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The justification, design and implementation of Ecological Risk Assessments of the effects of fishing on seabirds

BirdLife International/UK

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Abstract

Many marine species are threatened by high levels of incidental mortality in fisheries. This paper reviews the design of recent Ecological Risk Assessments (ERAs) of the effects of fishing on seabirds. Several aspects of the ERA methodology for seabirds are still in development, including the most appropriate way to estimate seabird distribution and fisheries overlap, the role of bycatch data, the best measure of productivity, and the way that data gaps should be handled. Other issues to be considered when undertaking an ERA include the appropriate selection of species, the definition of risk, the appropriate spatial and temporal resolution for the analysis, and establishing links between the ERA analysis and management responses. There are several benefits of undertaking ERAs: ERAs identify key areas and seasons in which bycatch may be occurring, highlight data gaps, and can be used to incorporate precautionary approaches and decision-making on bycatch into a broader fisheries management framework. However, experience so far highlights several methodological issues that need further consideration, and the possibility that an ERA may draw attention away from existing responsibilities and commitments to reduce bycatch *per se*. When selecting the best approach, it is vital to balance desired outputs against the availability of data for the assessment, and to deal with data gaps in a precautionary manner,

Introduction

The incidental mortality of non-target species in fisheries is widely-acknowledged to be a major threat to marine biodiversity, with the potential for deleterious long-term ecological impacts on ocean ecosystems (Baum et al. 2003; Lewison et al. 2004; Myers and Worm 2003). Many of the worst affected species are seabirds, particularly albatrosses and large petrels, which, as natural scavengers, are attracted to vessels by the availability of bait and discards (Cherel et al. 1996; Phillips et al. 2006). In longline fisheries, birds target baits during line setting, and can become hooked and drowned; in trawl fisheries, mortality is primarily the result of strikes with warp cables, although entanglement can also occur (Sullivan et al. 2006).

The FAO Code of Conduct for Responsible Fisheries and the UN Fish Stocks Agreement established the requirement in fishery management to minimize impacts on non-target species, and established the ‘Ecosystem Approach’ and the ‘Precautionary Approach’ as key approaches necessary to achieve sustainable management of the world’s fisheries (FAO 1995, United Nations 1995). However, many fisheries regulatory bodies around the world have struggled to embed the ecosystem and precautionary approaches into their management decision-making in a meaningful and practical way. Ecological Risk Assessments for the Effects of Fishing (ERAs) offer a framework through which fisheries managers can approach this, by identifying the species or areas where the risk of negative interaction is greatest, by risk assessment taking data scarcity and uncertainty into consideration, and, ideally, by linking risk assessment to pre-determined rules for decision-making.

Several national and international fisheries bodies, including the Commission for the Conservation of Antarctic Marine Living Resources (CCAMLR), International Commission for the Conservation of Atlantic Tuna (ICCAT), Western and Central Pacific Fisheries Commission (WCPFC) and the Ministry of Fisheries (MFish) in New Zealand have developed Ecological Risk Assessments (ERAs) of the impacts of fishing on seabirds (Phillips & Small 2007, Tuck et al. in press; Waugh et al. 2009, Kirby et al. 2009, Sharp et al. 2009, Filippi et al. 2010). The purpose of this paper is to review the methods used in these ERAs and, in so doing, to highlight key issues, and provide recommendations for their design and implementation in the future.

The ERA framework

Although the seabird ERAs undertaken by CCAMLR, ICCAT, WCPFC and MFish have used differing methodologies, all fall broadly within the ERA framework developed in 2002-2006 by the Commonwealth Scientific and Industrial Research Organisation (CSIRO, Australia) for the Australian Fisheries Management Authority (Smith et al. 2007). This framework was originally proposed by Sainsbury and Sumaila (2001), and involves three progressive stages, with assessment moving from one stage to the next depending on the level of risk identified, the data available, and the management response. Under the CSIRO framework, Level 1 of an ERA involves a comprehensive but largely qualitative “Scale, Intensity, Consequence” analysis, Level 2 involves a more focused and semi-quantitative “Productivity-Susceptibility” analysis, and Level 3 involves a highly quantitative model-based analysis. Level 3 is focused on species identified by the previous levels as being at high risk. Importantly, the framework envisages management responses at each level, and a precautionary approach exemplified by assigning high-risk scores where data are unavailable (Hobday et al. 2007).

Existing ERAs can be categorized as follows: the CCAMLR method is similar to a Level 1 analysis, the WCPFC and MFish ERAs correspond largely to Level 2, and the ICCAT seabird assessment corresponds to Levels 1-3, although only four breeding population are considered at the highest level. There are additional examples of Level 3 type analyses in the peer-reviewed literature, generally focused on a single species (Baker and Wise 2005, Lewison and Crowder 2003, Rolland et al. 2009, Tuck et al. 2001). More information on the key aspects of these ERAs can be found in Table 1 and Appendix 1.

