Ecological Risk Assessment for seabird interactions in surface longline fisheries managed under the Convention for the Conservation of Southern Bluefin Tuna

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Abstract

An analysis of risk of seabird interactions with surface longline fisheries was undertaken in 2012 using fishing data from the Commission for the Conservation of Southern Bluefin Tuna (Commission[1]), and biological and spatial data indicative of the distributions of a suite of albatross and petrel species known or likely to be captured or killed in the Convention for the Conservation of Southern Bluefin Tuna (CCSBT) fisheries; in this paper we update that analysis using improved spatial seabird distribution data layers utilising all available satellite tracking data for the same group of species. The analysis adapted methods developed in other regions and applied to assess risk of incidental mortality of highly migratory top predator species in other Regional Fisheries Management Organisations. Seabird species included in the analysis include rare species, such as Amsterdam Albatross, listed as Critically Endangered by the IUCN, and globally distributed species, such as white-chinned petrels (listed as Vulnerable by the IUCN) and sooty shearwater (Near Threatened). Simple representations of species spatial distributions were used in the first instance, with hotspots of activity defined around breeding localities for each species. These distributions were combined with spatial fishing effort data to define risk as a function of spatial overlap between these distributions on a seasonal (quarterly) basis. Risk is then a function of spatial overlap, species vulnerability to capture in longline fisheries, and species biological productivity. Results indicate that species at highest risk are primarily large albatrosses at temperate latitudes, followed by smaller albatrosses. Geographical areas of highest risk include the Tasman Sea and the area around New Zealand, primarily in the austral autumn and winter. The analysis has improved on previous work by utilising species spatial distribution information derived from satellite telemetry. Further improvements in the long term can be achieved by collecting fishery-specific information indicative of species capture rates to inform estimates of species vulnerability to CCSBT longline fisheries.

Keywords: Seabird, Ecological Risk Assessment, Fisheries, CCSBT, surface longline fisheries, Productivity-Susceptibility Analyses.

[1] Note that all references to the Commission can be read as references to the Extended Commission.

1. Introduction

1.1 Seabird fishery interactions

Seabird interactions with fisheries are a high-profile issue in many jurisdictions and for many Regional Fisheries Management Organisations (RFMOs) (FAO 2010). During fishing with longlines, seabirds may be caught on baited hooks or entangled in fishing lines, resulting in mortality. Three billion longline hooks are set annually around the globe, and it is estimated that 300,000 or more seabirds may be killed annually (Anderson et al. 2011). International agreements assert the need to reduce adverse effects of fishing on non-target catch and seabird populations, and to safeguard populations during migrations. These include the Convention on the Conservation of Migratory Species of Wild Animals (2003), the Fish Stocks Agreement (UNGA 1995), the Code of Conduct for Responsible Fisheries (FAO 1995), the Convention for the Conservation of Antarctic Marine Living Resources (CCAMLR 2007), the Western and Central Pacific Fisheries Commission (WCPFC 2007), the Indian Ocean Tuna Commission (IOTC 2006) and the Agreement for the Conservation of Albatrosses and Petrels (ACAP 2009).

To assist RFMOs in the aim of minimising impacts on non-target species, the Food and Agriculture Organisation of the United Nations has published best-practice guidelines for domestic fisheries and RFMOs (FAO 2008), detailing effective methods and processes for reduction of seabird bycatch as recommended by the FAO International Plan of Action for Reducing Incidental Catch of Seabirds in Longline Fisheries (IPOA) established 10 years earlier (FAO 1999). Following the publication of the IPOA in 1999, several jurisdictions developed in-country instruments, or National Plans of Action – Seabirds (NPOA-Seabirds). For New Zealand the initial NPOA-Seabirds was published in 2004, and has been replaced in 2013 by a revised version (New Zealand 2013). This policy recommends a global risk assessment following the methodology elaborated here (New Zealand 2013:50).

Defining the spatial and temporal aspects of incidental seabird catch is an important aspect of these guidelines. Specialised Ecological Risk Assessment (ERA) methods have potential to assist RFMOs in prioritising actions to species, locations and seasons where impacts may be highest (Small et al. 2011). Defining the extent and significance of incidental seabird catch is an issue for the Convention for the Conservation of Southern Bluefin Tuna (CCSBT) to address, and is the subject of discussions under the Ecologically Related Species Working Group (ERSWG) (CCSBT 2012a). Longline fishing activity reported to the Commission operates globally, with a major concentration of activity in the Indian Ocean with a hotspot south east of South Africa, but also in the temperate Pacific, and southern Atlantic Ocean (Figure 1).

The nature and extent of mitigation in place is an important component of understanding seabird-fishery interactions. It is also an important consideration in the assessment of risk. In CCSBT fisheries the required seabird mitigation measures to reduce incidental capture of seabirds are limited to streamer lines, while research into other forms of effective mitigation is strongly encouraged (CCSBT 2012a). Non-binding mitigation measures, such as use of thawed baits, and offal management are described (CCSBT 2012a). The area in which CCSBT fisheries operate overlaps with areas under the jurisdiction of several other fishery commissions. A further non-binding measure on Members of CCSBT is to comply with the mitigation measures in force under the IOTC, WCPFC and ICCAT agreements when CCSBT fisheries operate in the areas of competency of these commissions (Table 1).

CCSBT also encourages information exchange and fisher education to improve seabird bycatch reduction efforts (CCSBT 2012b). Although data to detail which mitigation is in use in which parts of the CCSBT longline fishery is lacking at present, in future the inclusion of these factors would assist in improving the analyses.

This study focuses on CCSBT surface longline fisheries, and applies Ecological Risk Assessment (ERA) methods reviewed at the CCSBT EERSWG 9 in 2012 (Waugh et al. 2012b):

- To assess which species are most likely to be adversely affected by surface longline fishing mortality from CCSBT fisheries; and,

- To identify which areas and seasons have highest risk of seabird mortality.

In this study we apply the methods reviewed at CCSBT ERSWG9, with a significant change only in the addition of more detailed species distribution data. This was enabled through the provision of remote-tracking data for the breeding and non-breeding periods from the BirdLife International Global Procellariiform Tracking Database, which covered 23 out of the 34 species included in the analysis (see Table 2). Other parameters and modelling choices remained the same as those presented to ERSWG9. For the remaining 11 species improved spatial distribution data for both parts of the annual cycle were unavailable, so the previously utilised maps, relying only on colony proximity and global range maps, were retained.

1.2 Species of conservation concern

Twenty-eight percent of seabird species are threatened with extinction according to the International Union for the Conservation of Nature (IUCN 2011), and there is a potential for seabird-fishery interactions to further threaten at-risk seabird populations. BirdLife International (2006) noted that several species of seabird spend more than 70% of their time in the areas of operation of CCSBT fisheries, as follows: Amsterdam albatross *Diomedea amsterdamensis* (100% of their time), Buller's albatross (97%), Chatham albatross *Thalassarche eremita* (71%), Indian yellow-noted albatross *Thalassarche carteri* (100%), northern royal albatross *Diomedea sanfordi (92%)*, shy albatross *Thalassarche cauta* (73%), southern royal albatross *Diomedea epomophora* (72%), Tristan albatross *Diomedea dabbenena* (69%) and Westland petrel *Procellaria westlandica (*100%).

All of these species are listed by the IUCN as threatened with extinction, including two species listed with the most severe threat ranking possible, "Critically Endangered": the Amsterdam and Tristan albatrosses (IUCN 2011).

Albatrosses are particularly vulnerable to adverse population effects of fishing mortality, partly due to their long-ranging foraging habits which expose them to fishing activity over large areas of ocean, and partly because of their extreme low-productivity life-history traits (Rivalan et al. 2010). For example, some albatross species breed at most once every two years, and take up to one year to raise a chick, and have an age at maturity of over 10 years. Should one adult die during its breeding period, the chick will most likely not survive, and the widowed mate may take several years to find another mate. Due to this low reproductive output, even occasional captures in fisheries can put pressure on seabird populations and contribute, long term, to declines in numbers of birds at breeding colonies

(Weimerskirch et al 2011). Significant declines have been observed in most albatross populations, the most threatened family of birds globally, of which 17 of the 22 species are threatened with extinction (IUCN 2011) and nearly one half of all seabirds have declining trends, with seabird bycatch in fisheries listed as an important influence for many species (Croxall et al. 2012).

