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Currently, 17 of 22 albatross species are listed as Vulnerable, Endangered, or Critically Endangered by the International Union for the Conservation of Nature (IUCN). Incidental mortality in fisheries is by far the most widespread cause of the population declines observed for these and other closely related species. In 2006, the International Commission for the Conservation of Atlantic Tunas (ICCAT) requested an assessment of the threat from their fisheries to all seabirds that breed or forage within their jurisdiction. Methods were developed to assess the potential consequences of fishing for more than 60 populations of seabird. The assessment framework involved the identification of at-risk populations, overlap analyses, estimation of total bycatch, and an evaluation of the impact of the bycatch on key selected populations for which there were sufficient data on bird distribution and demography. These were the wandering and black-browed albatrosses of South Georgia, and the Atlantic yellow-nosed and Tristan albatrosses of Gough Island. Summary results from the seabird assessment are presented, revealing that ICCAT longline fisheries catch substantial numbers of seabirds, with potentially significant conservation implications. If this mortality is not reduced, the numbers of breeding birds in some populations will continue to decline, threatening their long-term viability.

Keywords: Atlantic Ocean, ecological risk assessment, incidental mortality, longline, seabirds, trawl.

Introduction

The incidental mortality of seabirds during fishing operations, including pelagic longlining for tuna and tuna-like species, has been recognized as a threat to the long-term viability of many seabird populations, particularly albatrosses and petrels (Gales, 1993; Weimerskirch and Jouventin, 1997; Croxall *et al.*, 1998). Seabirds are attracted to baited longline hooks and discharged offal, and can drown if they swallow the hooks or become snagged. The extensive foraging distributions of pelagic seabirds frequently overlap with multiple fisheries, many of which have poor or non-existent bycatch-mitigation strategies. The resulting risk to seabirds from fishery interactions has led to the establishment of several international conservation agreements. Notable among these are those negotiated through the United Nations (UN) Food and Agriculture Organization (FAO) – the International Plan of Action (IPOA) for Reducing Incidental Catch of Seabirds in Longline Fisheries, and the Agreement on the Conservation of Albatrosses and Petrels – in addition to the articles within the UN Convention on the Law of the Sea, i.e. the Convention on Biodiversity, the Convention on Migratory Species, and the UN Fish Stocks Agreement. These urge or require States to minimize the impact of fisheries on non-target species. There is therefore an established framework of international fishery and environmental legislation that not only recognizes, but requires the adoption of, approaches integral to ecosystem-based fishery management (Smith *et al.*, 2007; Hobday *et al.*, 2011).

The International Commission for the Conservation of Atlantic Tunas (ICCAT) was established in 1969 and is responsible for the conservation of tunas and tuna-like species in the Atlantic Ocean and adjacent seas. Some 30 such species are considered by ICCAT, which compiles statistics from member states, coordinates research, and develops management advice relating to target and bycatch (principally shark) species.

In 2002, recognizing the FAO IPOA for Reducing Incidental Catch of Seabirds in Longline Fisheries, and the need to evaluate the incidental mortality of seabirds in their fishery, ICCAT passed a resolution (Res 02–14) that (i) urged member nations to implement national plans of action for seabirds, (ii) encouraged them to collect and provide information on interactions with seabirds in all fisheries under the purview of ICCAT, and (iii) initiated an assessment of the impact of the incidental catch of seabirds taken by all vessels fishing for tuna and tuna-like species in the Convention Area. In anticipation of further improvements on completion of the assessment, a recommendation (Rec 07–07) was implemented that required longline vessels south of 20˚S to use bird-scaring lines.

Here we describe the seabird assessment framework that was developed, the results of the assessment, and the subsequent recommendations made by the ICCAT Subcommittee on Ecosystems. This was the first time an assessment of this magnitude had been attempted; it encompassed the Mediterranean Sea, and the North and South Atlantic Ocean, assessed more than 60 seabird populations, and covered fishing fleets from >30 nations that use multiple gears to target valuable shelf, slope, and pelagic species of fish. It required the collaboration of seabird ecologists, fishery administrators and data managers, mathematical modellers, and statisticians.

Methods

An approach that has been successfully applied in the assessment of fishery impacts on target and non-target species is the Ecological Risk Assessment (ERA) framework developed by Smith *et al.* (2007) and Hobday *et al.* (2011). Given the potentially large number of seabird species requiring assessment, and the variable quantity and quality of the available data, the staged or hierarchical approach of an ERA was considered appropriate for the ICCAT seabird assessment.

The multilevel framework of an ERA moves from a comprehensive but largely qualitative risk analysis at the lower levels, through a more focused and semi-quantitative approach, to a fully quantitative model-based methodology at the highest level. This is efficient because many minimally affected species are screened out at the lowest levels, so the more intensive analyses are limited to high-risk seabirds. The ERA framework allows rapid identification of high-risk species and potentially detrimental fishing activities, which in turn can lead to immediate remedial action (risk-management response) without the need for a full quantitative assessment. The approach is also precautionary, in the sense that risks may be scored high in the absence of information, evidence, or logical argument to the contrary.

The seabird assessment framework was developed with input from many experts. The first phase related to data gathering, mapping, and summation (Objectives 1–4), and the second to the development and application of models for assessing impacts on seabird populations (Objectives 5 and 6). The six objectives of the assessment (described in more detail in the subsections beneath) were:

(1) identify the seabird species most at risk from fishing in the ICCAT Convention Area;
(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 (ICCAT longlining);
(4) review the existing estimates of bycatch rates for ICCAT longline fisheries;
(5) estimate the total annual seabird bycatch in the ICCAT Convention Area;
(6) assess the likely impact of this bycatch on seabird populations.