Key considerations in the design of seabird ERAs

A review of the existing seabird ERAs highlights a number of important issues.

1. Species or populations to include in an ERA

An early decision in the design of an ERA for seabirds involves which species or populations to include. The CCAMLR risk assessment restricts itself to albatrosses and petrels, on the basis that these are the species most often caught in its longline and trawl fisheries. The ICCAT risk prioritisation included only species that were recorded as bycatch in ICCAT fisheries, and five additional species that had been caught by tuna fleets in other regions. In the WCPFC and MFish risk assessments, however, if one species of a genus had been recorded as bycatch then all species in that genus were included. A restricted approach (e.g. ICCAT) has the advantage of ensuring that the outputs of the ERA are focused on those seabird species which are known to be vulnerable to capture. However, an inclusive approach is likely to be necessary in situations in which species-specific bycatch data are sparse. The most appropriate species selection will reflect the type of fishery: longline hook size will affect the range of species caught, and longline fisheries capture surface-feeders, including albatrosses and petrels, whereas gill nets also ensnare diving species, including shags, penguins, shearwaters, alcids, and ducks (Cherel et al. 1996; Sullivan et al. 2006; Žydelis et al. 2009).

There is also the question of whether to use species or populations as the appropriate units for analysis. The ICCAT ERA was based on breeding populations (island group or region). The advantage of this higher resolution is that populations may differ substantially in relative risk in terms of overlap with fisheries. However, the disadvantages are that it is impossible to assign bycatch or determine relative overlap with fisheries of a particular population without independent information on bird distribution (e.g. from tracking data, ring recoveries or morphological comparisons). For this reason, most ERAs have been based on species. Ideally, ERA methods should be flexible enough to allow inclusion of both species and populations in the ERA and, if data are available, to incorporate different parameter values for different populations. Whatever criteria are used, identification of the appropriate species or populations for inclusion in the analysis is critical to undertaking an ERA efficiently and effectively, and should be guided by expert opinion from the outset.

2. Defining risk

Although difficult, the definition of risk is important when undertaking an ERA as it will influence the choice of analysis, and the data and assumptions required, as well as the likely outcomes and consequent management responses. The CCAMLR, ICCAT and WCPFC ERAs use relative measures of risk. In the ICCAT assessment, risk scores from the productivity-susceptibility analysis were categorized as 'low' 'medium' or 'high', based on assigning around one third of the populations to each category, and expert opinion was used to confirm that the cut-off points were appropriate. Similarly, the WCPFC ERA divided risk scores into five evenly-populated categories, ranging from low to high risk, after exclusion of species for which the risk was considered to be negligible. In contrast, the MFish ERA had a quantitative estimate of population-level impact, whereby an *Impact Ratio* was defined as the estimate of current fishing mortality divided by potential biological removal (PBR).

The attraction of attempting a measure of absolute risk is that, if estimated with sufficient accuracy, it can form a response variable that can be monitored as management measures are implemented. The drawback, however, is that such an approach depends on the availability and accuracy of large amounts of census, demographic, distribution and bycatch data. It is also necessary for PBRs to adequately account for all other sources of mortality. The reality is that for many bycaught species, even basic data on population size and status are unknown. Similarly, many fisheries worldwide have insufficient levels of observer coverage to be able to adequately estimate species-specific bycatch rates of seabirds with representative spatial-temporal coverage. Hence, the quantitative estimation of impacts of bycatch is usually problematic, and often impossible (Waugh et al. 2009).

There are additional justifications for avoiding the definition of risk in terms of impacts on species or populations, notably because: (1) the Code of Conduct and UN Fish Stocks Agreement establish the duty to minimize bycatch *per se*, and (2) for threatened species, any additional sources of mortality may cause a decline and so should be avoided even if impacts of fisheries cannot be proven for the area in question. Bearing these issues in mind, an appropriate aim for an ERA in relation to seabirds might be as follows: *"An assessment of the risk of occurrence of incidental mortality of seabirds resulting from interactions with* *fisheries, in particular the risk of incidental mortality of threatened species, or of mortality known or likely to have an impact on populations".*

3. Focus on risk prioritisation and productivity-susceptibility analysis

Within ERAs, Level 3 type models can be very powerful in assessing population-level impact of fisheries on seabirds. However, they can only be applied to the limited number of species for which comprehensive data are available. They can also create situations in which, in contrast to a precautionary approach, the burden of proof is placed on an ERA to demonstrate population-level impacts before action is taken to reduce bycatch. Based on experience from existing seabird ERAs, the initial priority should be given to Level 1 and Level 2 analyses that focus on the risk ranking of most or all species or populations of interest. Level 3 type analyses can provide useful case studies that support the results from Level 2, but given the data requirements and the effort needed for a thorough analysis, they will only be appropriate for a limited number of species. However, it is possible that future methodical development would make a Level 3 approach applicable to a wider range of seabirds.