1.3 Ecological Risk Assessment (ERA)

To implement the management required to reduce the environmental effects of fishing called for under international agreements, such as the United Nations' Fish Stocks Agreement (UNGA 1995) and the Code of Conduct for Responsible Fisheries (FAO 1995), fishery managers are required to consider which of a suite of non-target species populations may be adversely affected by fishing mortality. To make best use of patchy and at times highly uncertain information, Ecological Risk Assessment (ERA) approaches have been developed (e.g. Hobday et al. 2006, Kirby 2006, Tuck et al. 2011). Productivity-Susceptibility Analysis (PSA) is a semi-quantitative ERA methodology, developed to identify the fishery-associated risks of adverse population effects on non-target species, and to help prioritize management across a broad suite of non-target taxa, such as turtles, sharks, non-target fish, and marine birds or mammals, exposed to different fishing methods (Hobday et al. 2006, Waugh et al. 2011). The need for detailed analysis which considers a suite of population factors along with catch estimation is reinforced by recent research showing that for highly fishery-impacted species, population collapse may occur even where fishery catch rates are closely monitored (Tuck 2011).

In this study, a spatially-explicit PSA methodology was used to estimate the relative impacts of seabird-fisheries interactions and the potential for adverse effects of fisheries mortality on seabird populations (Waugh et al. 2008a, Kirby et al. 2009, Waugh et al. 2012a, b, Richard & Abraham 2013). The 'risk' in this analysis refers to the probability of adverse effects on seabird populations arising from fishing mortality.

In many bycatch-management contexts, data characterising the frequency of capture and species identity of discarded, non-target catch is highly unreliable. Our approach maximises the use of robust available data, and can be applied wherever data is available to characterise the spatial and/or temporal distributions of both seabirds and fishing effort. The species information we use to characterise species productivity includes parameters which can be easily and robustly estimated even in the absence of long-term research programmes, i.e. demographic parameters such as breeding frequency (annual or biennial) and clutch size (one-, two- or multiple-egg clutches depending on the family).

PSAs are a semi-quantitative method of characterising population-level risk on two axes: one which describes the biological productivity of the species, the other its susceptibility to adverse impacts.

On the productivity axis those species with highest fecundity are considered better able to withstand and recover from fisheries removals than slower-breeding species. Susceptibility (i.e. exposure to impact) represents the frequency or probability of fishery-related mortality events for a particular species or population. Susceptibility is characterised by the spatio-temporal overlap between the species distribution and the distribution of fishing effort, multiplied by species 'vulnerability', i.e. a species-specific coefficient representing the relative likelihood that a seabird will be captured or killed in an encounter with fishing effort of a certain method (i.e. 'vulnerability' equates to 'catchability' in fisheries terms). By combining information on both productivity and susceptibility, the species-level risk can be characterised, and the differential effects of removals by a particular fishery on a species population can be assessed.

PSA studies sit in a suite of ERA methods that range from qualitative, such as assessments based on expert knowledge, to fully age-structured population models. Each method has its limitations. For example, expert workshop-based assessments, sometimes termed Level 1 Risk Assessment, such as that undertaken for CCAMLR fisheries (Waugh et al. 2008b, Rowe 2010, Waugh et al. 2011), may be constrained by the inherent biases of participants, and may not provide reproducible results. More complex (Level 3) modelling approaches, such as those undertaken for some species in the Atlantic Ocean, require high quality (and often long-term) datasets to estimate parameters necessary for population modelling (Tuck et al. 2004, Lewison & Crowther 2003, Inchausti et al. 2001), and hence may be applicable to only a small subset of the species potentially affected by fishery interactions. Semi-quantitative (or Level 2) ERA methods, such as those explored here, enable assessment of risk for a broad suite of species or systems including in data-poor settings, incorporating biological or environmental data as available. Representations of uncertainty in the risk calculations can be used to highlight where better quality information is needed. Estimates of risk can be updated and improved as new information becomes available over time. Management responses in relation to ERA findings can inform the development and application of effective mitigation measures, and the prioritisation of fisheries observer programmes or data collection to more accurately characterise fisheries risks.

ERSWG 9 considered an earlier version of this analysis (Waugh et al. 2012b) and concluded that the analyses were useful, and such outputs could help the Extended Commission determine where to implement risk reduction techniques. It was noted that such analyses could identify areas and species of greatest interest for risk reduction, but also highlight where data gaps occurred.

2. Methods

We analysed fishing catch and effort data sourced from the Commission for surface longline fishing effort. Seabird species data were collated from literature review and through compilations of data on species demography and ecology. Seabird range data from multi-research information holdings were accessed to describe the distribution of species globally. Species-specific risk scores were calculated as a function of the season-specific spatial overlap between seabirds and fishing effort, and of species demographic parameters and behavioural susceptibility to capture in longline fisheries, using methods adapted from a similar analysis of seabird interactions in longline fisheries in the Western and Central Pacific Fisheries Commission (Waugh et al. 2012a), CCSBT (Waugh et al 2012b) and in previous analyses (for example, see Kirby et al. 2009 and Filippi et al. 2010). Spatial overlap and risk score estimates were generated for annual and quarterly periods, to examine the effects of seasonally variable fishing effort and species distributions. Spatially resolved risk maps summed across all species in the analysis are summarized as seasonal (quarterly) and total annual risk as indicated below.

2.1 Fishing seasonal and spatial distribution

Fishing catch and effort data for surface longline vessels were extracted from databases held by the Commission Secretariat, and available for download from the internet (CCSBT 2012c). On the advice of the Commission Secretariat, we used total SBT catch (in tonnes) as the most effective proxy for the spatiotemporal intensity of CCSBT fishing (Figure 1). Fishing catch and effort was summarized for each five-degree longitude by five-degree latitude square over the period 2007 to 2010, averaging over four years of data to account for inter-annual variability. These years were selected to most appropriately represent current fishing patterns as this was a period over which relatively consistent regulations and homogenous fishery operations were in place. For quarterly fishing activity plots, data were presented in a negatively-lagged quarters of the year[2] (Q1 = Dec – Feb / austral summer; Q2 = Mar – May / austral autumn, etc), as for species distributions (see below). During the development of the analysis, we also explored alternative proxy representations of fishing intensity, including the reported total number of hooks deployed; in future this or other proxies for fishing intensity could be used as appropriate.

2.2 Seabird species seasonal and spatial distribution

We analysed data for 34 seabird species occurring in tropical or temperate oceanic systems known to interact with CCSBT longline fisheries (Table 2). We included all albatrosses, *Procellaria* petrels, and several petrels and shearwaters. Only some of these species have been documented as catch in CCSBT fisheries. Petrels and shearwaters were included due to the strong propensity for species from these groups to interact with longline fisheries. We used the same set of species analysed in Waugh et al. (2012b), to provide continuity of analysis. Future studies may be able to incorporate additional species.

Including different species with contrasting capture rates provides contrast in the analyses. Our low-probability species include Cape petrel *Daption capense* and Light-mantled albatross *Phoebetria palpebrata.* We chose also to include North Pacific albatrosses, as some Commission datasets include fishing reported in this region. However, for the final data selection, fishing in this region was not included because actual catches of SBT were zero (see above). Nonetheless retention of this group of species serves as a reference, with the expectation that they will rank among the lowest in the analysis, due to lack of spatial overlap with fishing retained in the final dataset.

