated at a species-group level rather than a species level, so its value can often be inferred by proxy rather than estimated directly; see below).

Note also that for species for which adequate observer data exists to utilise the direct observation approach in some areas, such that results from both the direct observation approach and the spatial overlap approach are combined in step 1.4, the observer-derived estimates are retained and flagged as to their origin throughout the impact assessment process.

1.5 Modify estimates in step 1.4 to include 'cryptic mortality' i.e. the proportion of birds killed but not recorded as 'captures' by fisheries observers

The impact metric reported by observers, *observed captures*, includes birds captured and released alive and possibly unharmed, but excludes birds fatally injured by their interaction with the fishing gear but not actually captured by fishers or observers. The latter category is termed '*cryptic kill*'. *Cryptic kill* is potentially a major source of seabird mortality but its estimation relies on a complex series of calculations requiring assumptions subject to considerable uncertainty. Reducing this uncertainty with new research or monitoring is a high priority. See the **Annex** for technical details pertaining to the estimation of *cryptic kill*.

1.6 Sum the estimates from step 1.5 across all fishery groups to yield total fisheries-related mortality at a species level. This is the per-species impact metric

.

This term refers to *total annual fisheries-related kills as a proportion of the total population*, not merely expressed as a number. This means that for bird species relying on observed capture data for part of the impact calculation (i.e. the direct observation approach, steps 1.1-1.2, or as part of the vulnerability calibration, step 3.2), this calculation requires an estimate of *total population size*. Note however that for species where the spatial overlap approach is used exclusively, and the *vulnerability* term is derived by comparison with other species rather than by calibration, then the *total fisheries-induced mortality* term can be estimated without reference to population size (see 2.2, below). This is one of the strengths of the spatial overlap approach.

2) Spatial overlap approach

The spatial overlap approach is a subset of the **impact assessment** process **(1)** described above. It was devised to overcome the inadequacy of the available fisheries observer data for estimating seabird kills in most areas and for most fisheries. The workshop ultimately decided that there was no seabird species in New Zealand for which observer data was adequate across all New Zealand fishing methods and areas. This means that obtaining an *absolute risk* score at the species level will currently require the use of the spatial overlap approach in at least some areas for every bird species. Where both paths are used the *observable capture* estimates from each are calculated separately and subsequently summed to yield estimates of impact.

Spatial overlap is essentially a way of estimating the relative probability or frequency of encounter between birds of a particular species and fishing effort of a particular fishery group. Combined with the **vulnerability calibration** (**3**, described below), the **spatial overlap** approach allows for seabird *mortality* estimation even in areas where observer data is inadequate or unavailable. The spatial overlap proceeds by the following steps:

For each fishery group:

2.1   Map the spatial distributions of fishing effort, by fishery group.

These data are recorded by fishers and are readily available from MFish databases. Most effort data are spatially precise, recorded as latitude/longitude coordinates; some effort (mostly inshore fisheries and smaller vessels) is recorded within a statistical area. Units of effort vary by fishing method (e.g. numbers of hooks for line fisheries, hours for trawl, net length for set nets).

2.2 Map the spatial distribution of the bird species

Seabird spatial distribution data layers essentially represent a *probability distribution per individual bird* of the species in question. I.e. the value of all map pixels sums to 1, and the value in any single pixel represents the probability that a single bird selected at random from the species population will occur in that cell at any given time. Multiplying this layer by a population estimate for the species in question transforms the layer into an actual *density distribution*, i.e. representing numbers of birds in each pixel. This is necessary in the empirical *vulnerability* calibration (step 3.1) but not for species for which *vulnerability* is estimated by other means (e.g. step 3.5).

Seabird distributions have been estimated from a variety of data sources, including sightings, electronic tracking, known ecological affinities and other published sources. These have been compiled and summarized as NABIS distribution layers (freely available on www.nabis.govt.nz) for 39 species. Distributions derived from electronic tracking studies are available for eight species from BirdLife International with additional species layers still in production. Binary distributions (i.e. maximum ranges only) are not used. New distribution layers will be generated through time as necessary to inform risk assessments for which data is currently unavailable. The effect of using differently derived distribution information, and the weightings given to each density layer are to be tested as part of the sensitivity testing for the methodology. This will assess how much weight the assumptions about species density within its documented range affects the outcome of the analyses.

A significant limitation of existing seabird distribution layers is that they do not represent seasonal variation associated with life cycle patterns such as breeding and nesting. Where both fishing effort and seabird distributions are in reality highly seasonal this is a likely source of error, requiring careful consideration in subsequent steps (see step 3.4, below).

2.3 Multiply the fishing effort distribution and the bird distribution layers

Multiplying these two density distributions yields the *spatial overlap* metric. Conceptually this layer represents the probability or frequency of encounter between a bird of a particular species and fishing effort of a particular fishery group

2.4 Multiply the spatial overlap metric (from 2.3) by the vulnerability term (from the ***vulnerability calibration*** *process, below)*

The *vulnerability* term represents the probability that a particular bird encountering fishing effort of a particular fishery group will be 'captured' (i.e. corresponding to reporting protocols for 'observed capture'), despite the fact that no actual observers are required. It is defined this way to ensure compatibility with observer data in the calibration process, and consistency with estimates derived from the direct observation approach (step 1.2). See the **vulnerability calibration** process, below, and Figure 2.

3) Vulnerability calibration

The **vulnerability calibration** process is the primary innovation of the risk assessment framework, devised to allow the direct use of fisheries observer data to derive empirical estimates of impact in areas where capture data are sufficient for this purpose, while retaining the universal applicability of the **spatial overlap** approach in areas where observer coverage is lacking and seabird distribution data are available. The calibration approach essentially combines the methods adopted by two previous partial risk assessments for New Zealand seabirds and in so doing overcomes the limits of both. Previous use of fisheries observer data in isolation (Baird and Gilbert unpublished) relied on the use of observer data applied outside of areas of reasonable inference. In contrast, previous use of a similar spatial overlap method (Waugh et al. 2008) was more universally applicable but lacked reference to quantitative impact data with which to derive species *vulnerability*, and produced only a relative risk ranking between species. The **vulnerability calibration** combines these two approaches to generate species mortality estimates even from patchy or non-existent observer data, as follows:

For each combination of *species*fishery group:*

3.1 Multiply the spatial overlap map (step 2.3 above) by an estimate of bird population size.

This operation transforms the probability distribution per individual bird into an actual density distribution for that bird species, i.e. when multiplied by the density of effort the units are expressed in units of: (effort)*(birds)*unit area^{-2}

3.2 Define species groups assumed to have internally consistent vulnerability characteristics (i.e. on the basis of physiological and behavioural similarities)

Rare bird species, or species that occur primarily in areas where fisheries are only poorly observed, are captured too infrequently to permit the estimation of capture rates using fisheries observer data. To achieve sufficient data, and to ensure that vulnerability can be estimated even for rare species, seabird species were assigned to groups on the basis of assumed similar vulnerabilities to capture by fishing effort. Eleven species groups have been defined for this purpose, as follows: 1) gannets; 2) gulls and terns; 3) large albatrosses; 4) large *Pterodroma* petrels; 5) mollymawks & small albatrosses; 6) other species; 7) penguins; 8) *Procellaria* petrels; 9) shags; 10) shearwaters; 11) small shearwaters. Species group affinity is indicated in Table 1.

3.3 Sum species-specific spatial overlap layers by species group, for each fishery group

Density overlap layers for all bird species in a species group are summed for each fishery group; the sum layer represents the estimated density of interaction (i.e. numbers of encounters) between birds, by species group, and fishing effort, by fishery group.

3.4 Select observed capture data for those fishery groups that are sufficiently well observed to provide reliable estimates of capture rates by species group, and combine observed capture data by species group

Data from fishing groups which have a small number of samples of observed sets may be unreliable, due to the rarity of bird capture events in absolute terms. Therefore, data from fishery groups with low numbers of observed fishing events are excluded from the calibration on the assumption that such a low level of observer coverage is insufficient to reliably estimate capture rates. This threshold is explored in analyses.

*3.5 Model capture rates to derive a vulnerability term V and populate a vulnerability matrix for all species group * fishery group combinations for which sufficient observer data is available.*

The *vulnerability* term *V* represents the probability that a particular bird encountering fishing effort of a particular fishery group will be captured. It is obtained by comparing actual numbers of observed captures for fisheries with adequate data (selected in step 3.4) with the estimated density of interaction between birds and fishing effort (i.e. bird density multiplied by fishing effort density, as represented by the output of step 3.3) in corresponding locations.

3.6 *Populate the remaining cells of the vulnerability matrix by subjective interpolation with reference to V values calculated in step 3.5*

The initial subjective assignment of vulnerability scores to species groups was done in the original RA workshop by experts with knowledge of bird biology and fishery-seabird interactions. These were assigned in a relative sense only (see Table 3) and subsequently should be compared with results from the calibration (from step 3.5).

Note that the calculation of the *vulnerability* term relies on the assumption that V is constant in space. Because *vulnerability* is a product of the behavioural and physiological characteristic of the bird (e.g. feeding behaviour, wing fragility) this is a reasonable assumption. Where this assumption may break down is where spatial variability arises from seasonal variability, i.e. due to seasonal bird movements or changes in behaviour related to life cycle patterns (e.g. nesting).