4. Measures of productivity

A measure of productivity is needed for a Level 2 ERA analysis, which ranks species as high relative risk if they have low productivity or high susceptibility to bycatch in fisheries. In fisheries contexts, the term *productivity* is usually considered to reflect the natural growth rate of a population in the absence of fisheries mortality.

In the ICCAT seabird assessment, productivity was measured by the single variable of *life history strategy* (see Appendix 1*)*. Additional variables, such as age at first breeding, were considered for inclusion, but it was concluded that *life history strategy* captured the key differences among species in natural population growth rate. A more quantitative approach was trialed in the MFish ERA; a value for *Rmax* was estimated for each species using available data or substitutions from related species (around 1/3 of the parameter values were substitutions). Reliable data on age of first breeding and adult survival are unavailable for many species, in particular for burrow-nesting seabirds for which it may be impossible to discriminate between permanent emigration and mortality. Moreover, past studies have shown extensive variation in demographic parameters and population growth rates among populations of the same seabird species (Frederiksen et al. 2005; Nevoux et al. 2010). In addition, there are few estimates of adult survival prior to the advent of large-scale industrial fishing, yet the productivity parameter should reflect mortality in the absence of fishing impacts; hence, there is risk of some circularity in the wider analysis. Thus, estimates of *Rmax* may be unreliable, and consequent ranking of species could be misleading, despite the impression of accuracy provided by this quantitative approach (Waugh et al, 2009). The WCPFC ERA (Filippi et al. 2010, Waugh et al. submitted) compared an *Rmax* based index with an adapted version of the *life history strategy* variable (weighting it by age at first breeding, and called the '*Fecundity Factors Index*'), and found them to be closely correlated. The use of the more straightforward measure for productivity is preferred by these authors, as it has been found to provide sufficient discrimination among species in relation to their capacity to buffer impacts of fisheries, and more appropriately reflect the quality of data that are currently available.

5. Measures of seabird distribution

In Level 2 ERAs, *susceptibility* is measured as the degree of overlap between seabird distribution and fishing effort, taking into account the behavior of each species in terms of their vulnerability to bycatch.

There are clear benefits in attempting to quantify seabird density-distribution and overlap with fisheries; without this, it is impossible to identify the areas and seasons with highest risk of bycatch. However, when choosing the method for estimating overlap between seabird distribution and fisheries, there is a need to strike a pragmatic balance between a simplistic "back-of-the-envelope" approach and more complex calculations. "Back of the envelope" estimates lack precision, but more complex methods can be thwarted by data gaps and untestable or invalid assumptions, and therefore can convey false impressions of accuracy, or limit the assessment to the minority of species for which sufficient data are available.

Options for methods to estimate seabird distribution include: (i) expert opinion, (ii) range maps (assuming homogeneous distribution throughout the range), (iii) use of a range map to represent non-breeding distribution and a foraging radius to represent breeding distribution, (iv) refining foraging radius based on known habitat preference (e.g. for shelf waters), (v) using a combination of range map, foraging radius and tracking data, as available, (vi) use only tracking data and limit the assessment to those species for which data are available, and (vii) develop a model of distribution, including for areas and populations for which data are lacking, based on analysis of habitat preference (from tracking data or at-sea observations), limiting the assessment to a minority of species.

In the CCAMLR approach, all available seabird distribution data are considered along with fishing distribution data, and used to create a qualitative risk score (1-5) for each of seventeen areas. The ICCAT, WCPFC and MFish analyses pursued a more quantitative estimate of seabird distribution using a combination of species range maps, estimates of foraging radii from the colony during the breeding season, information on the duration of the breeding and non-breeding periods, and assumptions about population structure (by age and breeding status). The WCPFC analysis also incorporated information from tracking data where available (Appendix 1).

Each of the existing ERAs used a slightly different approach to estimating seabird distribution, and it is an area that would benefit from methodological development. Based on the existing ERAs, a number of issues are apparent: (1) Sufficient tracking data are available only for a limited number of species (e.g. 5 of the 40 seabird populations in the ICCAT analysis); for many species the best available distribution data will consist of a range map and potentially an estimate of foraging radius during the breeding season. (2) Range maps are usually for an entire species, but the breeding population considered in an ERA may occupy only a portion of this overall area. (3) Foraging areas around colonies are rarely circular in shape, and often vary greatly with breeding stage and colony, hence the use of a single radius is frequently unrealistic; however, one partial solution is to exclude particular sectors based on knowledge of habitat preference. (4) Population age structure is rarely known with confidence, and is species-specific.