We used two kinds of distributional data for seabirds. First, we used BirdLife International’s Range Maps as a basis for the species global distributions (BirdLife International 2010). These represent the likely maximum range of a species throughout all seasons. They provide presence/absence information at a global scale by species. For each of the 11 species where range and colony distribution data were used, birds were assigned to either the breeding or the non-breeding distribution on a monthly basis, based on the breeding timetable for each species; monthly distributions were subsequently aggregated into quarterly distributions. We used an exponential decay function to describe the rate at which breeding seabird densities are expected to decline with distance from the colony during the breeding season, due to their central-place foraging pattern, extending up to their maximum foraging range radius (see Waugh et al. 2012a for further details; unpublished

[2] We have lagged the quarters negatively by one month, to better fit the season definitions applied in civil society with summer starting in December in the southern hemisphere, and running till the end of February. Other quarterly conventions apply in other contexts, e.g. financial year quarters, starting 1 January for Q1.

data compilation). The density of birds at a distance r from the colony following an exponential decay is defined with r representing the distance at the colony, thus, if $r >$ *range_max* then *breeder_density*(r) = 0, where *range_max* is the maximum range for a species foraging from its breeding site, and *breeder_density (r)* is the density of breeding birds at a point location.

For $r <=$ range_max:

$$breeder\_density(r) = e^{\frac{\ln(0.01)\times r}{range\_max}} \quad \text{(Eq. 1)}$$

Second, for 24 species for which satellite tracking data was available (see Table 2) we used breeding and non-breeding season distributions based on remote-tracking data from the BirdLife International Global Procellariiform Tracking Database, which consisted of ARGOS satellite telemetry locations, geo-locator system fixes, or Global Positioning System (GPS) logger locations.

We used 50%, 75%, 90% and 95% utility distributions (see BirdLife International 2004 for methods to determine kernel distributions of birds on the basis of these data). BirdLife International provided distributions of birds according to 'breeding season' and 'non-breeding season'. Breeding season maps represent the distribution of adult birds during the breeding season[3] and non-breeding maps represent all birds from that species outside the breeding season. There is likely to be an under-representation of juvenile and pre-breeding birds' distributions in these maps.

Species richness was highest in the Tasman Sea and eastern New Zealand areas when all seasons were considered together (Figure 2).

We established seasonal (quarterly) estimated distribution maps for each species using four quarters of the year that aligned with the breeding time-tables of most seabird species as for fisheries distribution data above. The quarterly hotspots of seabird density varied greatly between seasons, and in some cases demonstrate migratory patterns of birds moving to the northern hemisphere in autumn and spring (Figure 3).

Distribution layers for each species (i.e. combined for breeders and non-breeders, separately for each quarter) for birds from both spatial data groups were normalized such that the sum total of all cells in each layer equals one. In this way each layer represents a global probability distribution per seabird, i.e. the probability that an individual seabird drawn at random from the population will be found in that cell. Multiplying the layer by the appropriate population estimate and dividing by cell area then yields an actual density estimate of birds per km^2.

Estimated total density for all 34 species combined in one annual average plot is shown in Figure 4.

[3] Contrary to Waugh et al. 2012a, this group of birds included all breeding and non-breeding adult birds tracked during the breeding season, as with the non-breeding distribution.

2.3 Productivity-Susceptibility Analyses (PSA)

We used the distributions of fishing and species to calculate seasonal and average annual risk scores based on (a) the *Susceptibility* index and (b) the *Productivity* index.

2.3.1 Susceptibility

The *Susceptibility* index was calculated as the product of fishing distribution and normalised species distributions (i.e. *spatial overlap* on a quarterly basis) multiplied by the *Vulnerability* of the different species to longline fishing gear:

$$susceptibility(sp, se) = \frac{Vulnerability(sp) \times \int_{CCSBT} bird\_density(sp, se) \times effort\_density(se)}{bird\_population(sp)}$$

(Eq.2)

with *sp* and *se* representing respectively the species and the season.

Conceptually the *spatial overlap* is a proxy for the frequency or probability that an individual seabird of a particular species will encounter a fishing event in the fishery in question; *Vulnerability* then represents the likelihood of the seabird being captured or killed in a particular encounter.

2.3.2 Vulnerability

Vulnerability is a function of behavioural and physical characteristics, and differs among species (or species groups), i.e. different species will experience different mortality rates per fishing event for the same seabird density. In the New Zealand EEZ *V* has been estimated empirically for a large number of seabird species, including for albatrosses and petrels included in the CCSBT fisheries analysis, using observed capture rates of seabirds of particular species (or species groups) at different densities of those birds (Richard & Abraham 2013). For each species *Vulnerability* (*V)* relates the density of birds present at a fishing event (*D*) to the likelihood or number of fatal interactions associated with that event (*K*). This provides an instantaneous rate of capture as a function of seabird density. The average number of birds killed *K* per fishing event is then:

$$K = V D$$

(Eq. 3)

Units of *V* are probability of capture per 1000 longline sets, from fisheries observer data, here used as an index of relative likelihood of capture.

The New Zealand Ministry of Primary Industries (formerly Ministry of Fisheries, and Ministry of Agriculture and Forestry) observer data provides a consistent data source that has been used in similar ERA studies to estimate the number of birds caught as a function of spatial overlap with fishing distribution in the New Zealand EEZ (Filippi et al. 2010, Richard et al. 2011, Richard & Abraham 2013). In this CCSBT risk analysis, we use estimates of *V* modelled from observed capture rate data in the updated New Zealand EEZ seabird risk assessment (Richard and Abraham 2013) for vessels similar to those operating in CCSBT fisheries (i.e. longline vessels in excess of 28 m in fishing years 2004-05, 2005-

06 and 2006-07). In that study the species were first grouped together in the following guilds based on similar behavioural and physical characteristics affecting susceptibility to capture in longline fisheries (in descending order of *V*): large albatrosses, small albatrosses, giant petrels, *Procellaria* petrels, large *Pterodroma* petrels, dark shearwaters and southern petrels. Grouping species into guilds was necessary in order to achieve sufficient observed captures in each group to allow statistically robust estimation of capture rates. *V* was then estimated for each species group by fitting generalized linear models to observed capture rates as a function of the estimated density of seabirds in the time and place of each fishing event a (after Filippi et al. 2010, subsequently remodelled by Richard et al 2013). Unlike in previous analyses, by using data from well-observed fisheries and common species to inform estimates of *V* also for poorly observed fisheries and rare species, that study estimated *V* directly for each species or species group, including rarely observed seabirds such as light-mantled albatross and grey-headed albatross. Estimates of *V* were scaled relative to the vulnerability of white-chinned petrel (which was set to 1 as the base case).

It is likely that capture rates derived from the New Zealand EEZ study do not accurately represent the actual likelihood of capture in CCSBT fisheries in *absolute* terms, because variable fisher behaviour or differential use of mitigation between vessels will affect capture rates by altering *V*.

Instead the use in this analysis of the proxy *V* estimated for similar vessels inside the New Zealand EEZ is to approximate the effect of differential behavioural or physiological characteristics affecting susceptibility to longline capture between different seabird species, even while we lack the necessary data to quantify the differential effect of mitigation uptake or fisher behaviour between different vessels. Refinement of the estimates of *V* used in this and future analyses to incorporate the latter consideration would require robust observations of actual capture rates aboard vessels operating in CCSBT fisheries, e.g. by deploying independent observers. This is one of the areas where significant improvement in the data underpinning this and similar analyses can be made, informing improved understanding of fishery associated risks to seabirds by region, flag, and in relation to mitigation deployed.

2.3.3 Productivity

The *Productivity* risk factor is an inverted index of species reproductive potential. A ‘Fecundity Factors Index’ (FFI) was generated which provides a relative measure between species of the fecundity, here based on a normalised ‘Life History Strategy’ (annual breeding, multiple-egg clutches = 1; annual breeding, single-egg clutches = 2; biennial-breeding, single-egg clutches = 3) added to the normalised average age of first breeding, divided by 2, to give a range of values that fell between 0 and 1. This method relies on relatively easily-estimated parameters, and few assumptions, yet provides a robust method of differentiating between species in terms of their ability to recover from increased mortality.