The assumption of constant vulnerability in space includes the assumption of constant vulnerability in time, i.e. there is no seasonal variation in the vulnerability of a particular species group to fishing effort of a particular fishery group. However it is to be expected that seasonal effects exist and will confound the analysis to some extent: the assumption of constant vulnerability is an approximation, not a hypothesis. Where seasonal effects are thought to produce unreliable results it may be productive to constrain the effort data temporally or to re-visit the assumption of constant vulnerability by assigning different vulnerabilities to different areas (e.g. breeding vs. non-breeding areas).

4) Risk assessment: From impact to risk

The output of the final impact assessment step 1.6 is the *total fisheries-related mortality* M, represented as a *proportion of the New Zealand population killed by New Zealand fishing effort annually*. M is the impact metric. Moving from impact to risk is the fundamental basis of risk assessment. For ERA this is also the step requiring knowledge of the underlying ecological processes affecting shape of the relationship between impact and risk. However for management purposes New Zealand requires a framework that works for every seabird species and that can be applied without reference to species-specific population models and without repeated access to specialist biological knowledge. For this reason the RA framework utilises generic population biology metrics that are available from published literature for many seabird species, and for which international precedent provides guidance as to their use in the estimation of species-level risk. See the **Annex,** below, for relevant background and details of the derivation of the risk score equation.

The risk assessment step proceeds as follows for each species:

4.1 *Divide the fisheries-related mortality M by the productivity term R_{max} to yield the impact ratio M/R_{max}*

The R_{max} term represents the maximum theoretical productivity (growth) of an unconstrained population under ideal conditions (i.e. the slope of the population growth curve at the origin) (Sibly & Hone 2003). The Impact ratio M/R_{max} thus represents the proportion of maximum theoretical population productivity that is being appropriated by fisheries mortality. R_{max} estimates are available or can be calculated or inferred from available literature for most species. See Table 1, and Waugh and Filippi (2009c).

4.2 *Select an F value indicative of species population and threat status, for use in the final risk score equation.* $0 < F \leq 1$.

The F term is a scale-less ‘management factor’ underpinned by management targets, used to modify the output of the subsequent risk score estimation with reference to species population status. F ranges 0 to 1; the extent to which F is lower than 1 reflects the proportion of population productivity that is available for population growth, or for sources of non-natural mortality other than fisheries impact (NMFS 2005) such that high F values are used for stable or increasing populations at or near carrying capacity, whereas low F values are used for threatened, declining or depleted populations for which management goals include rapid population recovery.

Decision rules have been suggested to assist in the selection of F values with reference to various population status indicators. See Walker et al. (2009). Corresponding F values previously derived and used for New Zealand seabirds, and examples of published F values used or recommended elsewhere are included in Waugh and Filippi (2009b). The Seabird Stakeholder Advisory group (SSAG) has not completed discussing the selection F values as yet.

4.3 Calculate the [absolute risk] score as a function of the Impact ratio M/R_{max} and the selected F value

[Absolute risk] = 2M / F* $\mathbf{R_{max}}$ where $0.1 < F \leq 1$

[Risk score] = 1 ➔ maximum acceptable risk

The derivation of the risk score formula was selected with reference to the PBR approach (Wade 1998). It can be demonstrated algebraically that the [risk score] equation above and the PBR equation from published literature are functionally interchangeable (see Annex).

The risk score is the main output of the risk assessment process. Managers could interpret the risk score with the aid of a decision support framework under which scores within pre-defined ranges trigger associated management responses (e.g. Figure 3).

4.4 *Selecting an alternate F value (F_c) incorporating threats from influences other than NZ fisheries, for the calculation of [complete risk]*

While it is beyond the mandate and power of NZ fisheries managers to manage threats other than those arising from fishing, managers will nonetheless need to consider their own decisions in the context of the complete set of risks affecting New Zealand seabirds.

Decision rules for the downward adjustment of the previously selected F value to accommodate non-fishery threats are proposed in Walker et al. (2009). Where such threats exist, the complete risk score will be correspondingly higher than the absolute risk score.

4.5 *Use the alternate F value Fc (or direct estimation of non-NZ-fishery mortalities) to calculate [complete risk]*

The [complete risk] calculation is identical to that for [absolute risk], with F_c substituted for F.

An alternate option for the calculation of *complete risk* is to use actual mortality estimates from non-NZ fishing threats, adding these as additional sources of impact in step 1.6 and then simply repeating steps 4.1 – 4.3 (using the resulting *total mortality*, as opposed to total *fisheries-related* mortality calculated previously). This approach is more rigorous than subjectively adjusting F to account for ‘other threats’ in step 4.4., but relies on the existence of adequate data to estimate mortality rates arising from non-NZ-fishery threats. This approach may be feasible applied to international fisheries where data is typically available (e.g. from within the CCAMLR area) or for other direct impacts (e.g. juvenile birds taken by customary harvest). However it is unlikely to work for non-fishery threats which primarily exert negative impacts by affecting habitat or reproductive success (e.g. nest predation by feral animals or nesting habitat limitation).

DISCUSSION

Strengths and weaknesses of the proposed risk assessment framework

The described impact assessment framework provides a useful template that could be productively applied in other areas. The discussion below explores the strengths and weaknesses of the framework.

Consistency, transparency, and testability

The adoption of a consistent impact assessment framework greatly facilitates objective comparisons between fisheries utilising different fishing gears and/or operating in different areas. All risk assessments depend to some degree on the application of expert knowledge and are subject to unavoidable subjectivity, but by utilising expert knowledge within an open and systematic framework, the influence of personal bias is minimised (e.g. Macguire 2004, Kerns and Ager 2007). Furthermore within the framework even subjective estimates are nonetheless quantitative – hence testable and objectively scalable relative to one another – and the rules by which these estimates are combined with available data to produce estimates of total impact and risk at the species level are clearly stated, mathematically logical and defined with reference to existing international precedent.

Ease of modification

The proposed RA framework is deliberately designed to readily incorporate change. As new data becomes available or old assumptions are invalidated the associated numerical estimates can be changed; the consequences for final risk estimation then arise logically as defined by the framework without the need to repeat the entire RA process or re-visit other assumptions (see also Sharp et al. in press).

Disaggregation of risk to fishery level

By necessity fisheries-related risk to seabirds is calculated at the species level. However responsibility for implementing policy changes will fall back to individual fisheries and fishery managers, each of which is responsible for only a subset of total species-level risk. It is important then that managers have the ability to disaggregate species-level risk and examine the individual contributions of particular fisheries or other administrative entities (e.g. areas) to total risk. The ease with which this is accomplished is one of the strengths of the impact-based approach to risk assessment: impacts are estimated quantitatively, and the contributions of particular fisheries or areas to total risk can be assigned proportional to their estimated impact.

Risk disaggregation by fishery or geographic area (or even to the level of particular fishing companies or boats) may be a valuable tool for managers, for example to highlight particular problem areas and focus subsequent risk-reduction efforts. It also provides incentive for individual fisheries to improve their own fishing practices and reduce their own impacts, essential to avoid a 'tragedy of the commons' situation in which the costs of risk-reducing activities is privatised but the benefits are externalized. This is one reason why bird kill estimates derived from direct observation (step 1.2) are retained separately from those derived from the spatial overlap method (steps 2.1-2.5), i.e. to reward individual fisheries for locally improved performance that can be verified using observer coverage.

Note that assigning risk to individual fisheries proportional to their impact does *not* imply that risk itself is additive – impacts are additive but the relationship between impact and risk may be non-linear. But whatever the total risk associated with a given impact, it is reasonable that a particular fisheries' share of that risk be proportional to their share of total impact. Note also that assignment of *responsibility for risk reduction* is a more complex question, potentially requiring consideration of other variables, such as fishery efficiency (i.e. catch or profit per unit impact) or socioeconomic cost-benefit analysis of different risk-reduction options.

Explicit treatment of uncertainty

The treatment of uncertainty in risk assessment is an area fraught with difficulty. One advantage of the quantitative impact-assessment-based RA framework adopted here is that it allows for the rigorous and explicit treatment of uncertainty distinct from risk. In contrast, risk assessments utilising the likelihood-consequence approach often conflate uncertainty and risk on the same scale (e.g. Hobday et al. 2006), such that it becomes impossible to distinguish between a highly certain outcome indicating high risk and a highly uncertain outcome suggesting low to moderate risk. But in management practice the two situations are completely dissimilar: in the former a manager knows there is a problem, requiring action; in the latter it is impossible to know whether or not a problem exists, suggesting the need for better information.

There is uncertainty associated with every input to the RA framework in Figure 1, each of which contributes to uncertainty associated with the [risk score] outputs. The risk assessment process should involve explicit representation of estimated uncertainties for every data input (including subjective estimates) and every calculation. Figure 3 highlights the need to deal with uncertainty in a rigorous and quantitative manner, such that outputs (risk scores) as well as inputs can be expressed as a range or a probability distribution instead of a single number. Where risk scores are expressed as a probability distribution it is likely that the actual shape of that distribution will strongly inform the choice between risk reduction (mitigation) or uncertainty reduction (research) to address uncertain and marginal risk.

Note that where a marginal but uncertain risk output suggests the need for better information, it is possible that the most effective means of reducing that uncertainty may include activities normally outside the mandate of fishers or fisheries managers. For example where uncertainty in the [risk score] arises from poor data relating to species population (N) or population productivity (R_{max}) the most effective means of reducing that uncertainty may be to carry out nesting surveys or biological studies. This example illustrates the need to devise a rigorous means of tracking the propagation of uncertainty through the various data manipulations of the RA framework, such that the relative contributions of uncertainty associated with individual inputs or calculations can be assessed and efforts to reduce uncertainty can be targeted effectively.