Objective 1

The identification of the seabird populations most likely to be at risk from ICCAT longlining was a key objective of the ICCAT risk assessment. All species and populations of seabirds recorded as bycatch in ICCAT longline fisheries were considered, along with any additional species that although unrecorded were closely related in both taxonomic and ecological terms, so were deemed to be susceptible. Although each population could have been ranked according to the degree of risk based on expert knowledge of their biology, behaviour, and bycatch rates, a semi-quantitative method was preferred that could formalize this in a repeatable and impartial manner, and be subsequently verified using expert opinion. The risk priorities followed the Productivity Susceptibility Analysis (PSA) methods advocated by Hobday *et al.* (2011). This analysis characterizes risk as a function of the productivity of a population and its susceptibility to capture. The measure of productivity was based upon life-history strategy, specifically the frequency of breeding and clutch size. Although other measures of productivity were considered, such as age-at-first-breeding and adult survival, the selected life-history features were believed to be sufficient for purpose.

The productivity measure and scores were (a) life-history strategy: biennial breeder, single-egg clutch = 3, annual breeder, single-egg clutch = 2, annual breeder, multiple-egg clutch = 1. The measures of susceptibility and their scores were (b) global IUCN status: Critically endangered/Endangered = 3, Vulnerable = 2, Near Threatened = 1, and Least Concern = 0; (c) breeding population status: rapid decline (>2% per year) = 3, decline = 2, stable = 1, increase = 0; (d) degree of overlap with ICCAT fisheries: high = 3, medium = 2, low = 1; (e) behavioural susceptibility to capture: high = 3, low = 1. The last was based on the tendency of seabirds to follow fishing vessels, and the relative incidence of bycatch in ICCAT or other fisheries.

A precautionary approach was taken where data were lacking or were uncertain, the highest (risk) score being assigned in those cases. Relative risk was then calculated as the Euclidian distance to the origin of productivity measure (a) and the arithmetic mean of susceptibility measures (d) and (e). Populations were then ranked by risk score, with the high-risk category being >3.16, i.e. approximately one-third of all populations, according to Hobday *et al.* (2011).

Objective 2

Information on species distribution at some level (from the extent of at-sea range, to more-detailed density distributions based on year-round tracking of birds of different age and status) was a prerequisite for most of the analyses undertaken as part of the assessment. Seabird distribution depends on the age of the bird, its breeding status, and the stage of its breeding cycle. The distribution changes dramatically in most species from breeding to non-breeding periods (Phillips *et al.*, 2006, 2008). Although most albatrosses and large petrels have been tracked from at least one colony during the breeding season, data on juveniles, deferring breeders, and birds of any age during the non-breeding season are often lacking (BirdLife International, 2004). Indeed, for most seabird species in the Atlantic Ocean, of which the albatrosses and large petrels constitute a minority, no tracking data are available. For most analyses, distributions were therefore based on a combination of range maps presented in bird-identification guides, and foraging radius during breeding based on tracking data (usually, but not always from the focal population). For most species, the foraging radius varies significantly with breeding stage, and is greater during incubation than chick-rearing (Phillips *et al.*, 2004, 2006). Given that the overlap with fisheries is therefore likely to be greatest during incubation, and based on the precautionary principle, the average maximum range during that stage was used in the analysis.

Populations were assumed to consist of 70% adults, 10% pre-breeders (immature birds not breeding, but returning to the colony for some part of the breeding season), and 20% juveniles (immature birds from fledging until the first return to the colony as pre-breeders). Birds during the non-breeding season and juveniles throughout the year were assumed to be evenly distributed across the entire range of the species. Adults and pre-breeders during the breeding season were assumed to be restricted to and evenly distributed within the foraging radius from the colony. In the case of biennial breeders, 50% of the birds were assumed to have a non-breeding distribution during the breeding season.

Where a population identified as at risk included birds from more than one island, the distributions of birds during the breeding season, i.e. within the relevant foraging radius, was weighted by the number of breeding pairs at the respective sites. Species grids were created in a similar fashion.

Objective 3

For the purposes of the overlap analysis, the ICCAT area was defined as the 5×5^{o} grid cells for which longline fishing was reported during the years 2000–2005. Effort data, presented as the number of hooks set for a particular quarter in these cells, were obtained from the ICCAT Secretariat. From this dataset, the average number of hooks set in each grid cell for each month during the period 2000–2005 was calculated. The following overlap measures with seabird distributions were calculated for each month:

(i) for each population, the percentage distribution within the area of ICCAT longline effort;

(ii) for each population, the product of the percentage distribution and the average number of longline hooks set within each 5×5^{o} grid square;

(iii) for each species, the percentage of ICCAT longline effort within its range.

Although indicative of the possible encounter rate, the overlap indices do not consider susceptibility to capture. Populations may have a large degree of overlap, but this does not necessarily imply a large bycatch; of course, the inverse may also be true (see Objectives 5 and 6).

Objective 4

The review of seabird bycatch rates in ICCAT and other relevant fisheries took account of data quality and whether there was sufficient detail in the reported methodology to determine whether values were reliable. These were important considerations, because a lack of data from some fisheries and limited observer coverage in others clearly reduces the reliability of any estimates obtained as part of Objective 5. In addition, ICCAT members were encouraged to provide unpublished seabird bycatch data for the assessment.