More complex methods relying on increased numbers of assumptions have been shown to produce comparable results (e.g. in Waugh et al. 2012a the Pearson’s *r* comparing FFI and alternate methods was 0.91). Hence here we use FFI here for reasons of parsimony.

2.3.4 PSA risk scores

Season-specific fishery-associated risks to seabird populations were calculated by combining both *Productivity* and *Susceptibility* factors. We defined risk as the product of these two indices, noting that the inverse of the *Productivity* score is used so that the axes move intuitively from lowest risk near the origin to higher risk at higher values. In this way, birds with low productivity, but very little exposure to fisheries interactions could not achieve a high risk score:

$$risk\_score(sp, se) = \frac{susceptibility(sp,se)}{productivity(sp)}$$

(Eq. 4)

Risk maps per species/quarter represent total species-level risk spread in space proportional to the *spatial overlap* (i.e. seabird density map multiplied by *effort_density_map*) in that quarter (Eq 5); annual species risk maps are the average of the four quarters:

$$risk\_map(se, sp) = \frac{vulnerability(sp)}{productivity(sp)} \times \sqrt[4]{\frac{bird\_density\_map(sp, se) \otimes effort\_density\_map(se)}{bird\_population(sp)}}$$

(Eq. 5)

In the estimation of total species-level risk the units for both *Productivity* and *Susceptibility* were normalized between species so that values for each range from 0 to 1 prior to combining both factors to generate the species risk score (see Table 3).

Risk maps by 5 degree square for all species combined were calculated as:

$$Risk\_map\ (se) = \Sigma_{all\ species}\ risk\ map\ (se,sp)$$

(Eq. 6)

By summing un-normalized cell values across multiple species maps, species are weighted in the combined maps proportional to their species risk score; in this way the combined output assigns higher risk to high-risk cells for high-risk species than to high-risk cells for low-risk species. Species combined risk maps of this kind were produced for annual average risk, seasonal risk for each individual quarter, and maximum quarterly risk across all four quarters. We also summarize species risk scores and the parameters by which they are calculated in a series of tables.

3. Results

We discuss the results of what we consider our 'base-case' analysis first in each section. This is the outputs produced by analyses which used variable *V,* and used tonnes of SBT reported as an index of fishing effort. Secondary outputs were tested as a sensitivity, in which we assigned a uniform *V* parameter across all species, but the spatial results were similar and are not reported here.

This analysis builds on the preliminary findings of a study presented to the CCSBT ERSWG 9 (Waugh et al. 2012b), with a methodology agreed by the ERSWG as appropriate for providing advice to the Commission. The key difference is the introduction of more detailed distribution data for the seabird species, a revision requested during CCSBT ERSWG 9. We aimed to maintain the parameters, species included, and outputs similar to those used in 2012, for ease of comparison.

3.1 Biological parameters

Seabird species included in this study and biological parameters contributing to estimation of risk are summarized in Table 2 (see Waugh et al. 2012b for references). *Species group* denotes guild membership within which the *Vulnerability* parameter is assumed to be constant. *Age at maturity* and *life history strategy (LHS*) were combined to yield the *fecundity factors index (FFI)* which in turn affects estimation of the *Productivity* index. The timing of seasonal breeding affects the quarterly species distribution maps. Mean foraging distance is used for defining distributions for species without comprehensive remote-tracking data. Population estimates are used in this analysis to define density of birds. IUCN threat status rankings are provided here for information purposes only.

3.2 Species-specific seasonal and average annual risk maps

Spatially explicit risk maps (per quarter and combined annual total) were produced for each of the 34 species in this analysis. The means by which season-specific species distributions and fishing distributions were used to generate maps of species-level risk are illustrated here with reference to three species case studies.

For Westland petrel, we show the species quarterly distributions (Figures 5, left column 1). The seasonal fishing distribution (as in Figure 1 is shown for each quarter (figure 5, middle column) is shown, along with the species risk scores for each quarter which are derived from these two spatially defined datasets (figure 5, right column). The annual risk map for Westland petrel shows the highest score for each cell where fishing distribution overlaps with species distribution (Figure 6.). This shows a moderate level of risk occurring off western New Zealand, and coincides with the zone most used by breeding Westland petrels during the austral winter, also their breeding season, and a time of year when the CCSBT fisheries is active in this zone.

For the wandering albatross, in figures 7 & 8, we set out the same information as for Westland petrel, although it should be noted that this species has a breeding season that occurs across all quarters of the year, and the species breeds in several localities around the Southern Ocean, hence it has a much higher spatial overlap with CCSBT fisheries than in the Westland petrel case study above, and a correspondingly higher risk score in Table 3. The annual risk map for wandering albatross (Figure 8) shows a high level of risk occurring in the Indian Ocean and moderate levels south of Africa and south-west of Australia.

For the Amsterdam albatross figure 9 sets out the seasonal distribution of the species and the quarterly risk maps. The annual risk map is shown in figure 10. For this species, the risk areas are concentrated in a smaller area than for wandering albatross due to its more limited spatial distribution, and are in areas in proximity to the sole breeding site, Amsterdam Island, where a moderate level of risk is noted, occurring principally in the austral winter.

The annual risk maps for each of the 11 species for which risk was estimated to be High to Moderate (see below) are shown in Appendix 1.

3.3 Species-level risk scores

Species-level *Vulnerability, spatial overlap*, *Susceptibility*, and *Productivity* index values and the corresponding risk scores are summarized in Table 3. *Productivity* is calculated from biological parameters in Table 2; *Susceptibility* is calculated as the product of species *Vulnerability* and *spatial overlap* as in Eq 2, this time using values summed across all cells in the spatial domain rather than calculated on a per-cell basis. Species risk is the product of the species-level *Susceptibility* and inverse *Productivity* indices as shown in Figure 11. In the species-level risk scores both *Susceptibility* and inverse *Productivity* were normalized between species so that values range 0 to 1; the resulting species risk scores range 0 to 0.35.

These analyses indicate that the seabird species most at risk from CCSBT longline fisheries are primarily large albatrosses at temperate and sub-Antarctic latitudes: Antipodean albatross, wandering albatross, white-capped albatross, Gibson's albatross, Indian yellow-nosed albatross (risk score > 0.15), reflecting low biological productivity, high *Vulnerability* to capture by longline fisheries and high *spatial overlap* with recent CCSBT fishing distribution patterns (Table 3 and figure 11).

Species at moderate to high risk (risk score ranges 0.01 – 0.15) include the large albatrosses with lower *spatial overlap* (e.g. Amsterdam albatross, southern royal albatross) as well as smaller albatrosses with highest *spatial overlap* (e.g. Campbell albatross, Buller's albatross, Atlantic yellow-nosed albatross, black-browed albatross) but lower *Vulnerability* relative to large albatrosses, except for Buller's albatross which has very high *Vulnerability* compared to all other species.

Species at low to moderate risk (range 0.0001– 0.01) were large albatrosses (Tristan albatross, northern royal albatross), small albatrosses (sooty albatross, light-mantled albatross, shy albatross, grey-headed albatross), and petrel species (Westland petrel, cape pigeon, grey petrel, northern giant petrel, great-winged petrel).

Lowest risk species (risk < 0.0001) include abundant species for which *Vulnerability* is very low (e.g. sooty shearwater, several *Procellaria* petrels), or for species where *spatial overlap* is near to or at zero (e.g. southern giant petrel, the three north Pacific albatrosses, waved albatross).

The annual risk maps for each of the 11 species for which risk was estimated to be High to Moderate are shown in Appendix 1.

3.4 Species-combined seasonal and total annual risk maps

Combined seabird risk across all 34 species is represented by summing the untransformed species specific risk maps by season, as shown in Figure 12. The effect of using the untransformed species level risk layers is that species are weighted in the combined maps proportional to their species *risk* score in Table 3; in this way the combined output assigns higher risk to highest-risk cells for high-risk species than it does to highest-risk cells for low-risk species. In Figure12 lower seasonal risk in spring and summer (Q4 and Q1) reflects both the lower absolute level of fishing in these seasons and also the dispersed spatial distributions of many seabird species outside of their breeding season. Lower risk areas are denoted dark blue on the graphic.