Tracking uncertainty through a complex process involving multiple inputs and calculations is a statistically and computationally challenging process. Likely options to address this challenge, e.g. iterative modelling approaches utilising Bayesian statistics to test the implications of input distributions against known output priors, should be actively explored by fisheries managers, but may not be feasible in the short to medium term. A simpler approach would be to approximate the uncertainty by rerunning the RA for the likely range of parameter values to give the range of potential outputs.

Use of risk assessment outputs to inform management

The setting of risk scores through the RA framework allows for the setting of risk reduction targets and monitoring of changes in risk. The following section discusses options for implementation of the risk assessment framework. These and other options will be explored in greater detail in the implementation chapter of the NPOA framework document (in development).

Decision support framework

The results of the risk assessment process will be summarised for managers in two outputs, the *absolute risk* score and the *complete risk* score. The risk scores for all species will be on an identical scale designed to be insensitive to species population size and other biological or demographic variables, enabling objective comparisons between species. The scale is also absolute on a per-species basis, i.e. it reflects the risk to a particular species without reference to others, and can be used to track changing species risk over time.

Use of the risk scores to inform management action will utilise a decision support framework whereby risk scores within pre-defined ranges suggest automatic management responses, e.g. any risk score > 1 may be considered ‘unacceptable risk’, prompting appropriate action to reduce the species risk, but the speed and severity of the management response may be considered greater for a [risk score] > 2 than for a [risk score] between 1.0 and 1.2. See Figure 3.

One advantage of the quantitative and transparent nature of the RA framework is that it allows fishers and managers to consider alternative options to reduce species risk as opposed to prescribed management responses, and the consequences of different management interventions for RA framework outputs arise automatically as defined by the rules of the methodology. With the aid of the RA framework, fishers and managers can then select the most feasible or lowest cost option to reduce species risk.

The majority of likely options to reduce risk will involve

- adoption of altered fishing practice (e.g. offal discard protocols) or gear mitigation devices (e.g. tori lines) to reduce *vulnerability; or*
- changes to the spatial distribution of fishing effort to reduce *spatial overlap*

Where risk scores are highly uncertain such that it is unknown whether [risk score] > or < 1, other attractive options may include:

- the use of increased observer coverage to acquire data to reduce uncertainty
- the design of focused research or monitoring studies to reduce uncertainty (See below).

Absolute risk vs. complete risk

While it is beyond the mandate and power of NZ fisheries managers to manage threats other than those arising from fishing, managers will nonetheless need to consider their own decisions in the context of the complete set of risks affecting New Zealand seabirds. Where [absolute risk] < 1 but [complete risk] > 1 there is still a need for action to reduce complete risk, including action to reduce that portion of the risk attributable to fisheries. However where the gap between [absolute risk] and [complete risk] is great, indicating that the species faces considerable non-fishery threats, a preferred risk-reduction strategy may emphasise actions outside the mandate of the Ministry of Fisheries, e.g. terrestrial conservation or diplomatic efforts to reduce impacts outside the NZ-EEZ, necessitating a whole-of-government approach.

In instances where reduction of complete risk by non-fishery mechanisms is a high priority, it may be that the subjective and binary F term adjustments are too blunt an instrument with which to recognise progress. In these instances the government may wish to instead incorporate non-fishery threats (and out-of-EEZ fishery threats) by incorporating mortality estimates for these other threats in step 1.6. This approach will depend on the existence of appropriate data; these data may exist for some out-of-EEZ fishery threats (e.g. in the CCAMLR area) but are unlikely to be available for terrestrial threats.

Impact mitigation

Irrespective of the risk analysis, the impact assessment alone will allow managers and fishers to compare the relative impacts of different fishing methods and gear configurations, informing the adoption of minimum-impact fishing practices. Transparent and quantitative impact estimates provide tangible incentives for fisher-led innovation in the development of codes of conduct or technical gear modifications to reduce impact further (e.g. Robertson et al. 2006, Hobday et al. 2007, NPFMC 2009). Interim steps within the impact assessment, where bird mortalities are broken down by species, fishing method, geographic area, or fishing gear component (e.g. warp-strikes vs. net captures; see Annex) will provide managers and fishers with valuable information to target mitigation measures most effectively.

Allocation of research and observer resources

The existence of a risk assessment rich with internal detail and with explicit consideration of uncertainty (see above) will be a powerful tool for managers seeking to make optimum use of scarce resources, such as research money or available fisheries observer hours, to improve the impact and risk assessments over time. Managers familiar with the internal design of the assessment can identify which estimates or assumptions are the sources of the greatest uncertainty in the final RA outputs or ideally these can be made explicit in the RA outputs, and target research to improve estimation of the most critical steps. For example, a number of assumptions internal to the estimation of cryptic kill (step 1.5, see Annex) are highly uncertain and likely to exert significant influence on the final estimation of risk. Research to improve these estimates should be a priority.

The explicit use of only a subset of the fisheries observer data (steps 1.1 and 3.4) will yield immediate benefits for the optimal allocation of fisheries observers for whom seabird mortality estimation is a priority. Observer effort could be directed toward those areas where it is already used in the assessment, although these may not require annual updates (step 1.1) or where it is needed to fill remaining gaps in the vulnerability calibration (step 3.4). (e.g. inshore setnet fisheries).

Acknowledgements
We thank all those involved in the review workshop who have also contributed to the framework described in the document; Ed Abraham, Barry Baker, David Bilton, Chris Francis, David Kirby, Darryl MacKenzie, David Middleton, Stephanie Rowe, David Thompson. We also thank David Middleton and Neville Smith for review of the manuscript.

Appendix 1: Glossary

Impact = An effect of human activity on a natural resource. I.e. For the purposes of this assessment: *Impact = birds killed by NZ fishing activity*

Impact assessment = A systematic process for quantifying/ estimating/ assessing/ or allocating impacts.

- I.e. For this assessment: *Impact assessment = estimating the birds killed by fishing* (broken down by species, by fishery, by fishing method, by area, etc.)

Risk = The sum of all ecological consequences of an impact at a given level (or, in a probabilistic sense, the sum of consequences for all possible occurrences multiplied by their frequency of occurrence).

- I.e. for this assessment risk refers to the *consequences of birds killed by fishing* (e.g. range contraction, population decline; failure to meet rebuilding targets)

Risk assessment = A systematic process for estimating and assigning risk. This framework considers risks *to NZ bird species, from NZ fisheries*. Risk will be assessed at the scale of particular bird species – subsequent assignment of risk to particular fisheries or geographic areas is thus a question of *partitioning* species-level risk.

Relative risk = Risk *from NZ fisheries* for each bird species expressed relative to other species (i.e. species are ranked 'highest risk' to 'lowest risk')

Absolute risk =Risk *from NZ fisheries* for each bird species expressed using some absolute metric (i.e. not in reference to other species)

Complete risk = Risk for each bird species *including non-NZ fishery risks* (e.g. from terrestrial threats, climate threats)

Non-fishery risk = *Risks arising from threats other than NZ fisheries* (i.e. the additional threats considered to move from 'absolute risk' to 'complete risk'). Note that for purposes of this definition, *non-fishery risk includes risk from fisheries outside the NZ-EEZ*

Population size = the estimated total number of *adult* birds against which impact will be assessed. In most cases this will likely be all adult members of a species (or subspecies) resident in New Zealand; but where different colonies are known to have distinct and non-overlapping distributions, then distinct populations will be regarded as separate species for purposes of the risk assessment.

Population trend = known or estimated recent change in population size

Population status = current population level relative to historical levels or known carrying capacity (e.g. depleted, not depleted)

Spatial Overlap = a numerical metric expressing *the probability/frequency of encounter between a particular individual bird and a particular fishing method type.*

- Estimated as a function of the overlap of the bird species and the fishing effort distribution
- both species and effort distributions are expressed as kernel densities (not merely binary ranges)
- *Spatial Overlap = [Density distribution of effort] x [Probability distribution of birds]*
- units for Spatial overlap = (effort)(year) $^{-1}$(km) $^{-4}$

Vulnerability = a metric representing *the probability that a particular bird already encountering fishing effort of a particular method type will be captured.* Vulnerability is a function of the physiological and behavioural characteristics of a particular bird species, and of the specific gear configuration and gear deployment characteristics of a particular fishing method type. It is calculated using GLM relating observed captures to the estimated density of birds in corresponding locations.

Productivity = the potential rate of growth/ recovery of a bird species, as a function of its biological and life-cycle characteristics.
- Expressed as a function of various biological metrics. R_{max} was selected

$\mathbf{R_{max}}$ = the maximum per capita instantaneous rate of change (growth) of a species in an unconstrained environment (i.e. the slope of the growth curve at the origin).

PBR = Potential Biological Removals = the maximum fisheries-induced mortality that can be sustained without causing the failure of the management objective (e.g. stable population, or achievement of a rebuild target). The [risk score] equation adopted by this risk assessment framework is functionally interchangeable with the PBR equation from published literature

Fisheries-related mortality (M) = the *proportion* of a *particular bird species killed by NZ fisheries annually*.