Objective 5

Several methods have been used to estimate the seabird bycatch from specific fisheries (Klaer and Polacheck, 1997; Lewison *et al.*, 2004). However, the aim of Objective 5 was to integrate results from published and unpublished bycatch studies across the entire Atlantic Ocean. This meta-analysis took bycatch-rate information, where available, raised by fishing effort to provide an ocean-wide estimate of bycatch. Species-specific bycatch totals were also calculated when the relevant data were available. For regions where bycatch data were unavailable, assumptions were made to fill these gaps.

The region considered was the maximum geographic extent of ICCAT pelagic-longline fishing, based on fishing-effort data obtained from ICCAT. Pelagic-longline bycatch rates, by population if possible, from individual studies were then mapped as appropriate onto this region, given knowledge of the spatial distribution of each fishery. Where bycatch-rate data were missing for particular grid squares, 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. Most of the relevant bycatch studies were published in the past decade, so these analyses were based on pelagic-longline fishing effort carried out within the jurisdiction of ICCAT from 2003 to 2006.

Objective 6

Following the ERA approach advocated by Hobday *et al.* (2011), once populations have been ranked in order of potential risk through the productivity–susceptibility analysis, a more detailed quantitative assessment of high-risk species may be warranted. Whereas the overlap analysis (Objective 3) and the estimation of total bycatch (Objective 5) provide information on the likelihood of encounter, and the potential magnitude of annual bycatch, they cannot elucidate the historical effects of incidental take on populations, nor the long-term implications of continued fishery-related mortality (Tuck *et al.*, 2001; Arnold *et al.*, 2006; Rolland *et al.*, 2009).

The populations chosen for quantitative assessment in the ICCAT seabird assessment were determined according to their risk level (Objective 1), and the quantity and quality of data available for the modelling process (Objectives 2–5). Although vulnerable populations, for which minimal data exist on demography and fishery interactions, still require some management response, given the uncertainties associated with model outcomes, complex modelling for such species is less likely to produce results of practical use for fishery managers. Hence, just a few populations were included in this, the last and most detailed component of the assessment process.

The assessment model has components covering population dynamics, fishery bycatch, and estimation procedures. It caters for annual and biennial breeding schedules. Birds are categorized as actively breeding adults, those failing to breed in that year, non-breeding adults that were either successful or unsuccessful in their previous breeding attempt, juveniles, or chicks. The model is sex-disaggregated, and specifies the at-sea distribution of birds at each life stage in each month of the year. The quantity of birds caught is modelled as a function of fishing effort, bird numbers, their catchability by each fleet, and the spatial overlap of birds and fisheries. The estimated parameters are the fleet catchabilities (relating fishing effort to bycatch), a density-dependent chick mortality, the pre-fishing breeding success rate (chicks fledged or eggs laid), and the population size. A statistical best fit is then made between the observed and model-estimated annual breeding population size, numbers of fledglings, adult and juvenile survival rates, observed bycatch rates and, where available, the age distribution of the population (Tuck *et al.*, 2001; Thomson *et al.*, 2009).

Results

In all, 68 seabird populations were considered, comprising a total of 41 species (Anon., 2008). Of these, 22 were designated high priority across all risk-score methods, and 41 according to at least one method of prioritization. Of these 41 populations, 21 are known or suspected to be declining in abundance. The albatrosses from the Tristan de Cunha group and South Georgia had the highest risk scores. Populations of grey petrel (*Procellaria cinerea*), Balearic shearwater (*Puffinus mauritanicus*), white-chinned petrel (*Procellaria aequinoctialis*), southern giant petrel (*Macronectes giganteus*), and Mediterranean Cory’s shearwater (*Calonectris diomedea*) also had a high priority ranking. All these populations have a great degree of overlap with ICCAT longline fisheries and high susceptibility to capture. Sooty (*Phoebetria fusca*), Tristan (*Diomedea dabbenena*), wandering (*D. exulans)*, and grey-headed (*Thalassarche chrysostoma*) albatrosses are biennial breeders, so are particularly vulnerable to incidental mortality.

Foraging distributions varied by species, from the rather restricted range of southern giant petrels from Argentina on the Patagonian shelf, to the highly extensive distributions of most albatrosses and the white-chinned petrel. The Atlantic yellow-nosed (*T. chlororhynchos*) and Tristan albatrosses forage almost exclusively within the Atlantic Ocean, but the wandering and black-browed (*T. melanophris*) albatrosses of South Georgia, though foraging within the Atlantic Ocean, also spend considerable time elsewhere.

The often striking differences in distribution clearly have conservation implications when considering the impact that ICCAT's longline fisheries, and fisheries in other regions, are likely to have on each population (see the more-detailed analyses below). Although at some stages in the implementation of the assessment there were attempts to incorporate distribution data from other sources, the analyses considered most reliable were those involving the 22 seabird populations (10 species) for which tracking data were available (Table 1).

The bird distributions were compared with data on fishing effort by ICCAT vessels in each 5×5° grid square, by month, obtained from ICCAT. In all, data from 17 nations were identified by source, leaving just 17% of the global effort data within the category "other". The main longline fleets operating in the Atlantic Ocean are those of Japan, Taiwan, and to a lesser extent Brazil and Spain; in addition, Korean effort was high in the 1970s but has dropped since 1990.