Conversely, highest risk in autumn and winter (Q2 and Q3), denoted by warm colours, green, or pink, reflects the increased concentrations of at-risk seabirds around breeding sites, in locations that coincide with seasonally high CCSBT fishing, primarily in the Tasman Sea and around New Zealand. Other locations of elevated risk include the waters off southern Africa in autumn and winter, and southeast of Australia in winter and spring. These same areas are reflected also in Figure 13 which depicts maximum annual risk across all 34 species for each spatial cell.

3.5 Risk by flag

The flag for which fishing had the most risk of seabird mortality associated with it was Japan, followed by the Fishing Entity of Taiwan, and New Zealand (Figure 14). These three flags contributed 52%, 20% and 19% of the risk, or 91% in total of the risk in the analyses. As noted above, these risk scores largely reflect the location and quantum of fishing effort of these flag states and may alter if better estimates of vulnerability were available, since it is likely that capture rates derived from the New Zealand EEZ study do not accurately represent the actual likelihood of capture in CCSBT fisheries in *absolute* terms.

4. Discussion

During this study we applied methods developed for assessing the risk to populations of seabirds of incidental mortality to longline fishing activity conducted under the management of the Commission. We adapted the methods applied elsewhere in Pacific regions (Waugh et al. 2008, 2012a) using alternate datasets for species and fishing effort, and applying a different risk estimation approach using a quantitative *spatial overlap* metric yielding semi-quantitative estimates of species risk, in which risk scores can be expected to indicate the relative magnitude of fisheries-associated risk to different seabird populations, rather than merely ranking species in order of decreasing risk. The same method can also be used to track changing risk to particular species or groups of species over time in a given fishery or area, or alternately for global populations affected by fisheries in different areas, to compare the relative magnitude of risks arising from fisheries in different areas or under different jurisdictions. However in the absence of fishery-specific data indicative of species capture rates to inform estimates of species *Vulnerability* specific to CCSBT fisheries, this analysis relies on proxy estimates of V from a quantitative risk assessment in the New Zealand EEZ (as in Filippi et al. 2010, Richard and Abraham 2013). These estimates can be regarded as accurate in a relative sense to represent differences in *V* between species, but not in an absolute sense to estimate actual numbers of captures because *V* is affected also by vessel characteristics, gear configuration, fisher behaviour, and mitigation, the effects of which remain un-quantified in CCSBT fisheries in the absence of independent observers.

The results reported here build on preliminary analyses done in 2012, updated with more detailed spatial distribution information about species.

4.1 Species and areas of greatest risk of seabird-fishery interactions

The study suggests that the seabird species at highest risk from CCSBT longline fisheries include several species of temperate-distributed albatrosses, that risk is highest in the austral autumn and winter, and that geographical areas of highest risk include the large sections of the Tasman Sea and the areas south and east of New Zealand. All of these areas have previously been identified as potential problem areas for seabird bycatch (e.g. Abraham et al. 2010, Abraham and Thompson. 2011, Watkins et al. 2008, Tuck et al 2003, Glass et al. 2000). This analysis suggests that in addition to being areas of high density of individual birds, and corresponding increased numbers of seabird-fishery interactions, fishing activity in these areas also poses risks to rare species' populations. Vulnerable populations frequenting these areas are likely to be adversely affected by fisheries-associated mortality due to a combination of intense fishing effort and concentrations of individuals of these species at the same times.

Areas of importance for reducing risk to individual species are found in other areas, e.g. in the Southern Indian Ocean for wandering and Amsterdam albatrosses. See the Annexes for additional species-by-species plots.

The study stops short of exploring to what extent the incidental mortality of particular seabird species is likely to deplete their populations, instead assigning risk in a relative sense between species and between seasons/locations. The large albatrosses are shown to be at higher risk of adverse population effects compared with small albatrosses and petrels, due to the combination of their low-productivity life-histories, high spatial and temporal overlap with fishing activity, and high likelihood of capture when and where they do co-occur with surface longline fishing. Such analysis is available for a subset of the species and areas covered here as described in Richard and Abraham (2013).

The three flag states of Japan, Fishing Entity of Taiwan and New Zealand contributed to over 90% of the potential risk to seabird populations as a result of fishing mortality in CCSBT longline fishing. Risk is not evenly spread between the flag states, due to the distribution of the fishing activity, and also the nature of the species that each fishing group is likely to encounter, e.g. more or less vulnerable, or productive species, in different areas. Fishing activity occurring in the Tasman Sea and east of New Zealand is likely to carry a relatively high risk, due to the high density, and the high vulnerability of some of the species which frequent these areas.

Other, more localised risk situations can occur, such as for highly vulnerable species which have restricted ranges (see Appendix 1 for high-to-moderate risk species risk maps).

4.2 Amsterdam albatross case study

We examine the results for the high risk, and rarest species in the study, the Amsterdam albatross.

The Amsterdam albatross, the species with the highest identified risk score in this analysis (0.066), has a breeding population of 29 pairs occurring at Amsterdam Island; the estimated annual distribution and associated risk map is shown in Figure 9. The main foraging areas for both adult and juvenile Amsterdam Albatross coincide with areas of high fishing activity density under the IOTC as well as CCSBT fisheries, as identified in the current analysis.

Understanding risks to this species by area and season is vital, due to the current critical status of the species' population. Weimerskirch et al. (2011) and Rivalan et al. (2010) note that as few as 6 individuals removed from the population due to bycatch or other anthropogenic effects would be sufficient to result in extinction of the species within 10 years. Although this species ranked 9th in the analysis for most at-risk species, its population status make it highly vulnerable to added mortality.

4.3 Study limitations and next steps

The study has benefited from updated species distribution information, compared with previous analyses, and represents the current state of knowledge of species ranges for those species studied with remote sensing during breeding and non-breeding seasons.

A key strength of the risk assessment method is that it is designed to be easily updated as new data becomes available, and risk scores adjust accordingly.

In addition it may be beneficial to explore datasets indicative of capture rates from the fisheries of several members of the Commission, to better estimate the *Vulnerability* estimates by fishery and region, and begin to understand the effects of different gear configurations or mitigation options on *V*. In future, it may also be useful to examine the use of alternate metrics of fishing effort distribution (total SBT catch was used here).

A high priority remains to conduct a global analysis of ecological risk assessment for seabirds across all longline fisheries.

This analysis did not attempt to characterise uncertainty in the risk estimates; rather we have used what we consider 'best estimates' for each parameter used to estimate risk, rather than plausible ranges or prior distributions.

More sophisticated and data-hungry approaches (e.g. Tuck et al. 2004, Inchausti et al. 2001, Richard et al. 2011, Richard & Abraham 2013) are required to examine the extent to which uncertainty in species and fishery distributions and in other input parameters combine to generate uncertainty in the output estimate of species risk. Our previous experience in the field indicates that the two variables for which uncertainty is likely to be high are the species distribution layers (a factor generally poorly explored in analyses of risk, and for which it is difficult to characterise uncertainty using quantitative methods) behavioural or physiological factors, along with the effects of different vessel or gear configurations and of mitigation on the *Vulnerability* parameter, the improved estimation of which would benefit from both improved species distribution and fishery-specific capture rate information. These latter data require independent fisheries observer coverage, deployed at appropriate times and in appropriate areas, to improve our understanding of the ways that seabird species interact with different fisheries in different areas.