Despite these issues, it is possible to offer the following general advice: (1) The best available measure of foraging radius is likely to be the mean maximum of all trips based on tracking data; this is preferable to the mean of all fixes, or the absolute maximum in the dataset (the latter is often far greater than the average maximum). (2) For species for which no tracking data exist, data substitutions from similar species should be treated with considerable caution. (3) Estimation of distribution at least by year quarter is highly desirable, given the often highly seasonal nature of both seabird and fishing effort distribution. (4) Experts should be invited to review the bird distribution maps and refine as necessary. (5) It is valuable for an ERA to test sensitivity to assumptions (e.g. Waugh et al. 2009) to assess uncertainty in overlap estimates. (6) Ultimately, the ERA need only match the resolution of the data on bird distribution to that available for fishing effort – if the latter are at 5x5 degree resolution, then some of the finer scale inaccuracies in estimating bird distribution may be of little consequence. Spatial scale is also an important consideration: in small, localised fisheries, the information on bird distribution may not be of sufficient resolution to be able to estimate overlap reliably. (7) Further development of methods to estimate seabird distribution are needed.

6. Calculating overlap with fishing effort

The ICCAT seabird ERA used three measures of overlap between seabird distribution and longline fishing effort, and calculated overlap by month (Appendix 1). The most appropriate of these was considered to be the product of proportion of the overall seabird distribution, and fishing effort, within each 5 degree grid square, per month. The MFish ERA (Waugh et al. 2009) focused on annual overlap, since bird distributions were estimated for the year as a whole, and it used number of birds rather than percent distribution to calculate overlap (number of birds x fishing effort per 0.1 degree square). The WCPFC analysis developed both of these overlap calculations further, calculating risk as (i) the product of species distribution and fishing effort per square kilometer and year-quarter, which allowed spatial and temporal risk to be illustrated on maps, rather than just overlap, and (ii) also weighting seabird distribution by population size to create a second overlap score reflecting likely numbers of birds caught. The second approach permitted the identification of areas and seasons in which bycatch was likely to be higher in absolute terms, in addition to those areas and seasons in which bycatch impacts were likely to be most severe at a species level.

Key conclusions from existing ERAs are that wherever possible analyses of overlap should take account of the usually substantial seasonal changes in seabird distribution and fishing effort (and hence in seabird-fishery overlap). This allows the identification of key periods as well as regions in which bycatch rates are likely to be highest, leading to better targeting of monitoring effort and bycatch mitigation. However, in most cases, given data limitations for estimating bird distribution, the most appropriate resolution for this may be year-quarter estimates of seabird distribution (rather than monthly), at a spatial scale comparable to that of the fishing effort data. Overlap calculations based on percent seabird distribution or numbers of birds may both be useful depending on the questions being addressed.

7. Role of seabird bycatch data

Data on seabird bycatch are often sparse and biased in relation to geographical and seasonal extent (Anderson et al. 2011). As such, they can be used to confirm where bycatch is occurring, but, for most fisheries, areas and seasons, it would be unwise to use seabird bycatch data to infer that bycatch is not occurring.

In the MFish and WCPFC ERAs, bycatch data were used to calculate *Vulnerability* for each of several sets of species, based on observer data from New Zealand, and involving the calculation of a catchability metric for seabirds at several thousand sampling locations, correcting for estimated density. The *Vulnerability* measure was therefore an index of the likelihood of capture of each species within the relevant group, and was applied to estimates of seabird-fishery overlap in order to estimate the number of birds killed per year. This approach is relatively simple, and addresses the limitation of overlap scores which do not incorporate information on relative behavioural susceptibility of different species to bycatch, for example those calculated in the ICCAT Level 2 ERA. However, as noted by Waugh et al. (2009), even in the New Zealand context, data to calculate *Vulnerability* were sparse for some species groups, and bycatch data of sufficient quality are even less likely to be available for many other fisheries.

8. Dealing with data gaps

It is important that data scarcity and uncertainty are dealt with appropriately within the ERA. One approach to fill empty cells in an analysis is to apply the precautionary principle and assign a score associated with high risk (Hobday et al. 2007). This approach was used in the ICCAT ERA. The alternative is to fill data gaps by substituting a value from a species that is preferably closely related and an ecological analogue, or to exclude species for which data are not available (e.g. WCPFC and MFish ERAs). If the latter approach is taken, clearly great care is needed not to underestimate risk. In specific cases where values are uncertain and have high leverage in the outputs, sensitivity analyses are useful (Waugh et al. 2009).