South of 20˚S, where albatrosses and petrels are dominant, 52 million hooks were reported to ICCAT in 2006 (Figure 1). Demersal longline and trawl fisheries targeting shelf and slope species including Patagonian toothfish *Dissostichus eleginoides*, ling *Genypterus blacodes*, kingklip (*G. capensis*), and hake *Merluccius* spp. also operate within the Atlantic Ocean, but are not managed by ICCAT. Major demersal-longline fishing nations include Brazil, Uruguay, Argentina, Namibia, and South Africa (Tuck *et al.*, 2003). Notable trawl fleets operating within Atlantic waters are those of Namibia, South Africa, the Falkland Islands, Argentina, and Uruguay.

Results from the three overlap measures described under Objective 3 above indicate a high degree of overlap with Cory's shearwater, Atlantic yellow-nosed, and Tristan albatrosses, with >75% of their year-round distribution within the area of ICCAT longline fishing [overlap measure (i)]. Likewise, the percentage of ICCAT effort within the distribution of Cory's shearwater [overlap measure (iii)] is high throughout the year. The populations showing the greatest average overlap across all months according to the product of the percentage distribution and the average number of longline hooks set within each 5×5° grid square [overlap measure (ii)] were Cory's shearwater, Atlantic yellow-nosed albatross, Cape gannet (*Morus capensis*), Tristan and sooty albatross (Table 1). For albatrosses and petrels, the greatest overlap with ICCAT longline fisheries was during the months March–August.

Figure 2 compares the distribution in January and July of the 22 seabird populations for which sufficient data (Table 1) with the corresponding distribution of longline effort were available. Selection based upon data availability can bias interpretation of this relationship because it does not represent all populations equally: albatrosses and petrels are well represented, whereas populations in the Mediterranean Sea and North Atlantic are under-represented. In addition, some high-risk populations (Balearic shearwater, southern giant petrel, and grey petrel) and those inhabiting the central Atlantic Ocean (Atlantic petrel, great shearwater, and great-winged petrel) are not included. However, there is clearly a broad overlap between ICCAT longline fisheries and seabird distributions, with high densities of birds (and overlap) south of 20˚S and within the Mediterranean Sea. More southerly distributions overlap considerably with demersal longline and trawl effort, in particular off the Patagonian shelf and southwestern Africa (Tuck *et al.*, 2003).

In all, 37 species of seabird have been recorded as bycatch in ICCAT fisheries (Anon., 2008). Several papers have documented substantial bycatch rates in Atlantic Ocean pelagic-longline fisheries (Cuthbert *et al.*, 2005; Laich *et al.*, 2006; Bugoni *et al.*, 2008; Petersen *et al.*, 2008; Huang *et al.*, 2009; Jiménez *et al.*, 2009). Bycatches have also been reported in demersal-longline (Laich *et al.*, 2006; Otley *et al.*, 2007; Bugoni *et al.*, 2008; Petersen *et al.*, 2008) and trawl fisheries (Sullivan *et al.*, 2006; Gonzalez-Zevallos *et al.*, 2007; Petersen *et al.*, 2008; Watkins *et al.*, 2008). The species composition of bycatch depends upon the region, the time of year, and the operational characteristics of the vessel. Major bycatch species in the southern Atlantic Ocean are wandering albatross, Tristan albatross, black-browed albatross, Atlantic yellow-nosed albatross, shy-type albatrosses (*Thalassarche cauta* and *T. steadi*), grey-headed albatross, and white-chinned petrels. Fewer data are available from the North

Atlantic Ocean. Species documented as bycatch there include Cory’s shearwater, Balearic shearwater, Yelkouan shearwater (*Puffinus yelkouan*), and northern fulmar (*Fulmarus glacialis*).

Seabird bycatch rates from the pelagic longline fleets operating within the Atlantic Ocean vary considerably (Table 2). There is some evidence that bycatch rates have reduced over time through better awareness and mitigation measures, but the paucity of comprehensive studies across the major distant-water fleets and the Mediterranean Sea is cause for concern. For example, countries known to engage in longline fishing in the Mediterranean Sea, but for which no seabird-bycatch data were available, included Algeria, Cyprus, France, Greece, Italy, Japan, Korea, Libya, Malta, Morocco, Taiwan, Tunisia, and Turkey (Cooper *et al.*, 2003). Bugoni *et al.* (2008) provide a comprehensive summary of bycatch rates for fisheries operating in the southwestern Atlantic Ocean.

The estimated total seabird bycatch from ICCAT longline fisheries has declined from some 16 500 seabirds in 2003 to 12 000 birds or less in subsequent years (Anon., 2010). This decline is attributed to both a drop in fishing effort and a shift in fishing distribution to more northerly latitudes, reducing overlap with several albatross and petrel species.

On a per-species basis, the greatest proportion of bycatch that could be identified to species level was that of black-browed albatrosses (32%), followed by Atlantic yellow-nosed albatrosses (17%). These populations suffered an average annual bycatch of 3900 and 2000 birds, respectively, between 2003 and 2006. Unspecified albatrosses accounted for an additional 6%, and other unspecified seabirds made up 42% of the total.

Because of the extensive foraging ranges of the birds and their known interaction with multiple gear types, effort statistics for all key fisheries that may be impacting seabirds are needed for a comprehensive assessment to be made. As such, fleets in waters other than the Atlantic Ocean, and those using gears other than pelagic longline, were also considered.