5. Conclusions

The study used ERA methodology to examine which species, areas, and seasons hold most risk of adverse effects of population change for seabird species captured in CCSBT longline fisheries. At a species level, albatross species in temperate and sub-Antarctic waters have highest likelihood of adverse impacts, particularly Antipodean albatross, wandering albatross, white-capped albatross, Gibson's albatross, Indian yellow-nosed albatross, southern royal albatross, Buller's albatross, Campbell albatross and Amsterdam albatross. The case for measures to mitigate capture of Amsterdam Albatross is particularly strong, given the highly fragile nature of its population and strong exposure to fishing effort in the southern Indian Ocean. Key areas for highest likelihood of adverse impacts are in the Tasman Sea and to the east and south of New Zealand, when all species are considered together. Autumn, winter and spring were the seasons when highest likelihood of adverse population impacts occurred, in these areas, as well as in the southern Indian Ocean.

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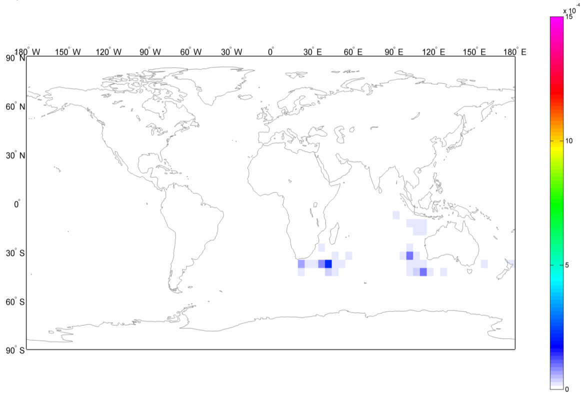
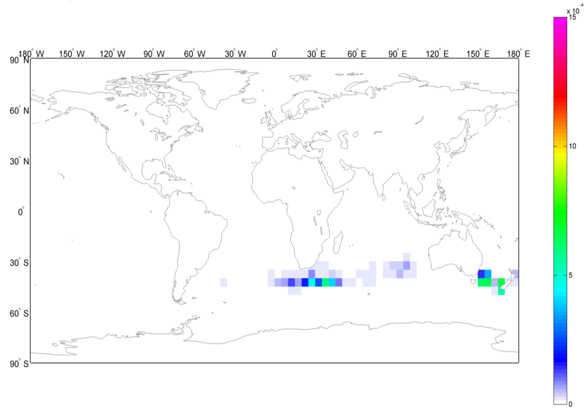
Figure 1. Spatial distribution and intensity of fishing effort (2007-2010) in CCSBT surface longline fisheries by 5-degree cell. The index of fishing effort intensity is average annual total reported catch of SBT (tonnes).

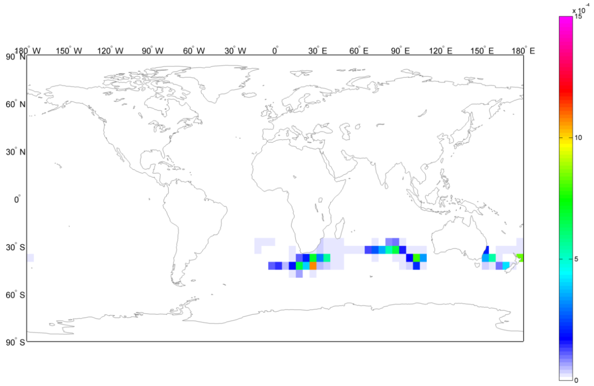
a) Q1 – austral summer



b) Q2- austral autumn



Q3 - austral winter



d) Q4 – austral spring

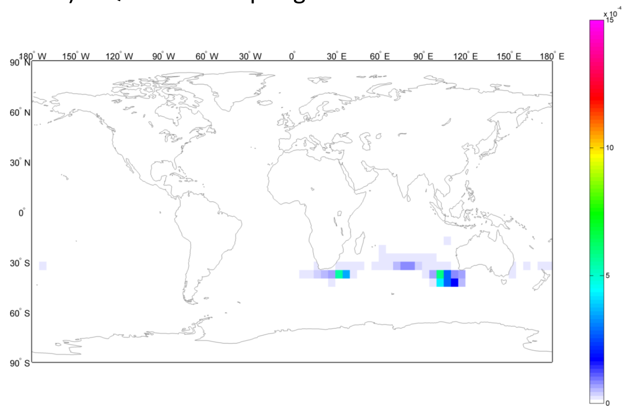


Figure 2. Plot of seabird diversity (number of species per 5×5 degree area) for the 34 species of albatross and petrel included in the analysis. These distributions were generated by combining BirdLife International Species Range Maps (BirdLife International 2010) with colony locations and other literature-based information indicative of foraging distances, and remote tracking data from the BirdLife International Global Procellariiform Tracking Database (BirdLife International 2004).

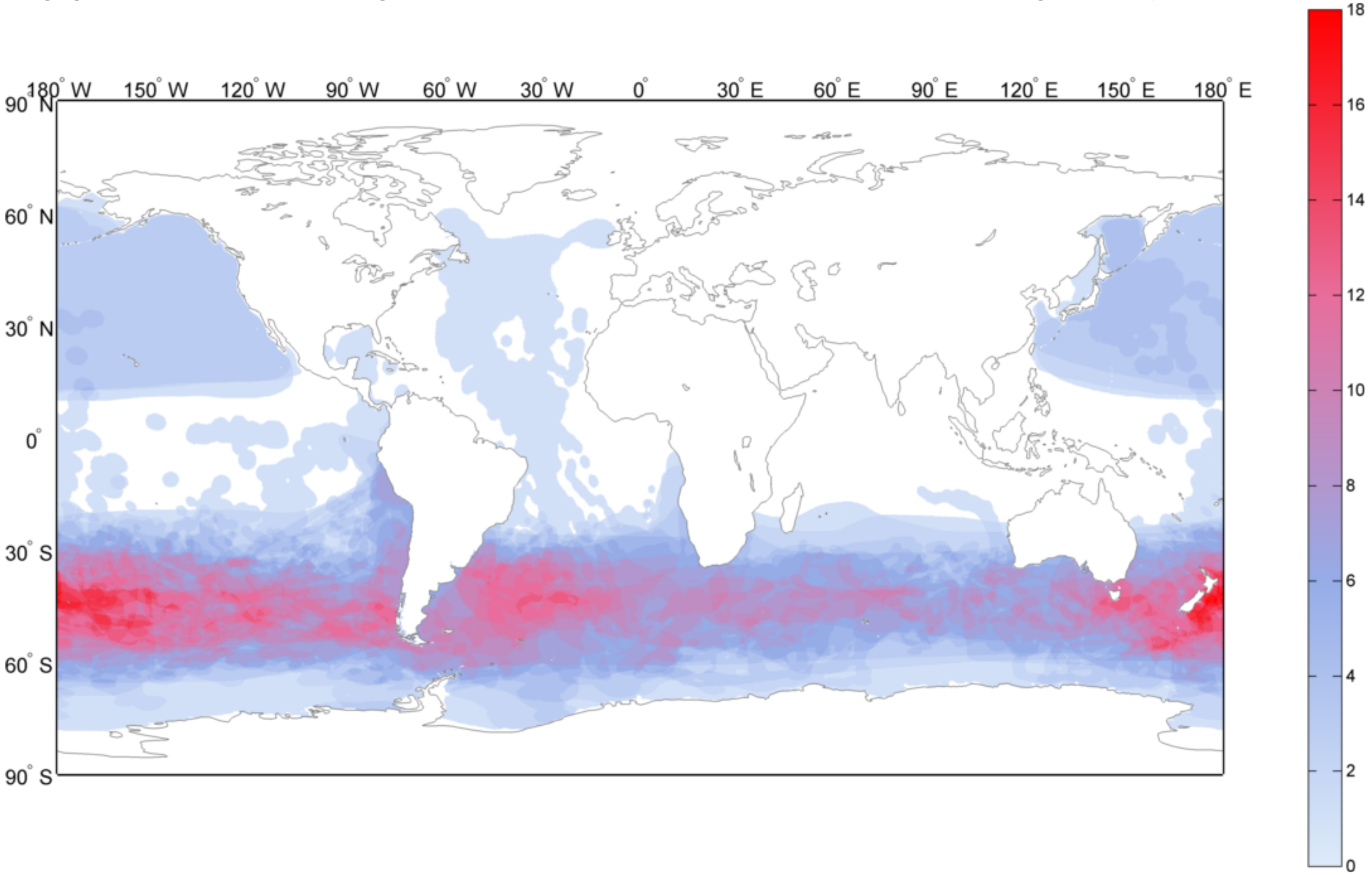
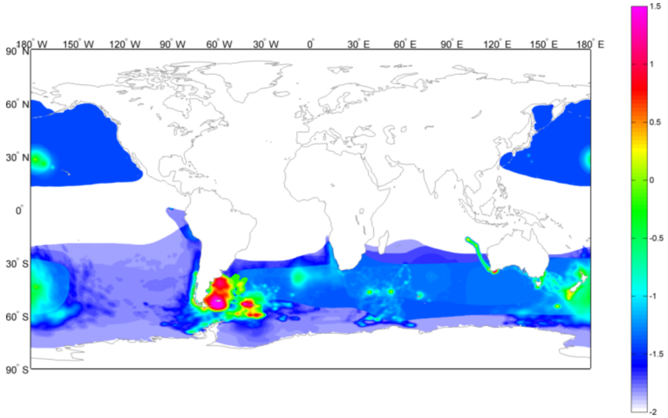
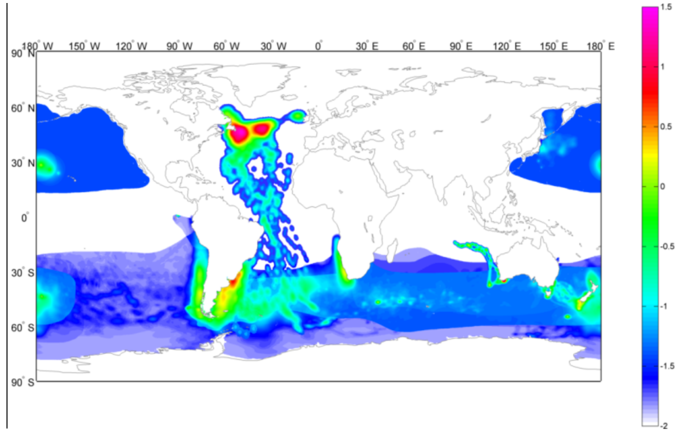