Kill = *numbers* of birds killed by NZ fisheries (at any scale; not specific)

Observed capture = numbers of birds sampled by fisheries observers (e.g. because they are injured and fall on the deck of the boat, but are released alive)
- kill is a subset of capture

Estimated observable capture = the estimated number of captures that would be observed if observer coverage were comprehensive and adequate throughout the entire area of spatial overlap (i.e. *observed capture* scaled up to the entire species range, but not adjusted to incorporate *cryptic kill*)

Cryptic kill = birds killed or fatally injured as a result of their interaction with fishing effort but not recordable as 'capture' by fisheries observers even where observers are present (e.g. because they acquire injuries that are not immediately fatal, or do not come into the physical possession of the observers)
- e.g. 'warp strikes' are a likely major source of cryptic kill

Warp strike = forceful contact between a trawl warp and a seabird; studies cited in this risk assessment stipulate that contact must be sufficient to move the bird through the air; wing contacts are only counted if above the wrist.

Surface warp strike = warp strike in which a bird resting or hovering on the surface of the water is overtaken and potentially entangled by a moving warp line.

Aerial warp strike = warp strike in which a flying bird collides with the warp under its own momentum

Net capture = that portion of *observed capture* (for trawl fisheries) consisting of birds entrapped or entangled in the net

References

Abraham, E.R., Thompson, F.N., (in prep) Modelling total seabird bycatch.

Abraham, E.R., Thompson, F.N., (2008). Warp strike in New Zealand trawl fisheries, 2004-05 to 2006-07. Draft Aquatic Environment and Biodiversity Report.

Australian/ New Zealand Standards. 1999. Standard for Risk Management. AS/NZS 4360. Standards Association of Australia, Stratfield, NSW: 46 pp.

Barbraud, C., Delord, K., Marteau, C. & Weimerskirch, H. 2009. Estimates of population size of white-chinned petrels and grey petrels at Kerguelen Islands and sensitivity to fisheries. *Animal Conservation* xxxx: 1-8.

Barbraud C., Marteau C., Ridoux, V., Delord K., Weimerskirch H. 2008. Demographic response of a population of white-chinned petrels *Procellaria aequinoctialis* to climate and longline fishery bycatch. *Journal of Applied Ecology*, 45, 1460-1467.

Baird, S.J., Gilbert, D.J., (in press). Initial assessment of risk posed by commercial fisheries to seabirds breeding in New Zealand waters. Draft Aquatic Environment and Biodiversity Report.

Beer, T. 2006. Ecological Risk Assessment and Quantitative Consequence Analysis. Human and Ecological Risk Assessment, 12: 51-65.

BirdLife International 2009. http://www.birdlife.org. Accessed 12 June 2009.

Brothers, N., Gales, R., Reid, T., (1998). The influence of environmental variables and mitigation measures on seabird catch rates in the Japanese tuna longline fishery within the Australian Fishing Zone, 1991-1995. Biological Conservation **88**; 85-101.

Campbell, M.L., and C. Gallagher. 2007. Assessing the relative effects of fishing on the New Zealand marine environment through risk analysis. ICES Journal of Marine Science, 64: 256-270.

Crawford, C. 2003. Qualitative risk assessment of the effects of shellfish farming on the environment in Tasmania, Australia. Ocean and Coastal Management, 46: 47-58.

Croxall. J.P. 2008. The role of science and advocacy in the conservation of Southern Ocean albatrosses. *Bird Conservation International* 18:1-17.

Dillingham, P.W. & Fletcher, D. 2008. Estimating the ability of birds to sustain additional human-caused mortalities using a simple decision rule and allometric relationships. *Biological Conservation* 141: 1783-1792.

Dillingham, P.W. & Fletcher, D. 2006. Unpublished report to the Ministry of Fisheries.

FAO. 1999. The International Plan of Action for Reducing Incidental Catch of Seabirds in Longline Fisheries. Food and Agriculture Organisation of the United Nations, Rome.

FAO. 2008. Report of the Expert Consultation on Best Practice Technical Guidelines for IPOA/NPOA-Seabirds. Bergen, Norway, 2–5 September 2008. FAO Fisheries and Aquaculture Report No. 880. Food and Agriculture Organisation of the United Nations, Rome.

Fletcher, W.J. 2005. The application of qualitative risk assessment methodology to prioritize issues for fisheries management. ICES Journal of Marine Science, 62: 1576-1587.

Fox, D.R. 2006. Statistical issues in ecological risk assessment. Human and Ecological Risk Assessment, 12: 120-129.

NMFS 2005. Guidelines for preparing stock assessment reports pursuant to Section 117 of the Marine Mammal Protection Act. National Marine Fisheries Service Policy Directive 02-204-01. Available on https://reefshark.nmfs.noaa.gov/f/pds/publicsite/index.cfm

Gales, R., Brothers, N., Reid, T., (1998) Seabird mortality in the Japanese Tuna Longline fishery around Australia, 1988-1995. Biological Conservation **86**; 37-54.

Hayes EH, Landis WG. (2004) Regional ecological risk assessment of a near shore marine environment: Cherry Point, WA. Human and Ecological Risk Assessment 10:299–325.

Hobday, A. J., A. D. M. Smith, H. Webb, R. Daley, S. Wayte, C. Bulman, J. Dowdney, A. Williams, M. Sporcic, J. Dambacher, M. Fuller and T. Walker. 2007. Ecological Risk Assessment for the Effects of Fishing: Methodology. Report R04/1072 for the Australian Fisheries Management Authority, Canberra: 174 pp.

International Union for the Conservation of Nature 2009. www.iucnredlist.org. Accessed 12 June 2009.

Kaplan, S. 1997. The words of risk analysis. Risk Analysis, 17(4) 407-417.

Kerns, B.K. and A. Ager. 2007. Risk assessment for biodiversity conservation planning in Pacific Northwest forests. Forest Ecology and Management, 246: 38-44.

Lewison, R.L. & Crowder, L.B. 2003. Estimating fishery bycatch and effects on a vulnerable seabird population. *Ecological Applications* 13:743-753.

Link JS, Broadziak JKT, Edwards SF, Overholtz WJ, Mountain D, Jossi JW, et al. (2002). Marine ecosystem assessment in a fisheries management context. Canadian Journal of Aquatic Sciences 59: 1429–40.

Macguire, L.A. 2004. What can decision analysis do for invasive species management? Risk Analysis, 24: 859-868.

McCallum, H. 2000. *Population parameters. Estimation for Ecological Models*. Methods in Ecology. Wiley-Blackwell.

Ministry of Fisheries. 2009. Status of New Zealand Fisheries website. http://www.fish.govt.nz/mi-nz/SOF/default.htm. Accessed 12 June 2009.

Niel, C. & Lebreton, J.D. 2005. Using demographic invariants to detect overharvested bird populations from incomplete data. *Conservation Biology* 19: 826-835.

North Pacific Fishery Management Council (NPFMC). 2006. Multi-criteria decision tool to evaluate proposals for change in Steller sea lion protection measures in the Gulf of Alaska and Bering Sea/Aleutian Islands groundfish fisheries. Stellar Sea Lion Mitigation Committee, North Pacific Fishery Management Council, Anchorage, AK.

Robertson, G. M. McNeill, N. Smith, B. Wienecke, S. Candy and F. Olivier. 2006. Fast sinking (integrated weight) longlines reduce mortality of white-chinned petrels (Procellaria aequinoctialis) and sooty shearwaters (Puffinus griseus) in demersal longline fisheries. Biological Conservation, 132: 458-471.

Sharp, B.R., S.J. Parker and N. Smith. 2009. An impact assessment framework for bottom fishing methods in the CCAMLR area. *CCAMLR Science*, in press.

Sibly, R.M. & Hone, J. 2003. Population growth rate and its determinants: an overview. Pp 11 – 40. In Sibly, R.M., Hone, J. and Clutton-Brock. (eds). *Wildlife Population Growth Rates.* Cambridge University Press, Cambridge. Pp 11 – 40.

Small, C. 2005. Regional Fisheries Management Organisations Their duties and performance in reducing bycatch of albatrosses and other species. BirdLife International. Cambridge.

Smith, A.D.M., E.J. Fulton, A.J. Hobday, D.C. Smith, and P. Shoulder. 2007. Scientific tools to support the practical implementation of ecosystem-based fisheries management. ICES Journal of Marine Science, 64: 633-639.

Standards New Zealand and Standards Australia. (2006) Environmental risk management- Principles and process. Standards Australia/ Standards New Zealand. Wellington: Standards New Zealand.

Sullivan, B.J., P. Brickle, T.A. Reid, D.G. Bone and D.A.J. Middleton. 2006. Mitigation of seabird mortality on factory trawlers: trials of three devices to reduce warp cable strikes. Polar Biology, xxxx.

US EPA, 1992. A framework for ecological risk assessment. Document EPA/630/R-92/001. Risk Assessment Forum, U.S. Environmental Protection Agency, Washington, DC.

US EPA, 1998. Guidelines for ecological risk assessment. Document EPA/630/R-95/002F. Risk Assessment Forum, U.S. Environmental Protection Agency, Washington, DC.

Wade, P.R. (1998). Calculating limits to the allowable human-caused mortality of cetaceans and pinnipeds. *Marine Mammal Science* 14:1-37.

Watkins, B.P., Peterson, S.L., Ryan, P.G., (2008). Interactions between seabirds and deep-water hake trawl gear: an assessment of impacts in South African waters. *Animal Conservation* **11;** 247–254.

Walker, N., Waugh, S., Sharp, B. (2009) Seabird risk assessment ‘Interim’ F values. *SSAG 29 Technical meeting, May 2009.*

Waugh, S., Fillipi, D., Walker, N.A., Kirby, D.S., (2008). Updated preliminary results of an ecological risk assessment for seabirds and marine mammals with risk of fisheries interactions. Report to the Scientific Committee for CCAMLR, WG-FSA-08/51.