Although generally comprehensive, the effort data of some nations were incomplete, poorly maintained, not publicly available, or in some cases non-existent. In those cases, effort data were modelled using auxiliary information, such as target-fish catches, catch rates, or numbers of vessels. Nonetheless, the modelled effort data may be incomplete, e.g. for the Brazilian small-scale hook-and-line fleet (Bugoni *et al.*, 2007). Fishing-effort data were broadly categorized into one of four superfleets based on similar physical and operational characteristics. These consisted of pelagic-longline fleets, regulated demersal-longline fleets, demersal-longline fleets engaged in IUU (illegal, unregulated, and unrecorded) fishing, and trawl fleets.

Four populations were chosen as candidates for population modelling. These four were of great concern in the prioritization process, and had sufficient data available on bird distribution and demography. The populations chosen were the wandering and black-browed albatrosses of South Georgia, and the Tristan and Atlantic yellow-nosed albatrosses of Gough Island. Only a preliminary exploration of the data and models for Tristan albatross was conducted, however.

The ability of the model to reproduce the demographics of each population varied. For the black-browed albatross population of South Georgia, which is wide-ranging over the southern Atlantic Ocean, with non-breeding and juvenile birds also foraging off eastern Australia, agreement between the predicted number of breeding pairs and the available census data was good (Figure 3a). Census data show that this population has halved in just two decades, from >150 000 breeding pairs in the mid-1980s to 70 000 – 75 000 in recent years, with substantial bycatches noted from trawl, pelagic- and demersal-longline fisheries. The Atlantic yellow-nosed albatross population of Gough Island is largely restricted to the southern Atlantic Ocean, and declined from some 7000 breeding pairs during the 1980s before recovering in the late 1990s (Petersen *et al.*, 2008). The population model provided reasonable fits to the observations (Figure 3b), but the model was unable to match a recently observed increase in the number of breeding pairs, without downweighting estimates of juvenile survival (Anon., 2010). Fits to the wandering albatross population of South Georgia, which is extremely wide-ranging across the Southern Ocean and is known to interact with longline fisheries both within the ICCAT region and elsewhere, were satisfactory (Figure 3c). The lack of fit to observed data for these models could be due to the highly stochastic nature of bycatch, or a poor match between fishing-effort data and the mortality caused by a fleet or a component of a fleet (Anon., 2010; Tuck, 2011).

The models showed that, of the three populations, the Atlantic yellow-nosed albatross population was most productive and therefore most likely to recover following a reduction in bycatch (Anon., 2010). Both the wandering and black-browed albatross populations showed negligible estimated density-dependence, however, so the model indicates that any additional mortality above that experienced naturally is unsustainable by these populations.

Discussion

The increasing concern over the threats posed by fisheries to non-target species, communities, and habitats has led to several internationally binding agreements that aim to ensure that fishers demonstrate greater environmental accountability (FAO, 2008). As part of this process, ERAs, though still at an early stage of development, are being used increasingly to identify and quantify these impacts (Small *et al.*, 2010; Hobday *et al.*, 2011). The strengths of an ERA are its hierarchical approach, the inclusion of precautionary principles, and the capacity to incorporate management responses at each stage of the process. Recent examples dealing with seabirds include those applied by the Commission for the Conservation of Antarctic Marine Living Resources (CCAMLR), New Zealand, the Western Central Pacific Fisheries Commission (WCPFC), and ICCAT (Waugh *et al.*, 2008; Kirby *et al.*, 2009; Arrizabalaga *et al.*, 2011). Although broadly adopting the form advocated by Hobday *et al.* (2011), these ERAs differ in their scope and eventual management response (Small *et al.*, 2010); the ICCAT seabird assessment followed the methods suggested by Hobday *et al.* (2011) more closely than the other ERAs carried out to date.

The ICCAT seabird assessment demonstrated the advantages of undertaking an ERA, as highlighted by Small *et al.* (2010). Succinctly, it identified gaps in both fishery and seabird data (e.g. in spatio-temporal distributions and observer coverage), identified the species most at risk from fishing using a semi-quantitative framework that is readily updateable as new information becomes available, identified fisheries, seasons, and areas of high bycatch, and provided a unified and focused study that enabled issues to be discussed and addressed with fishery managers in a more systematic manner than would have been possible otherwise.

The six objectives of the ICCAT seabird assessment moved from initial data-collection and prioritization through to a specific population-level assessment of impacts. The prioritization of species and populations of greatest concern followed the PSA methods suggested by Hobday *et al.* (2011). The populations with the highest risk ranking were the albatrosses of South Georgia and Gough Island (Anon., 2008). These were populations with recorded observations of incidental fishing mortality, a great degree of fishery overlap, and historical declines in breeding population size (Croxall *et al.*, 1998; Cuthbert *et al.*, 2003, 2005; Phillips *et al.*, 2005; Arnold *et al.*, 2006; Wanless *et al*., 2009). Although a degree of subjectivity in some elements was unavoidable, quantifying the productivity and susceptibility categories provided a scientific, transparent, and defensible means of identifying populations at risk. The assignment of high scores to populations that lacked information was precautionary (as advocated by Hobday *et al.*, 2011), and could possibly have led to a higher risk ranking than necessary. However, as further studies on bycatch become available, these rankings can be adjusted (which, of course, allows the risk scores of populations to increase as well as to decrease).