Figure 3. Estimated density of seabirds [log10(birds/km2)] combined for all 34 species of albatross and petrel included in the analysis per 5×5 degree cell during four seasonal quarters: (a) Q1- Austral summer; b) Q2 - Austral autumn; c) Q3 – Austral winter, and d) Q4 - Austral spring. Season-specific densities are estimated for each species by proportionally assigning birds to either breeding or non-breeding season distributions on a monthly basis and subsequently aggregating into quarters. Breeding birds are constrained within their maximum foraging distance from known colony locations. Transformed values are displayed here to aid visual interpretation, but not used in calculations.

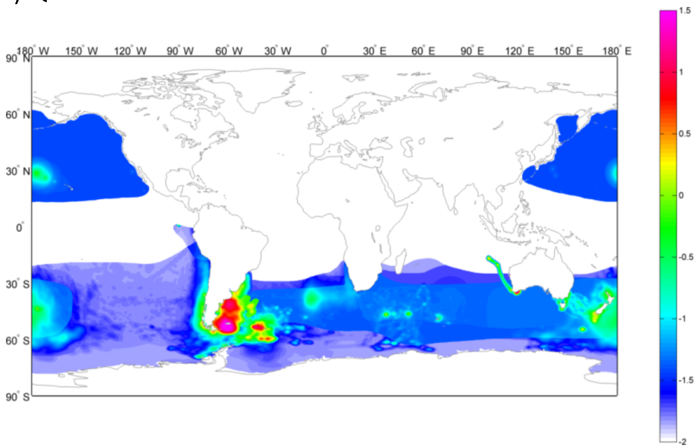
a) Q1- Austral summer



b) Q2 - Austral autumn



c) Q3 – Austral winter



d) Q4 - Austral spring

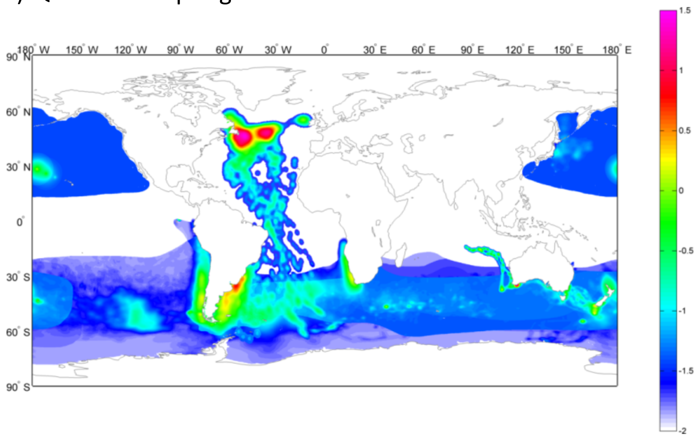


Figure 4. Estimated density of seabirds [log10(birds/km2)] of all 34 species combined per 5×5 degree cell averaged across all four seasonal quarters. Note that this figure and Figure 3 depict total bird densities, such that abundant species dominate the graphic, whereas rare species are under-emphasised. Log_{10} transformed data are displayed here to aid visual interpretation, but are not used in calculations.