Waugh, S., Fillipi, D. (2009). Progress Report: Ecological Risk Assessment for Seabirds in New Zealand fisheries -- Calculating Vulnerability. *SSAG 29 Technical meeting, May 2009.*

Waugh, S., Fillipi, D. (2009b). Progress Report: Ecological Risk Assessment for Seabirds in New Zealand fisheries **--** Choosing F**.** *SSAG technical meeting***,** *23 April 2009*

Waugh, S., Fillipi, D. (2009c). Progress Report: Ecological Risk Assessment for Seabirds in New Zealand fisheries -- Calculating Rmax. *SSAG 29 Technical meeting, May 2009.*

ANNEX

Technical details re: inputs and calculations internal to the RA framework

Step 1.5 - Cryptic kill
This term refers to birds that are likely to die as a result of their interaction with fishing gear but that (even where fishing effort is observed) will not be recorded as 'captures', because they are not recovered onboard the fishing vessel. As presently defined, *observed capture* refers to any bird (or portion of a whole bird) that that comes into the physical possession of the observers or fishers onboard the ship. *Observed capture* therefore includes birds that become entangled by fishing gear but are subsequently released alive, but excludes other potentially fatal interactions such as warp strikes. *Observed capture* also excludes deck strikes and any seabirds recovered in a state of decomposition that indicates that the fishing event wasn't the cause of death. Resolving these issues is essential if observer data is used to derive estimates of total fisheries-induced mortality.

Longline fisheries
The potential for cryptic kill in longline fisheries arises from i) the potential for hooked birds to fall off the line prior to or during line retrieval and ii) undetected actions to undermine observation effectiveness (i.e. deliberate covert discards).

Most experts at the workshop thought that the number of hooked birds that were subsequently lost and not counted as observed captures by longline fisheries was likely to be low. The workshop agreed that the only available source of data to inform this calculation would come from overseas literature, and that MFish should build in an appropriate adjustment factor after consulting the relevant publications (Brothers et al. 1998, Gales 1998). The workshop agreed that deliberate discarding by fishers was difficult to assess, but that the required correction factor could perhaps be inferred from these same publications, and in any event was likely to be small relative to other sources of uncertainty.

A potential problem was identified whereby inconsistent data recording protocols for bird capture in longline fisheries may produce misleading results. For surface longline and trawl fisheries all fishing effort (and therefore all seabird capture) is observed on a particular vessel for an entire voyage. However in bottom longline fisheries there is an observation target of 45% of hooks observed; birds captured outside the observed period are not recorded by the observer deck log but will nonetheless be retained for necropsy and recorded as 'non-fish bycatch' by the vessel. A discrepancy may arise if the birds captured off-shift are subsequently recorded as *observed captures*, in which case scaling up from observed effort to total effort will overestimate seabird kill for bottom longline fisheries. The workshop agreed that bottom longline data should be interpreted with caution, that there was potential to resolve this issue by careful comparison of observer deck logs against vessel bycatch reporting, and that data recording protocols for bottom longline fisheries should be cleaned up to avoid potential misinterpretation in future.

Trawl fisheries
The majority of discussions of cryptic kill were devoted to warp strikes in trawl fisheries. This is potentially a major source of cryptic kill, but one for which there is very little data available. The necessary correction factors rely heavily on subjective assessment, expert

knowledge, and in some instances outright guesswork. Cryptic kill in trawl fisheries may therefore be a major source of uncertainty in subsequent mortality estimates, and in the risk assessment process more generally. New research to reduce this uncertainty is a high priority.

The only source of data on warp strike rates in New Zealand fisheries is Abraham & Thompson (2008). This research estimated the following warp strike rates for albatrosses and for petrels, mainly associated with squid fisheries:

- albatrosses: 200 warp strikes: 1 observed capture
- petrels: 6000 warp strikes: 1 observed capture

'Warp strike' in this study is defined as any heavy contact between the bird and the warp sufficient to deflect the bird's flight trajectory; wing contacts were only included if above the wrist. No attempt was made to assess the degree of damage to the birds arising from this contact.

The workshop acknowledged that without some estimate of what proportion of warp strikes are ultimately fatal, these numbers were a tremendous source of uncertainty, i.e. implying a multiplicative correction factor ranging from as low as 1 (i.e. warp strikes never fatal) to as high as 200x for albatrosses and 6000x for petrels (i.e. all warp strikes fatal). Considerable discussion was devoted to describing the nature of warp strike contacts and the observed responses of the birds. It was agreed that warp strike as currently defined actually refers to two different kinds of contact, such that future discussions should distinguish between three different categories of contact:

- *'surface warp strike'*: birds resting or hovering on the surface of the water are overtaken and potentially entangled by a moving warp line, or struck by warp movement arising from the movement of a vessel
- *'aerial warp strike'*: flying birds collide with the warp
- *'net capture'*: birds become entrapped or entangled in the net

The following observations were deemed relevant to the estimation of outcomes for each category of contact:

Surface warp strike:

- petrels tend to sit on the surface of the water, whereas albatrosses may sit on the water with their wings raised. Consequently petrels will be more robust to impact, i.e. less likely to be entangled or fatally injured.
- smaller birds such as petrels are unlikely to become entangled on the warp so will almost never be counted among 'observed captures'.
- an albatross hit in the back or the wing will tend to get caught with the warp under its armpit and its wing wrapped around the warp, and will be subsequently pulled underwater by the force of the warp moving through the water.
- albatrosses pulled underwater will either fall off, encounter a sprag and be impaled on it, or be pulled all the way to the trawl door (800-900 m) where they may be subsequently retrieved (and counted among observed captures).
- Watkins et al (2008) recorded ***15** fatal surface warp strikes* (not total warp strikes) per observed capture for white capped albatrosses in South Africa, with 'fatal' defined as birds pulled underwater that failed to surface again .

Aerial warp strike:

- albatrosses are highly susceptible to wing damage from aerial warp strikes, and any tendon damage to the wing is likely to lead to eventual death.

- opinions varied on the extent to which petrels were also susceptible. Some experts felt that petrels would be more robust due to their more compact form, shorter wings, and lower body weight. Others disagreed.

Net capture:

- birds can be caught inside the net or entangled on the outside.
- birds can be caught either on shooting the net or on hauling, with the majority caught on hauling
- cryptic net kills arise from i) birds caught internally and subsequently lost through the slack meshes during the haul; and ii) birds entangled externally subsequently falling off the net
- mitigation has reduced the number of observed captures attributable to warp strikes
- it was thought that the majority of birds killed by the net are retained and counted among 'observed captures' such that the necessary multiplier from observed net captures to total net kills is small. A correction factor of 1.5 was proposed.

Clearly any meaningful discussion of total kill must distinguish between these different sources of kill. At present, observer reporting protocols do specify the source of kill for observed captures, but available warp strike data does not distinguish between the two categories of warp strike. *Amending the data protocols and designing research to properly estimate rates for these three categories of contact is a high priority.*

In the absence of any quantitative information, workshop experts attempted to estimate relative frequencies of occurrence for the different warp strike categories. It was generally accepted that surface warp strikes (i.e. contact on the water) were more common than aerial warp strikes; *the estimated ratio of surface: aerial warp strikes varied from **2:1** to **10:1.*** Resolving this ratio properly and in a species-specific manner is a high priority.

Biologists present at the workshop were asked to estimate what proportion of warp strikes in each category is ultimately fatal, for two general classes of birds (petrels and albatrosses). See Table 4. There was some discussion about how the estimates in Table 2 would ultimately be used, reflecting concerns that 'observed captures' is an inappropriate multiplier because it is already confounded by other unknowns. The workshop agreed that any use of the observed captures data to estimate *total kill* would have to be handled carefully. See below.

Total kill

The sequence of necessary calculations to estimate *total kill* from *observed captures* is as follows, with available data sources identified. This sequence of estimates remains a considerable source of uncertainty, noting that we have not yet quantified how uncertainty propagates though to risk score.. Research to inform these estimates with new data is a high priority.

Trawl

A. [*observed kills] = [observed captures*] – [birds released alive and not fatally injured]

Source: estimate only, from data collected by fisheries observers, not obtained yet

B. [*observed net kills*] = A * [proportion of total observed kills that are due to net capture]
   Source: estimate only, from fisheries observers, not obtained yet
C. [*observed warp strike kills*] = (A – B)
D. [*Total warp strikes*] = C * [total warp strike multiplier]
   = approx. 200 for albatrosses, 6000 for petrels (Source: Abraham and Thompson 2008; see also Sullivan et al. 2006)
E. [Total *surface warp strikes*] = D / [fraction of warp strikes that are surface warp strikes]
   = approx 0.66 – 0.91 (i.e. 2:1 – 10:1 ratio) Source: workshop biologists; (but should be disaggregated by species)
F. [Total aerial warp strikes] = (D – E)
G. [Total surface warp strike kills] = E * [proportion fatal surface warp strikes]
   = 0.4 for albatrosses, 0.2 for petrels (Source: workshop biologists, see Table 1; compare also with Watkins et al (2008) who asserts G / C = 15).
H. [Total aerial warp strike kills] = F * [proportion fatal aerial warp strikes]
   = 0.5 for albatrosses, 0.3 for petrels (Source: workshop biologists, see Table 1; see also Sullivan et al. 2006)
I. [Total net kills] = B / [fraction of net kills retained on net]
   = perhaps approx. 0.66-0.80 (Source: ratio estimate only, from workshop)
J. [Total kills] = G + H + I

Derivation of the Risk* score: *R_{max}, PBR, and F

[see also Walker et al. (2009), Waugh and Filippi (2009b, 2009c)]

Step 4.1: Productivity metric - R_{max}
The workshop agreed that population biology/ life history characteristics were directly relevant for a risk assessment of the effects of fishing, as these would determine a population's growth and recovery rates. Subsequent discussion focused on the selection of an appropriate metric to most effectively summarize the relevant characteristics. After some discussion the workshop agreed to use R_{max} as the quantitative metric against which mortality would be assessed to calculate [*absolute risk*]. Biologists present at the workshop reviewed the assumptions inherent in the calculation of R_{max} from the data. The assumptions behind the use of R_{max} include: i) population far below carrying capacity; ii) optimal growth; iii) stable age distribution; iv) fecundity and adult survival are constant. Workshop biologists concluded that these assumptions were acceptable for the adoption of a productivity metric; however this does not imply that actual populations exhibit these characteristics. Rather the index is a consistent metric of population productivity to enable risk calculations and inform comparisons between species; it is recognised that the population growth rate represented by R_{max} is under theoretical ideal conditions, and that real-world growth rates will be lower.