The overlap analysis indicated the potential for seabirds to encounter pelagic-longline hooks within the Atlantic Ocean. Unfortunately, for many seabird populations, data were not available to specify confidently the spatio-temporal distributions of all breeding stages. Similarly, although the fishing-effort data maintained by the ICCAT Secretariat were extensive both temporally and spatially, and appeared to be reliable, those from other fishery agencies were not necessarily of the same quality (Tuck *et al.*, 2003). For some nations, statistics on the magnitude and spatio-temporal distribution of fishing effort were compromised by a lack of robust estimates, or indeed any public estimates at all. As such, the number of hooks set or trawl hours reported for those nations are likely to underestimate substantially the true level of effort being deployed. In order to quantify fishery interactions and facilitate better management outcomes, cooperation and transparency between fishery agencies and analysts needs to be improved. Despite these limitations, the assessment clearly demonstrated major overlaps between the extensive foraging distributions of seabirds in the Atlantic Ocean and ICCAT longline fisheries.

The estimation of total seabird bycatch, population-specific where possible, indicated that large and potentially unsustainable numbers of seabirds are being caught by longline vessels in the Atlantic Ocean (Anon., 2010). The estimation process was, however, hampered by inadequate observer coverage of most fleets. There was a lack of information on the bycatch composition by species or population, and in some cases poor spatial and temporal coverage, e.g. of high-sea fleets (see Huang *et al.*, 2009). Where data on bycatch rates were unavailable, those from associated fleets/areas/seasons were used. The results of this study therefore highlight the need for improved observer coverage of all national fleets operating within the Atlantic Ocean. Although the uncertainty in the total bycatch is statistically unquantified, the magnitude of our best estimate clearly indicates the potential for substantial population impacts. For example, the viability of the wandering albatross population in South Georgia is clearly in question, given the estimated bycatch (150 birds annually) and the consequent impact on breeding success, relative to the overall population size, which dropped from 2230 breeding pairs in 1984 to an estimated 1383 pairs in 2011 (Poncet *et al.*, 2006; British Antarctic Survey, unpublished data).

Overlap studies and estimates of total bycatch cannot determine the direct impacts that a fishery may have on a population, so the assessment also included quantitative modelling of a few high-risk populations. Although the models did not always fit every aspect of the observed data well, given the inadequacy of presently available data, they did demonstrate the major impacts of fishing (for all gear-types) and highlighted the unsustainability of current bycatch levels. The low density-dependence in these long-lived populations suggests that they have little ability to recover from mortality above that which they would experience naturally.

As a result of the seabird assessment, the ICCAT Ecosystems Subcommittee agreed that ICCAT fisheries do impact populations of seabirds, including some that are threatened with extinction, and that reducing the fishery-related seabird mortality would improve population status. Various recommendations were made with regard to improving observer coverage, data collection to estimate bycatch rates, and on-board mitigation. During the assessment, seabird bycatch-awareness material was produced and disseminated to various parties. The subcommittee also encouraged further research and assessment. In particular, the Standing Committee for Research and Statistics recommended that ICCAT should, at a minimum, require Contracting Parties to use bird-scaring lines in combination with at least one other effective mitigation measure throughout the Convention Area (not just south of 20˚S), until it can be demonstrated that bycatches of seabirds are insignificant. Such recommendations would afford appreciable protection for the four species considered in detail, as well as reducing the risk to others for which data are limiting. The recommendations of the Committee were not endorsed by the ICCAT Commission in 2009, but remain on the table for consideration in the future.

A key need in future ERA applications is an explicit link between the outcomes of the assessment and agreed management responses (Hobday *et al.*, 2011). It was unfortunate that a clearer link did not exist in the present case, considering that assessments of this nature require considerable resources, which are difficult to obtain despite the value of fisheries and the clear conservation concern for seabirds on a global scale.

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Figure legends

Figure 1. The annual number of hooks deployed south of 20˚S and reported to ICCAT for Japan, Taiwan, and all other nations.

Figure 2. The overlap of ICCAT pelagic longline-fishing effort with the combined distribution of 22 populations (10 species) of seabird for the months January (left) and July (right). Longline fishing effort (millions of hooks) averaged over the years 2000–2005 is shown proportional to the diameter of the circle (see key). Contours of seabird density (numbers per degree square) give equal weight to each of the ten species and are illustrated as relative density. Darker shades (of brown) depict a greater density of birds

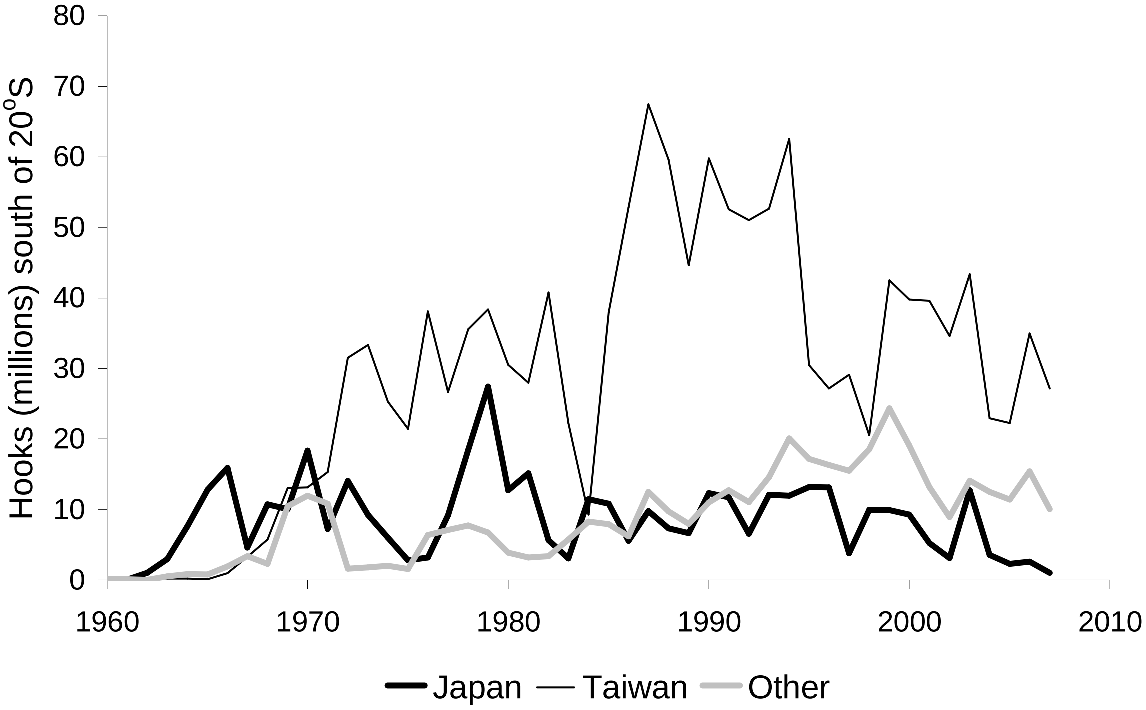
Figure 3. Model-estimated (line) and observed (points) numbers of breeding pairs for (a) the South Georgia black-browed albatross, (b) the Gough Island Atlantic yellow-nosed albatross, and (c) the South Georgia wandering albatross.