9. Implementation of seabird ERAs and links to management

Within the CSIRO ERA framework, each of the three levels of analysis are linked to management responses (Hobday et al. 2007). This is also the case for the ERA undertaken by CCAMLR, with the risk scores linked to pre-determined management decisions. MFish is also planning to base management responses based regular updates of the ERA. In contrast, ICCAT and WCPFC ERAs were not pre-linked to management responses, and, to date, management decisions have not yet directly arisen from them. Before an ERA is undertaken, it would be beneficial to plan in advance how the outputs of the risk assessment will be used, what type of management responses would be appropriate, and, ideally, to identify some pre-agreed management decisions. This is important both to ensure that management responses are taken, and that these management responses are appropriate to the type of outputs that the ERA can provide. There is clearly a benefit to carrying out an ERA under the auspices of a relevant working group within a fisheries management body, in order to ensure engagement with the process. This does not overcome the problem that any decisions or recommendations by a specialist working group do not necessarily result in management decisions at higher (e.g. Commission) level (Tuck et al. in press).

Conclusions

Although seabird bycatch can be addressed in the absence of formal risk assessment, a number of benefits may derive from undertaking a dedicated ERA process. Even where data are lacking, ERAs can be used to refine understanding of the species at risk from bycatch, and can be used to aid identification of key areas, seasons and fisheries in which bycatch may be occurring. ERAs can also highlight data gaps and research priorities, including the need for higher levels of observer program coverage. Furthermore, ERAs present risk in terms that are familiar to fisheries managers and can be used to incorporate precautionary approaches and decision-making on bycatch into a broader long-term fisheries management framework.

However, experience so far highlights several issues that need further consideration, including the importance of dealing with data gaps in a precautionary manner, the benefits of establishing links between ERA outputs and management decisions, and the possibility that an ERA may draw attention away from existing responsibilities and commitments to reduce bycatch *per se*. In addition, as described above, ERA methodologies for seabirds are still in development and several issues remain to be resolved. When selecting the best approach for a particular fishery or suite of species, there is a need to balance desired outputs, data availability, and complexity of the process. The ideal output would be for an ERA to quantify absolute impact from fisheries in a way that can be monitored in relation to management response. However, in almost all cases, insufficient data are available to do so. Our conclusion is that, at the present time, undertaking ERAs to determine relative risks to species remains a more pragmatic and useful goal. Further work to develop ERA methodology for seabirds would be very useful, particularly in relation to methods for estimating seabird distribution, and taking account of data gaps.

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Table 1. Summary of methods used in existing Ecological Risk Assessments for the effects of fishing on seabirds (ERAs). (CCAMLR – Commission for the Conservation of Antarctic Marine Living Resources, ICCAT – International Commission for the Conservation of Atlantic Tunas, MFish – New Zealand Ministry of Fisheries, WCPFC – Western and Central Pacific Fisheries Commission).

	CCAMLR (Waugh et al. 2008)	**ICCAT** (Phillips & Small, 2007; Tuck et al. in press)	**MFish** (Waugh et al. 2009; Sharp et al. 2009)	**WCPFC** (Kirby et al. 2009; Filippi et al. 2010)
ERA Levels	Level 1	Six stages, covering Levels 1-3 of an ERA	Levels 1 & 2	Levels 1 & 2
Species or populations to include in ERA	Albatrosses and petrels	Species that had been recorded caught in tuna longline fisheries (mostly albatrosses, petrels, some shearwaters), most divided into populations	All species of a genus if one has been recorded as bycatch. Some species excluded on basis of data gaps.	All species of a genus if one has been recorded as bycatch. Some species excluded on basis of data gaps.
Definition of 'risk'	Qualitative: risk score of 1-5 assigned to areas	Semi-quantitative: in Stage 1 (comparable to ERA Level 1), 3 measures of risk score were used, based on *life history* strategy, population trend, IUCN Red List status, overlap with fishing effort and behavioural susceptibility to capture. Two measures summed the above attributes. The third (most analogous to an ERA Level 2 productivity-susceptibility analysis) calculated risk as the square root of 1/productivity x susceptibility, where susceptibility was the average of overlap with fishing effort and behavioural susceptibility (both scored as low, medium or high). Population risk scores were assigned to low, medium, high categories with around 1/3 of the populations assigned to each category. The appropriateness of the cut-offs were checked by expert opinion.	Quantitative: *Impact Ratio* calculated based on the ratio of likely captures to the index of productivity	Semi-quantitative: risk was calculated as susceptibility divided by productivity. Six risk ratings from high to negligible were calculated by dividing the risk scores into five categories including similar numbers of species, with the negligible level set very low (<0.01 out of a range of 0 – 1) to remove noise from the lower end of the scale. In addition: (a) Risk scores were also calculated per square km, allowing risk maps to be generated. (b) Risk scores by species were summed, indicating species most at risk from longline fisheries interactions at the population level. (c) Risk scores for all species and areas were calculated by fishing fleet and used to determine which fleets posed the greatest risk across species.
Measure of productivity	Not used	*Life History Strategy* (1=multiple eggs, 2=single egg, 3=biennial)	Calculated as 0.5* Rmax * F (where F is between 0-1, based on IUCN Red List status) in an approach analogous to potential biological removal. Data substitutions were necessary for	Compared RMax and *Fecundity Factors Index* (similar to the ICCAT *Life History Strategy,* but weighted by age at first breeding), and found them to be correlated. FFI used for the