Step 4.3 Risk score equation
The workshop agreed that an upper limit to acceptable risk could be set as a function of the ratio between M (fisheries-induced mortality) and R_{max}. R_{max} represents the maximum theoretical productivity of a population under ideal unconstrained conditions, whereas M represents the deaths due to fishery impacts.,i.e. if $M > R_{max}$ then the population would inevitably decline. Mathematically:

[Risk score] = C * (M/ R_{max}); C > 1

[Risk score] > 1 ➔ unacceptable

[Risk score] < 1 ➔ acceptable

The constant C represents the fact that real-world conditions will never approximate the idealized unconstrained conditions assumed by the R_{max} term, such that the upper limit of acceptable M values is actually lower than R_{max}.

Derivation of an appropriate constant C was achieved with reference to the Potential Biological Removals (PBR) formulation. The Potential Biological Removal is a technique that was developed by the United States National Marine Fisheries Service to calculate the maximum number of animals that may be removed from a marine mammal stock, not including natural mortalities, while allowing that stock to reach or maintain its optimum sustainable population size. PBR is calculated as follows:

$$PBR = N_{MIN} \tfrac{1}{2} R_{MAX} F_R$$

Where:

N_{MIN} = the minimum population estimate of the stock;

F_R = a recovery factor between 0.1 and 1.0

It can be demonstrated algebraically that the [risk score] equation above and the PBR equation are in fact interchangeable where C = 2/F, i.e.

$$[Risk] = 2M / F^* R_{max} \qquad \text{where } 0.1 < F \leq 1$$

Step 4.2 Use of management factor F

The incorporation of the management factor F provides a simple and transparent mechanism by which management objectives can be explicitly incorporated into the calculation of the risk score, with guidance from international precedent. Literature associated with the use of the PBR metric labels F a ‘rebuilding factor’ with values ranging from 0.1 to 1. The extent to which F is lower than 1 reflects the proportion of population productivity that is available for population growth (or other sources of mortality) rather than ‘harvest’ (NMFS 2005). High F values would therefore be recommended for healthy, stable or increasing populations, whereas low values of F would be recommended for declining, globally threatened or severely depleted populations where management goals include rapid population recovery.

The F term can provide a simple and transparent mechanism by which other relevant information (i.e. population status; global threat status) can be incorporated into the calculation of the risk score. The risk assessment workshop explicitly agreed that the following information should be considered by managers and incorporated into the calculation of ‘*absolute risk*’.

- population size (with small populations considered to be at higher risk)
- population trend (increasing, decreasing, stable)
- population status (depleted, not depleted)

The workshop further agreed that calculation of ‘complete risk’ (i.e. including risk arising from threats other than fishing), would need to consider the following:

- other sources of bird mortality (e.g. out of zone fishery mortality)
- terrestrial threats (habitat loss, on-shore depredation by predators)
- global threat status (DoC or IUCN ratings)

See Walker et al. (2009)

Figure 1. Flow diagram of the risk assessment framework. Numbers correspond to numbered steps identified in the text. Ovals denote inputs, for which uncertainty should be made explicit. Boxes denote calculated values. The vulnerability term is derived in a nested subroutine (Figure 2). For each bird species the assessment produces two outputs, the *absolute risk* and *complete risk* scores, for use by fisheries managers.

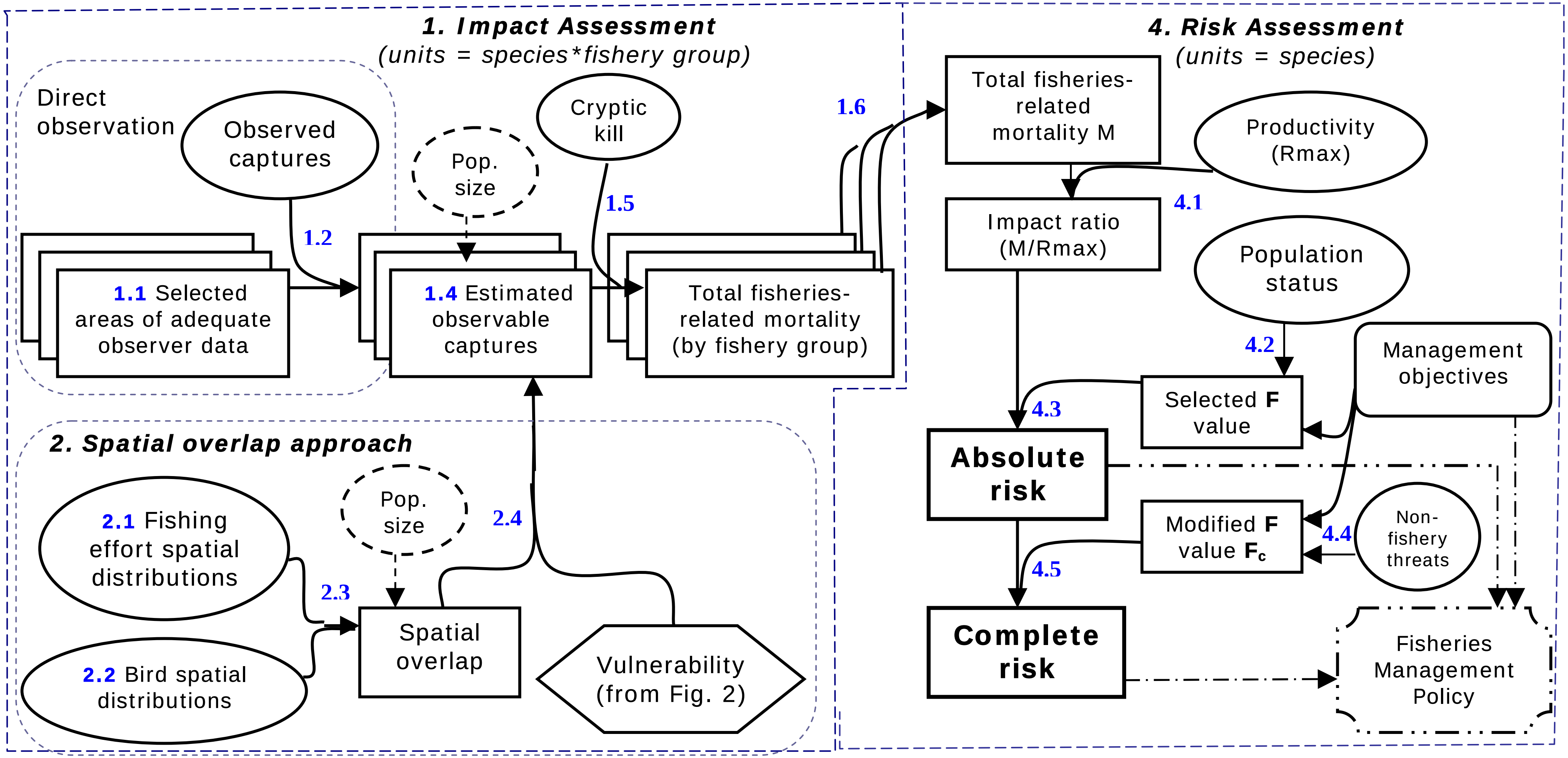

Figure 2: Flow diagram of the vulnerability calibration subroutine (nested within Figure 1). Numbers correspond to numbered steps identified in the text. Ovals denote inputs; boxes denote calculated values.