Table 1. Values (ordered by average score) for overlap score (ii), the product of the percentage seabird distribution and the average monthly ICCAT pelagic-longline hooks set per 5×5˚ grid square between 2000 and 2005, for the months of January and July, and the average over all calendar months.

Species	**Breeding colony**	**January**	**July**	**Average**
Cory’s shearwater	Mediterranean	160 408	172 790	155 082
Atlantic yellow-nosed albatross	Tristan de Cunha	26 934	105 297	76 062
Cape gannet	Namibia/South Africa	22 092	59 864	52 905
Tristan albatross	Gough	8 672	67 169	46 633
Sooty albatross	Tristan de Cunha	6 970	29 808	25 474
Sooty albatross	Indian Ocean	2 600	29 808	15 971
White-chinned petrel	South Georgia	1 316	10 820	9 981
Black-browed albatross	South Georgia	421	13 380	8 381
White-chinned petrel	Prince Edward	1 181	10 820	7 322
Wandering albatross	South Georgia	1 006	8 501	6 501
Black-browed albatross	Falklands	358	13 380	5 645
Black-browed albatross	Crozet	358	13 380	5 596
Black-browed albatross	Kerguelen	358	13 380	5 596
White-chinned petrel	Crozet	301	10 820	5 453
White-chinned petrel	Kerguelen	301	10 820	5 453
Wandering albatross	Crozet	1 002	8 273	5 398
Wandering albatross	Prince Edward	1 002	8 273	5 398
Grey-headed albatross	South Georgia	315	3 288	4 362
Grey-headed albatross	Prince Edward	483	3 288	4 234
Grey-headed albatross	Crozet and Kerguelen Is.	311	3 288	3 211
Grey-headed albatross	Chile	311	3 288	3 212
Southern giant petrel	Argentina	103	2 677	2 976

Table 2. Seabird bycatch rates reported for pelagic-longline fisheries in the Atlantic Ocean.

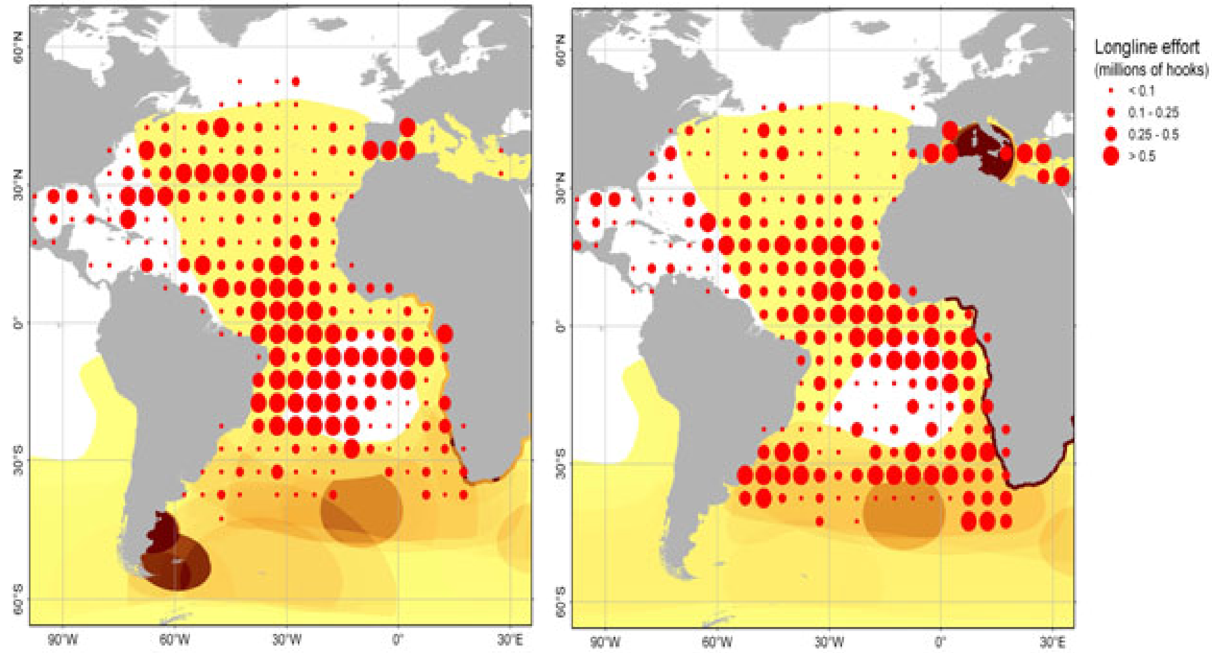
Country of fishery	**Average bycatch rate per 1000 hooks**	**Data collection period**	**Source**
Brazil	1.35	1987–1990	Vaske (1991)
Brazil	0.12	1994–1995	Neves and Olmos (1998)
Brazil	0.09	2000–2005	Neves *et al.* (2007)
Brazil	0.13	2001–2006	Bugoni *et al.* (2008)
Canada	0.004–0.011	2001	Anon. (2007)
Japan	0.31	2001–2002	Kiyota and Takeuchi (2004)
Namibia	0.07	2004–2006	Petersen *et al.* (2008)
South Africa (foreign)	2.6	1998–2000	Ryan *et al.* (2002), Petersen *et al.* (2008)
South Africa (domestic)	0.8	1998–2000	Ryan *et al.* (2002), Petersen *et al.* (2008)
South Africa (foreign)	0.51	1998–2005	Petersen *et al.* (2008)
South Africa (domestic)	0.23	1998–2005	Petersen *et al.* (2008)
Spain	0.25	1998	Belda and Sanchez (2001)
Taiwan	0.037 (south of 25˚S)	2002–2004	Chang *et al.* (2008)
Uruguay	4.7	1993–1994	Stagi *et al.* (1998)
Uruguay	0.42	1998–2004	Jiménez *et al.* (2009)
Uruguay	0.26	1998–2006	Jiménez and Domingo (2007)