	CCAMLR	ICCAT	MFish	WCPFC
			around 1/3 of species.	analysis as considered more robust.
Measure of seabird distribution	Qualitative: expert opinion based on a variety of sources	Stage 1 of the ERA used expert opinion (low/medium/high overlap with ICCAT area). In Stage 2, juveniles were assumed to be homogeneously distributed within the species range throughout the year. Breeding adults and immatures were assumed to be distributed homogeneously within a foraging radius from the colony during the breeding season and within the species range in the non-breeding season. The population structure was assumed to be 70% breeding adults, 20% pre-breeders, 10% juveniles, and distribution was estimated by month.	For 24 species, only a range map was available and birds were assumed to be distributed homogeneously across the range throughout the year. For 38 species, data were used from the NABIS database, with three data layers per species. Layers equated to 10% of the population (in the area of 100% NABIS distribution), 40% of the population (90% distribution) and 50% of the population (NABIS hotspot). For one species, tracking data were used.	Non-breeding birds were assumed to be homogeneously distributed within the species range. Breeding birds were assumed to be distributed within a foraging radius from the colony, with density decaying exponentially with distance. Where foraging radii were unavailable, substitutions were made from other species in the genus of similar weight. Where tracking data were available, these were used to supplement the breeding and non-breeding distributions, and maximum density was selected. The population structure was assumed to be 50% breeders (40% for biennial-breeding species) and 50% non-breeders. Breeding season duration was estimated to the nearest month and composite maps were produced for each year quarter.
Measure of overlap with fishing effort	Qualitative: expert opinion based on fishing effort and seabird distribution data	In Stage 3, three calculations of overlap were used: (1) % population distribution within area of ICCAT longline fishing effort, by month, (2) % population distribution in each 5x5 grid square by month, multiplied by number of hooks, (3) % fishing effort within seabird distribution, by month.	For each species, an estimate was made of the *likely captures* per year, based on seabird distribution x fishing effort x *Vulnerability* per 0.1 degree square	Calculated as the product of the normalized species distribution and fishing effort per square kilometre, with fishing effort averaged across eight years (2002-2009). Susceptibility was calculated as the overlap weighted by *Vulnerability*.
Bycatch data	Informs qualitative scoring but not used in quantitative way	A qualitative 'behavioural susceptibility to bycatch' variable was used in the Stage 1/Level 1 analysis. Bycatch data were not used in Stage 3 overlap calculations. Overall bycatch estimates were undertaken in Stage 4.	New Zealand observer data were used to generate a *Vulnerability* score for each species group, based on the observed mortalities from New Zealand observer data, taking seabird density into account.	New Zealand observer data were used to generate a *Vulnerability* score, based on the observed mortalities from New Zealand observer data, taking seabird density into account.
Data gaps	Expert led and precautionary	Data gaps were assigned a high risk score in the Level 1 risk prioritisation	Data substituted from a closely related species/fishery, or excluded from analysis	Data substituted from a closely related species/fishery, or excluded from analysis
Links to	Risk scores linked to pre-agreed	Not linked	Not linked	Not linked

	CCAMLR	**ICCAT**	**MFish**	**WCPFC**
management	management decisions			

Appendix 1. Summary of methods used in Ecological Risk Assessments for seabirds

Commission for the Conservation of Antarctic Marine Living Resources (CCAMLR) (Waugh et al. 2008)

CCAMLR was a pioneer in incorporating the ecosystem and precautionary approaches into fisheries management, and in developing risk assessments for seabirds in fisheries: the latter first carried out in 1997. The CCAMLR approach to risk assessment is simpler than the others discussed below. The decision was made that adopting an approach of 'sustainable catch' of seabirds was neither appropriate nor possible for such a large geographical area given the requirements for data on seabird distribution, ecology and demography, together with an understanding of all sources of mortality. Instead, the aim was to identify the relative risk of capture of seabirds in fishing operations. The CCAMLR risk assessment approach uses "statistical areas" as units of analysis, not species. Each year, each of seventeen areas is assigned a risk rating of 1-5, based on expert-led consideration of seabird distribution within each area (using data from satellite tracking, at-sea surveys and band returns). The assessment is restricted to albatrosses and petrels, on the basis that these are known to be vulnerable to incidental catch. CCAMLR's Working Group on Incidental Mortality Associated with Fishing (IMAF) then considers the risk ratings in relation to seabird bycatch data (which are available from high levels of observer coverage). IMAF makes recommendations for changes or additions to the suite of CCAMLR Conservation Measures, which are applied by risk rating.