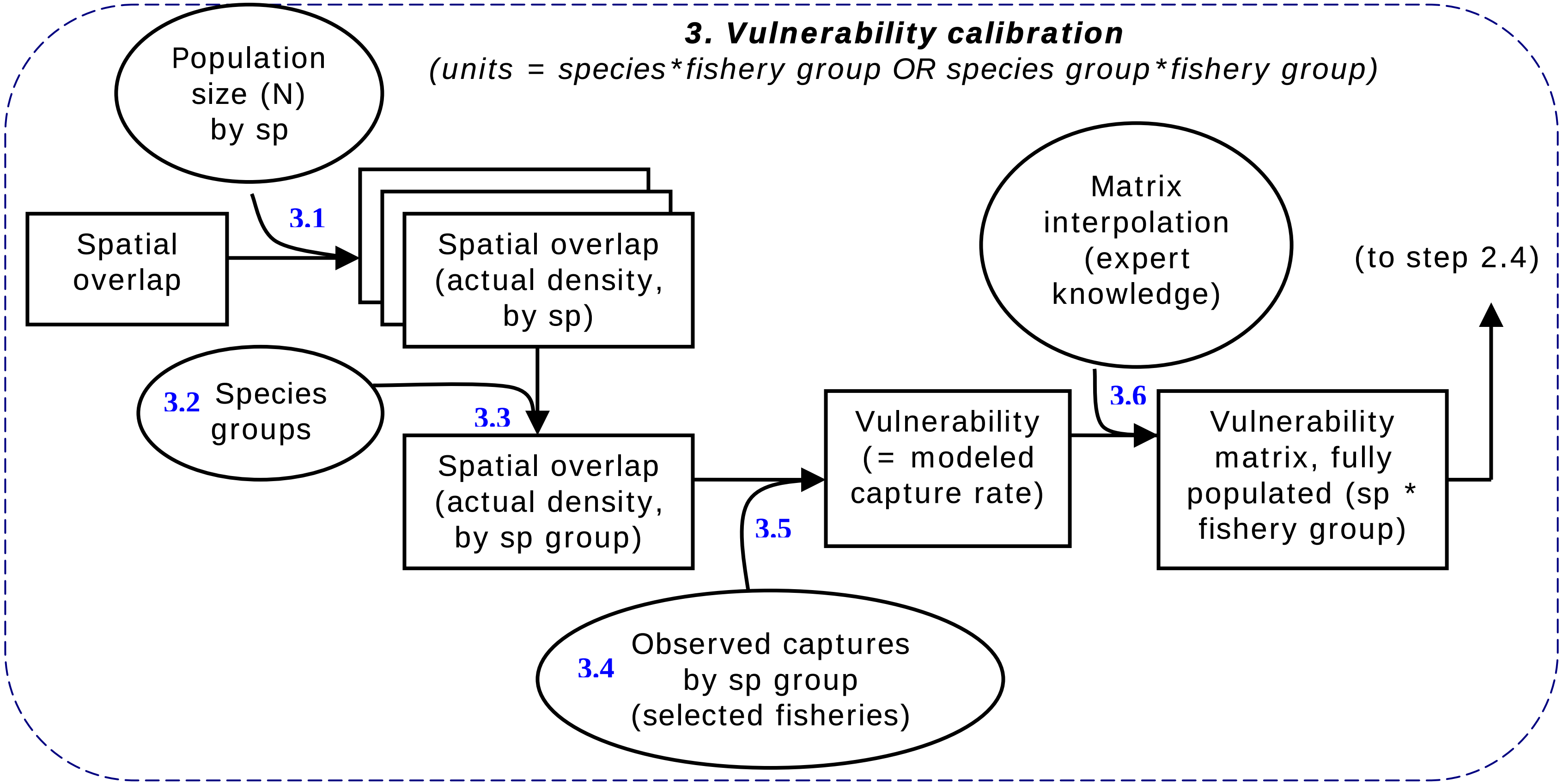

Figure 3. Decision support framework diagram illustrating different zones of risk corresponding to values of the [absolute risk] and/or [complete risk] scores. Each color-differentiated zone would be associated with recommendations for risk-reduction management interventions of increasing urgency at higher [risk] scores. The plotted curve represents a theoretical risk trajectory over time for a bird species for which risk-reduction interventions have been implemented. Advanced versions of this plot would include explicit representations of the shape of the uncertainty curve associated with each [risk] score estimate.

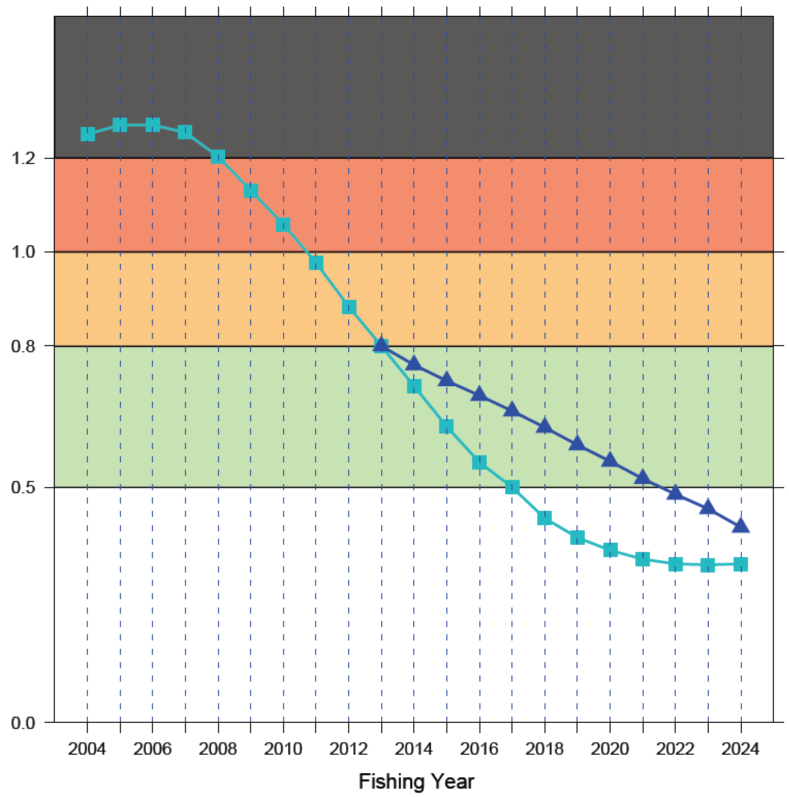

Table 1. *Seabird species included in the study, including species group affiliation and population parameters used in the calculation of R_{max} (i.e. annual adult survival and age at first breeding). Where these are unavailable, proxy species indicated from which substitute values were used (Waugh and Filippi 2009c).*

Species group	***BLI Scientific name***	***BLI Common name***	***age mat average***	***surv average***	***Rmax***	***Lambda max***	***Proxy species value***	***Distribution data sources***
Gannets	*Morus serrator*	Australasian Gannet	5	90	0.113	1.113		BirdLife range map
Gannets	*Sula dactylatra*	Masked Booby	3	92.5	0.151	1.151		BirdLife range map
Gulls&terns	*Gygis alba*	Common White Tern	5	84	0.136	1.136	Black noddy	BirdLife range map
Gulls&terns	*Larus bulleri*	Black-billed Gull	3.7	85.6	0.167	1.167	Red-billed Gull	NABIS
Gulls&terns	*Larus dominicanus*	Kelp Gull	4	81	0.173	1.173		BirdLife range map
Gulls&terns	*Larus novaehollandiae*	Silver Gull	3.7	85.6	0.167	1.167		-
Gulls&terns	*Procelsterna cerulea*	Blue Noddy	3	75	0.248	1.248	Black noddy	-
Gulls&terns	*Sterna caspia*	Caspian Tern	3	89	0.18	1.18		BirdLife range map/NABIS
Gulls&terns	*Sterna fuscata*	Sooty Tern	3	80.5	0.226	1.226	Bridled tern	BirdLife range map
Gulls&terns	*Sterna nereis*	Fairy Tern	3	84	0.208	1.208	Little tern	BirdLife range map
Gulls&terns	*Sterna striata*	White-fronted tern	2	85	0.29	1.29	Common tern	-
Gulls&terns	*Sterna vittata*	Antarctic Tern	1.8	90	0.263	1.263	Antarctic tern	BirdLife range map
Large albatross	*Diomedea antipodensis*	Antipodean Albatross (Auckland I)	7	97	0.057	1.057		BirdLife range map/BirdLife Tracking/NABIS
Large albatross	*Diomedea antipodensis*	Antipodean Albatross (Antipodes I)	7	97	0.057	1.057		BirdLife range map/BirdLife Tracking/NABIS
Large albatross	*Diomedea epomophora*	Southern Royal Albatross	7	97	0.057	1.057		BirdLife range map/BirdLife Tracking/NABIS
Large albatross	*Diomedea exulans*	Wandering Albatross	9	96	0.052	1.052		BirdLife range map/BirdLife Tracking
Large albatross	*Diomedea sanfordi*	Northern Royal Albatross	7	94.6	0.07	1.07		BirdLife range map/BirdLife Tracking
Large Pterodroma petrels	*Pterodroma lessonii*	White-headed Petrel	5.5	93	0.09	1.09	Pterodroma phaeopygia	BirdLife range map
Large Pterodroma petrels	*Pterodroma macroptera*	Great-winged Petrel	6.5	93	0.083	1.083		BirdLife range map/NABIS
Large Pterodroma petrels	*Pterodroma mollis*	Soft-plumaged Petrel	6.5	93	0.083	1.083		BirdLife range map
Large shearwaters	*Puffinus carneipes*	Flesh-footed Shearwater	5	93	0.101	1.101		BirdLife range map/NABIS
Large shearwaters	*Puffinus griseus*	Sooty Shearwater	6	93	0.087	1.087		BirdLife range map/BirdLife Tracking/NABIS
Large shearwaters	*Puffinus pacificus*	Wedge-tailed Shearwater	4	93	0.118	1.118		BirdLife range map
Other birds	*Catharacta lonnbergi*	Brown Skua	6	93	0.087	1.087		BirdLife range map
Other birds	*Daption capense*	Cape Petrel	6	94	0.083	1.083		BirdLife range map
Other birds	*Fregetta grallaria*	White-bellied Storm-petrel	4	91	0.132	1.132	Wilson's storm petrel	BirdLife range map
Other birds	*Fregetta tropica*	Black-bellied Storm-petrel	4	91	0.132	1.132	Wilson's storm petrel	BirdLife range map