80
70
60
50
40
30
20
10
0
Hooks (millions) south of 20^{o}S
1960
1970
1980
1990
2000
2010
Japan
Taiwan
Other

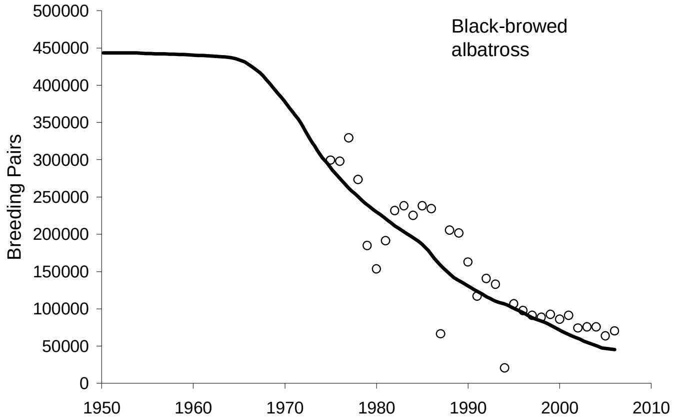
Figure 1

Figure 2

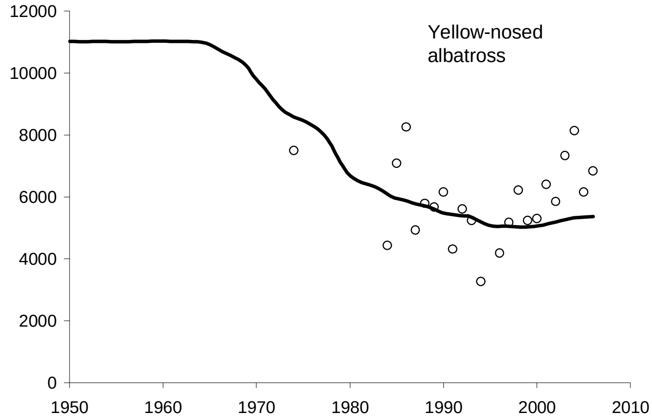


Longline effort
(millions of hooks)
< 0.1
0.1 - 0.25
0.25 - 0.5
> 0.5
60°N
30°N
0°
30°S
60°S
90°W
60°W
30°W
0°
30°E

(a)


Black-browed
albatross
Breeding Pairs
500000
450000
400000
350000
300000
250000
200000
150000
100000
50000
0
1950
1960
1970
1980
1990
2000
2010

(b)


Yellow-nosed
albatross
12000
10000
8000
6000
4000
2000
0
1950
1960
1970
1980
1990
2000
2010

(c)

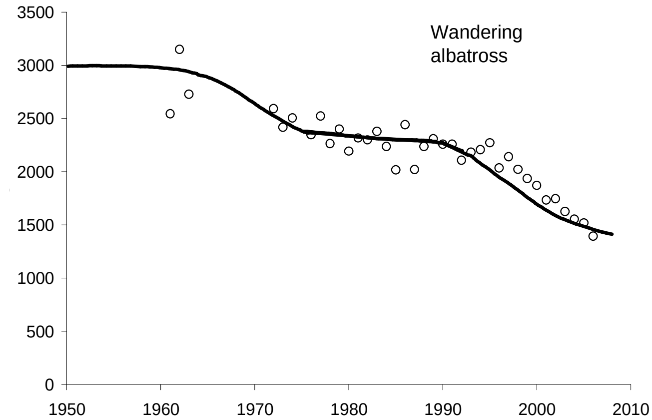

Wandering
albatross
3500
3000
2500
2000
1500
1000
500
0
1950
1960
1970
1980
1990
2000
2010

Figure 3