International Commission for the Conservation of Atlantic Tunas (ICCAT) (Phillips & Small 2007, Tuck et al. in press.)

A six stage ERA methodology was developed for the ICCAT convention area by the ICCAT Sub-Committee on Ecosystems: (1) identify the seabird species most at risk from fishing; (2) collate the available data on at-sea distributions of these species; (3) analyse the spatial and temporal overlap between species distribution and longline fishing effort; (4) review the existing estimates of bycatch rates; (5) estimate the total annual seabird bycatch; (6) assess the likely impact of this bycatch on seabird populations.

Stage 1 (Phillips & Small 2007), which corresponded to a Level 1 and Level 2 type analysis, used a mix of populations and species as the units of assessment, and included 68 populations (41 species) in the analysis, of which 37 species had been recorded as bycatch within ICCAT longline fisheries, and five additional species included on the basis of being caught in similar fisheries elsewhere. The risk prioritisation used *life history strategy* (1=multiple eggs, 2=single egg, 3=biennial) as the measure for productivity. Susceptibility was calculated as the average of degree of overlap with fisheries (low, medium, high) and behavioural susceptibility to bycatch (low, medium, high), both based on expert opinion. Three different risk-score methods were used, and risk was categorized as 'low', 'medium' and 'high' based on approximately one third of the populations falling into each category. As such, the risk categorization is strictly relative, not absolute. However the results were then circulated to experts to check that the categorizations matched expert opinion. Of the 68 populations, 22 were designated high priority across all risk-score methods, and 41 according to at least one method of prioritization.

Stages 2 and 3 of the ERA calculated overlap based on an estimate of seabird distribution derived from species range maps, estimates of foraging radius during breeding, breeding season duration, and population structure (70% breeding adults, 20% pre-breeders, 10% juveniles), and data on ICCAT longline fishing effort, available at a resolution of 5x5 degree grid squares. Juveniles were assumed to be homogeneously distributed within the range throughout the year, breeding adults and immatures assumed to be distributed homogeneously within the foraging range during the breeding season, and within the species range in the non-breeding season. Three calculations of overlap were used: (1) % distribution within area of ICCAT longline fishing effort, by month, (2) % distribution in each 5x5 grid square by month, multiplied by number of hooks, and (3) % fishing effort within seabird distribution, by month. While this overlap analysis was considered valuable in that it enabled identification of areas and seasons of likely high overlap between fishing effort and seabirds, the number of assumptions that had to be adopted meant that the results were not considered necessarily more robust than the simplistic 'low, medium, high' estimates of overlap in Stage 1.

Stages 4 and 5 of the assessment attempted to estimate the total number of seabirds caught per year in ICCAT longline fisheries. Bycatch rates from individual studies were mapped on to the ICCAT area by 5 degree grid square, given knowledge of the spatial distribution of each fishery. Where bycatch rates were unavailable for particular grid squares and fisheries, values were substituted from the nearest and most appropriate cells. These rates were multiplied by the reported effort to produce bycatch estimates for each grid square, which were then summed across the entire ICCAT area. Stage 6 developed population models for 4 populations for which detailed demographic and distribution data existed, seeking to identify impacts of ICCAT longline fisheries on these populations. Although the models did not fit every aspect of the observed data well, given the inadequacy of data currently available on bycatch rates, they nevertheless clearly demonstrated the major impacts of fishing (for all gear-types) and highlighted the unsustainability of current bycatch levels (Tuck et al. in press).

New Zealand Ministry of Fisheries (MFish) (Sharp, Waugh & Walker. 2009, Waugh et al 2009)

The MFish ERA for seabirds differs from others in that it estimated absolute risk for all the species under consideration. An absolute, as opposed to relative-risk score was considered advantageous as it allows a measure of changing risk through time, which can be used to monitor the long term impacts of management interventions (e.g. bycatch mitigation). This approach was also considered necessary as the MFish ERA differs from the other ERAs described here in that it encompassed both trawl and longline fisheries, and the likelihood of capture by any one fishing event differs greatly between fishing methods; therefore, an absolute metric was sought to compare the 'relative' contribution of risk of different fishing methods.