Species group	BLI Scientific name	BLI Common name	age mat average	surv average	Rmax	Lambda max	Proxy species value	
Other birds	*Garrodia nereis*	Grey-backed Storm-petrel	4	91	0.132	1.132	Wilson's storm petrel	BirdLife range map
Other birds	*Halobaena caerulea*	Blue Petrel	6	84	0.116	1.116	Salvin's Prion	-
Other birds	*Macronectes giganteus*	Southern Giant-petrel	7	93	0.079	1.079		BirdLife range map
Other birds	*Macronectes halli*	Northern Giant-petrel	7.5	93	0.075	1.075	Southern Giant petrel	BirdLife range map/NABIS
Other birds	*Oceanites maorianus*	New Zealand Storm-petrel	5	86	0.129	1.129	White faced storm petrel	BirdLife range map
Other birds	*Pachyptila crassirostris*	Fulmar Prion	5	84	0.136	1.136	Fairy prion	-
Other birds	*Pachyptila desolata*	Antarctic Prion	4.5	84	0.149	1.149	Salvin's Prion	BirdLife range map
Other birds	*Pachyptila turtur*	Fairy Prion	4.5	84	0.149	1.149	Salvin's Prion	BirdLife range map
Other birds	*Pachyptila vittata*	Broad-billed Prion	4.5	84	0.149	1.149	Salvin's Prion	BirdLife range map/NABIS
Other birds	*Pelagodroma marina*	White-faced Storm-petrel	5	86	0.129	1.129	White faced storm petrel	NABIS
Other birds	*Pelecanoides georgicus*	South Georgia Diving-petrel	2	81	0.322	1.322	Common DP	NABIS
Other birds	*Pelecanoides urinatrix*	Common Diving-petrel	2	81	0.322	1.322		BirdLife range map
Other birds	*Phaethon rubricauda*	Red-tailed Tropicbird	3.5	90	0.152	1.152		-
Other birds	*Pterodroma axillaris*	Chatham Petrel	6.5	93	0.083	1.083		BirdLife range map
Other birds	*Pterodroma cervicalis*	White-necked Petrel	6.5	93	0.083	1.083		BirdLife range map
Other birds	*Pterodroma cookii*	Cook's Petrel	6.5	93	0.083	1.083		BirdLife range map
Other birds	*Pterodroma inexpectata*	Mottled Petrel	6.5	93	0.083	1.083		BirdLife range map
Other birds	*Pterodroma magentae*	Magenta Petrel	6.5	93	0.083	1.083		BirdLife range map
Other birds	*Pterodroma neglecta*	Kermadec Petrel	6.5	93	0.083	1.083		BirdLife range map/NABIS
Other birds	*Pterodroma nigripennis*	Black-winged Petrel	6.5	93	0.083	1.083		-
Other birds	*Pterodroma pycrofti*	Pycroft's Petrel	6.5	92	0.087	1.087		BirdLife range map
Other birds	*Puffinus assimilis*	Little Shearwater	5	93	0.101	1.101		BirdLife range map
Penguins	*Eudyptes filholi*	Eastern Rockhopper Penguin	6	86	0.112	1.112		BirdLife range map/NABIS
Penguins	*Eudyptes pachyrhynchus*	Fiordland Penguin	3.5	85	0.177	1.177	Fiordland penguin	BirdLife range map/NABIS
Penguins	*Eudyptes robustus*	Snares Penguin	4	85	0.16	1.16	Fiordland penguin	BirdLife range map
Penguins	*Eudyptes sclateri*	Erect-crested Penguin	5	85	0.134	1.134	Fiordland penguin	BirdLife range map
Penguins	*Eudyptula minor*	Little Penguin	2.5	74.5	0.298	1.298		-
Penguins	*Megadyptes antipodes*	Yellow-eyed Penguin	2	87	0.272	1.272		BirdLife range map/NABIS
Procellaria petrels	*Procellaria aequinoctialis*	White-chinned Petrel	6.5	89	0.096	1.096		BirdLife range map/NABIS

Species group	*BLI Scientific name*	*BLI Common name*	*age mat average*	*surv average*	*Rmax*	*Lambda max*	*Proxy species value*	
Procellaria petrels	*Procellaria cinerea*	Grey Petrel	7	93	0.079	1.079		BirdLife range map/NABIS
Procellaria petrels	*Procellaria parkinsoni*	Parkinson's Petrel	7	88	0.094	1.094		BirdLife range map/BirdLife Tracking/NABIS
Procellaria petrels	*Procellaria westlandica*	Westland Petrel	6	88	0.105	1.105		BirdLife range map/BirdLife Tracking/NABIS
Shags	*Phalacrocorax campbelli*	Campbell Island Shag	3	86	0.197	1.197	Great & Spotted shag	BirdLife range map/NABIS
Shags	*Phalacrocorax carunculatus*	New Zealand King Shag	3	86	0.197	1.197	Great & Spotted shag	BirdLife range map/NABIS
Shags	*Phalacrocorax chalconotus*	Stewart Island Shag	3	86	0.197	1.197	Stewart Is shag	BirdLife range map/NABIS
Shags	*Phalacrocorax colensoi*	Auckland Island Shag	3	86	0.197	1.197	Great & Spotted shag	BirdLife range map/NABIS
Shags	*Phalacrocorax featherstoni*	Pitt Island Shag	2	95	0.175	1.175	Spotted Shag	BirdLife range map/NABIS
Shags	Phalacrocorax onslowi	Chatham Island Shag	3	86	0.197	1.197	Great & Spotted shag	BirdLife range map
Shags	Phalacrocorax punctatus	Spotted Shag	2	95	0.175	1.175		-
Shags	Phalacrocorax ranfurlyi	Bounty Island Shag	3	86	0.197	1.197	Great & Spotted shag	BirdLife range map/NABIS
Shags	Phalacrocorax varius	Large Pied Cormorant	3	88	0.184	1.184	Great cormorant	-
Small albatrosses	Phoebetria palpebrata	Light-mantled Albatross	7	97.3	0.046	1.046		BirdLife range map/NABIS
Small albatrosses	Thalassarche bulleri	Buller's Albatross (Northern)	5	91.3	0.109	1.109		BirdLife range map/NABIS
Small albatrosses	Thalassarche bulleri	Buller's Albatross (Southern)	5	91.3	0.109	1.109		BirdLife range map/NABIS
Small albatrosses	Thalassarche carteri	Indian Yellow-nosed Albatross	9	93.5	0.064	1.064	Average S for mollymawks	BirdLife range map
Small albatrosses	Thalassarche chrysostoma	Grey-headed Albatross	10	95.3	0.053	1.053		BirdLife range map/BirdLife Tracking/NABIS
Small albatrosses	Thalassarche eremita	Chatham Albatross	7	93.5	0.076	1.076	Average S for mollymawks	BirdLife range map/BirdLife Tracking/NABIS
Small albatrosses	Thalassarche impavida	Campbell Albatross	10	94.5	0.055	1.055		BirdLife range map/BirdLife Tracking/NABIS
Small albatrosses	Thalassarche melanophrys	Black-browed Albatross	7	95.1	0.068	1.068		BirdLife range map/BirdLife Tracking
Small albatrosses	Thalassarche salvini	Salvin's Albatross	7	94	0.074	1.074		BirdLife range map/NABIS
Small albatrosses	Thalassarche steadi	White-capped Albatross	7	94	0.074	1.074		BirdLife range map/NABIS
Small shearwaters	Puffinus bulleri	Buller's Shearwater	5	93	0.101	1.101		BirdLife range map
Small shearwaters	Puffinus gavia	Fluttering Shearwater	5	93	0.101	1.101		BirdLife range map
Small shearwaters	Puffinus huttoni	Hutton's Shearwater	5	93	0.101	1.101		BirdLife range map

Table 2. *Fishery groups defined on the basis of vessel size, method and target species.*

Fishery Group	Description	Primary method
2,3	Setnet	Setnet
1	Inshore trawl	Trawl
4	Bluenose	Longline
5	Bottom longline small	Longline
6	Snapper longline	Longline
9	Bottom longline autoline	Longline
10	Surface longline large	Longline
11	Surface longline small	Longline
12	Middle-depth processor	Trawl
13	Middle-depth fresher	Trawl
15	Southern blue whiting	Trawl
16	Scampi	Trawl
17	Mackerel	Trawl
18	Squid	Trawl
19	Deepwater	Trawl
20,21	Flatfish	Trawl

Table 3. *Relative vulnerability scores subjectively assigned by experts at the NPOA seabird risk assessment workshop for for all species groups and three generic fishing method types. Ratings range from 0 (negligible effect) to 5 (high likelihood of effect). Shaded cells are those adjusted following examination of empirically derived V values from the vulnerability calibration (see text). Pending values are those requiring revision following new information about captures of shags and penguins in inshore trawl fisheries.*

Species group	***Longline***	***Trawl***	***Setnet***
Gannets	0	0	3
Gulls & terns	0	0	0
Large albatrosses	5	4	0
Large *Pterodroma* petrels	2	1	0
Large shearwaters	2	2	2
Other petrels	1	1	0
Penguins	0	Pending	5
Procellaria petrels	5	4	0
Shags	0	Pending	5
Small albatrosses	4	5	0
Small shearwaters	2	2	4

Table 4: *Expected outcome of warp strikes: proportion of warp strikes resulting in fatal injury*

	Albatrosses		Petrels	
	Median	Range	Median	Range
Aerial warp strike	0.55	(0.2 – 0.9)	0.4	(0.2 – 1)
Surface warp strike	0.45	(0.05 – 0.8)	0.1	(0.01 – 0.6)