Of the 120 seabird species found in New Zealand waters, c. 60 species were excluded due to lack of data on distribution (though most of these were *Pterodroma* species and gulls, and thought unlikely to interact with fisheries). Sixty-three species were included in the analysis, although the final analysis reported on the 39 species that interact with longline and trawl fisheries; the remainder were excluded due to lack of data in the relevant fisheries (e.g. pot and gillnet). The assessment examined the risk and impact of fisheries with regard to the New

Zealand population of the species in question. For each species, an estimate was made of the number of birds killed per year, based on seabird distribution x fishing effort x *Vulnerability* per 0.1 degree square, where the *vulnerability* criterion was calculated on the basis of New Zealand observer data and seabird densities for each of 11 groups: 1) gannets; 2) gulls and terns; 3) large albatrosses *Diomedea*; 4) small albatrosses *Thalassarche* and *Phoebetria*; 5) large *Pterodroma* petrels; 6) *Procellaria* petrels; 7) other petrels; 8) large shearwaters; 9) small shearwaters 10) penguins; 11) shags. Small and large albatrosses were treated separately as there were sufficient data to determine specific rates of vulnerability to capture for these groups, but small shearwaters and petrels were grouped in the end, as data were inadequate to robustly describe a rate of capture at a finer taxonomic scale. For seabird distribution, only a range map was available for 24 of the species, and birds were assumed to be distributed homogeneously across the range. For 38 species, data were used from the NABIS database, which has three data layers per species, equating to 10% of the population (in the area of 100% NABIS distribution), 40% of the population (90% distribution) and 50% of the population (NABIS hotspot). For one species, tracking data were used.

Impact ratios were then calculated for each species, on the basis of the estimated number of birds killed in New Zealand fisheries, divided by an index of productivity. The latter was calculated as 0.5* Rmax * F (where F is between 0-1, based on IUCN Red List status), in an approach analogous to Potential Biological Removal (PBR) (Wade 1998, Dillingham and Fletcher 2011, Barbraud et al. 2009). A range of sensitivity tests were then conducted to assess uncertainties in the inputs and assumptions.

One of the benefits of the above approach to calculate absolute risk is that it can respond to changes in seabird catch in different fisheries through time. However, problems were recognised in relation to data availability: many species were excluded from the analysis, and frequently data substitutions were necessary, with around 1/3 of the values needed to calculate Rmax values being substituted. The PBR index was considered to be the best measure of relative vulnerability of each species to fisheries impacts, but was thought unlikely to represent an accurate measure of the number of individuals that can be removed from a population without causing a decline.

Western and Central Pacific Fisheries Commission (WCPFC) (Kirby et al. 2009, Filippi et al. 2010, Waugh et al. submitted)

In 2006, WCPFC established a 3 year program to develop a multi-taxa ERA. In the first year, results for seabirds were presented alongside other taxa. Later, the seabird risk assessment was developed separately. This focused on a productivity-susceptibility analysis, corresponding to Level 2 under the CSIRO framework. Seabird species were included in the analysis if any of the family had been recorded as bycatch. However, 192 species were subsequently excluded on the basis that: (1) they were considered unlikely to be caught (storm petrels and diving petrels), (2) there were no data on their distribution. In total, 70 species of albatross, petrel and shearwater were considered, of which 36 had been recorded as captured.

Two methods were used to estimate productivity. The first used Rmax, derived from age at first breeding and adult survival. Since data were missing for many species, substitutions were made from similar species (accounting for around 1/3 of all values). The second method developed the ICCAT *life history strategy* variable, weighting it by age of first breeding to create a *Fecundity Factors Index*. These two measures were found to be correlated, and the FFI was used in subsequent analysis on the basis that it relied on fewer assumptions.

Seabird distribution was estimated from range maps, foraging radii and remote tracking data, in which non-breeding birds were assumed to occupy the range map with a homogeneous distribution, and breeding birds were assumed to be distributed within a foraging radius from the breeding colony, with density decaying exponentially with increasing distance. Where foraging radii were unavailable, substitutions were made from other species in the genus of similar weight. Where tracking data were available, these were used to supplement the breeding and non-breeding distributions, and maximum density was selected. It was assumed that 50% of the total population consisted of breeders (40% for biennial-breeding species). Birds were considered to occupy breeding or non-breeding ranges according to the month, and composite maps were then produced by year quarter. Susceptibility was calculated as the product of the normalized species distribution and fishing effort per square kilometre, with fishing effort averaged across eight years (2002-2009), weighted by a *Vulnerability* factor, based on the observed mortalities from New Zealand observer data. Risk was calculated as susceptibility divided by productivity. The distribution of risk was then analysed by area, season, species, and fishing fleet (flag state):

a) The risk scores for species-fishery interactions were mapped (noting that single species maps could also be produced by this method), to give an overall 'risk-map' for the study area. These were presented as average annual maps, quarterly maps, and a quarterly maximum. Six risk ratings from high to negligible were calculated by dividing the normalized risk scores into five categories including similar numbers of species, with the negligible level set very low (<0.01 out of a range of 0 – 1) to remove noise from the lower end of the scale.

b) Risk scores by fishery area and species were summed, and a species ranking calculated. This showed which species were most at risk from longline fisheries interactions at the population level.

c) Risk scores for all species and areas were calculated by fishing fleet and used to determine which fleets posed the greatest risk across species.