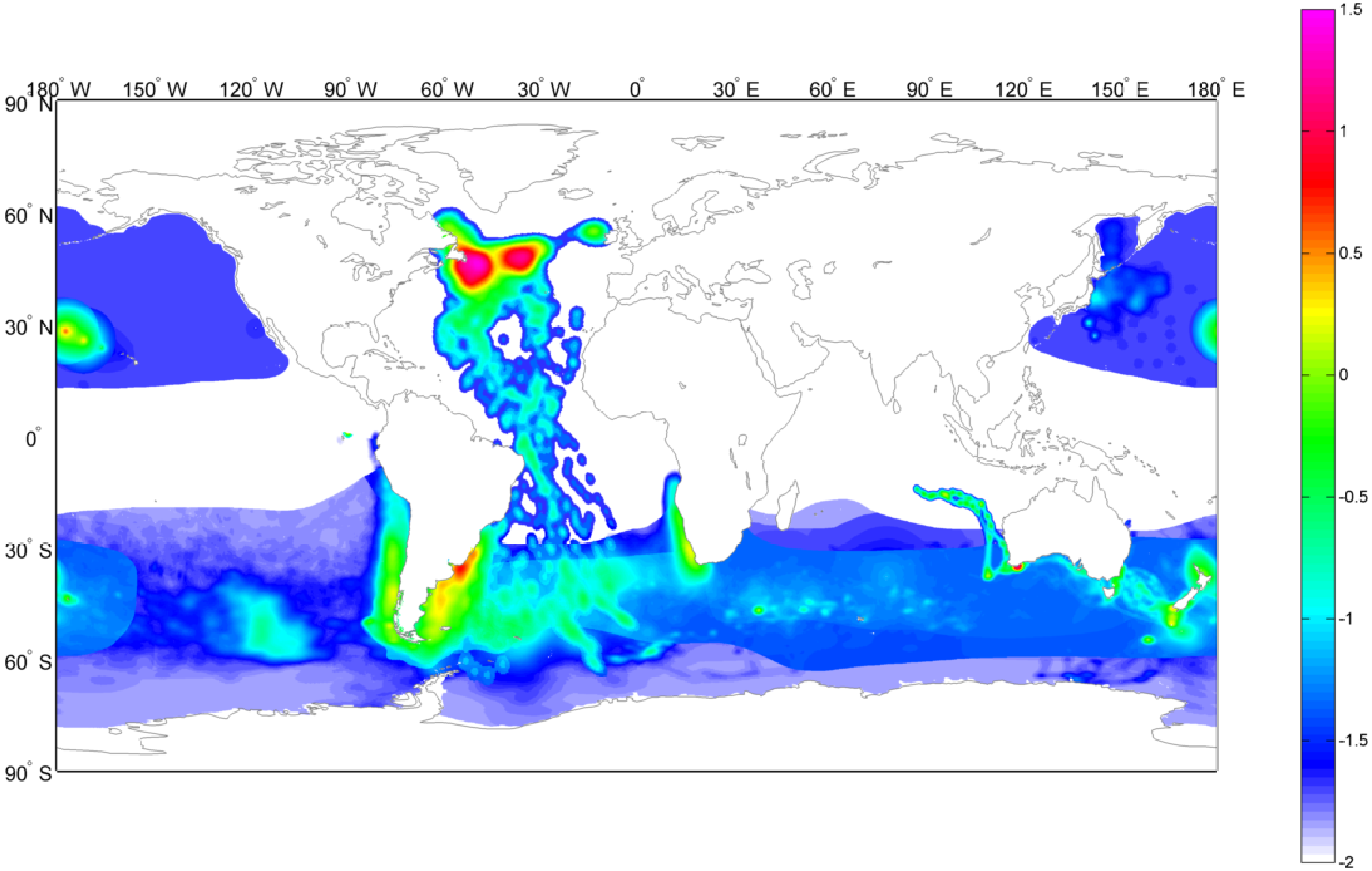
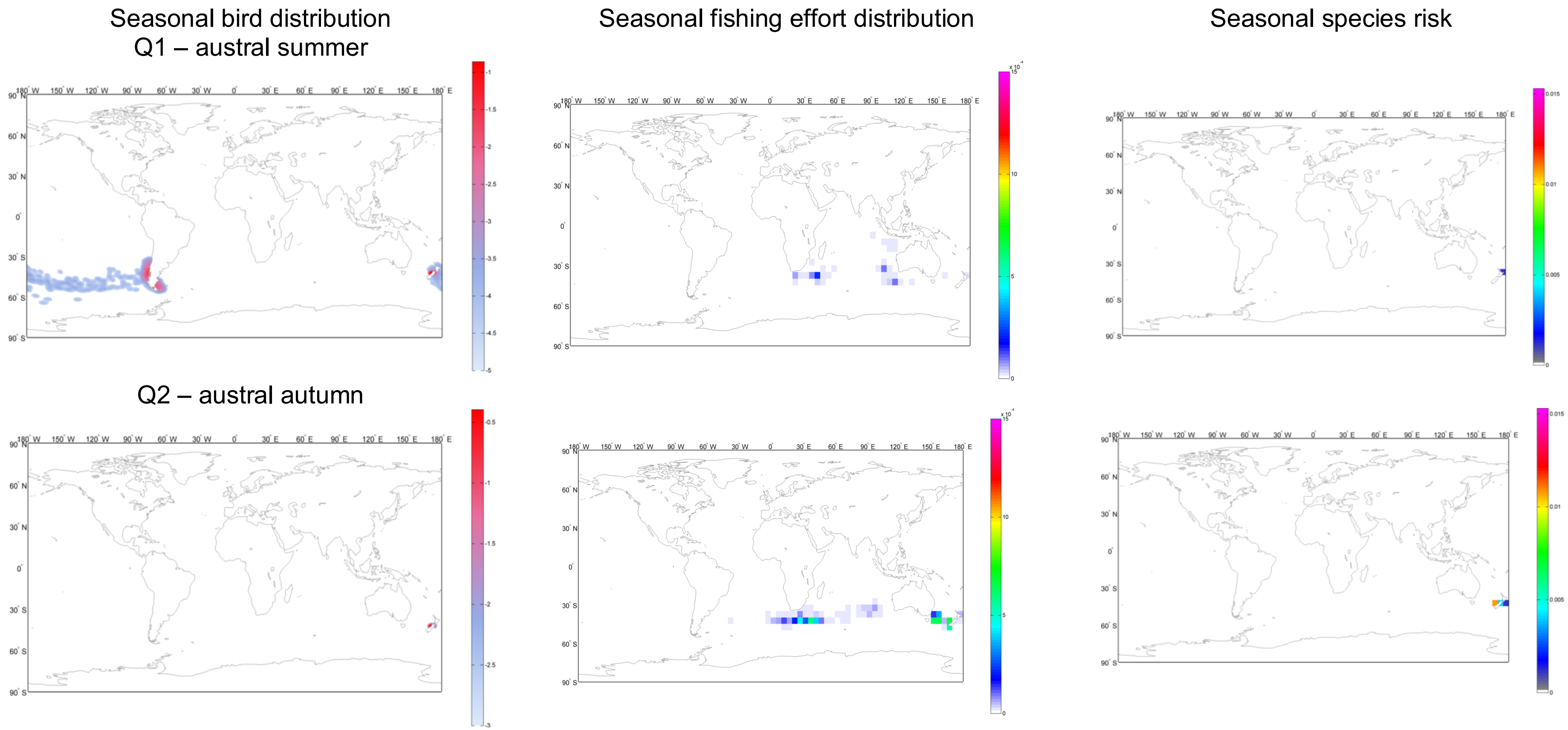
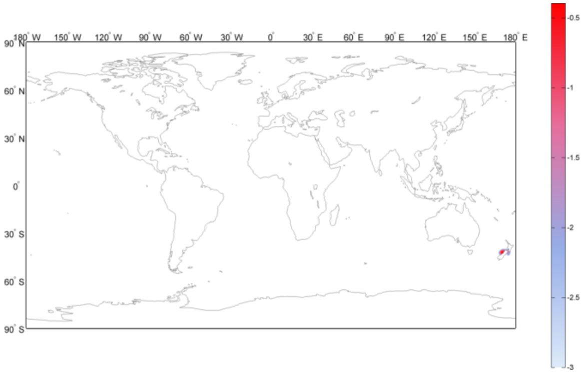
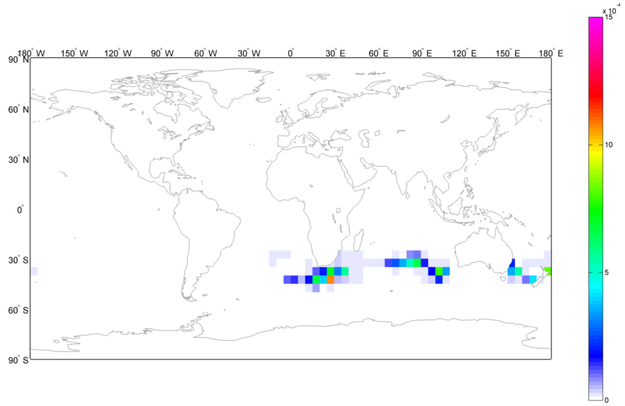


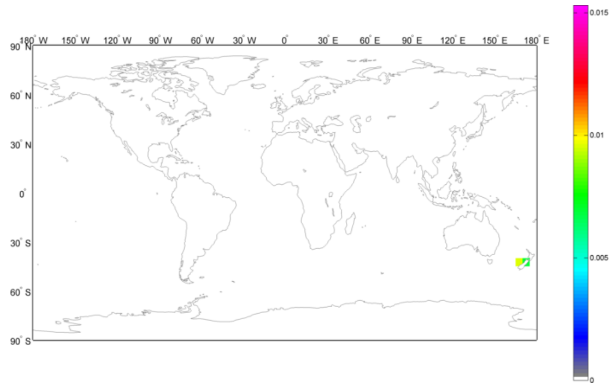
Figure 5. Seasonal spatial bird distribution, effort distribution and seasonal risk map for Westland petrel, to illustrate the means by which spatial input data layers combine to yield the species risk map. Note that this species breeds in the austral autumn and winter in the New Zealand region. The first column shows the bird spatial distribution derived from basic range maps and satellite tracking information for each quarter, the second column is the seasonal distribution of fishing effort using catch of SBT as the index of effort; the third column shows the spatial distribution of risk for Westland petrel.



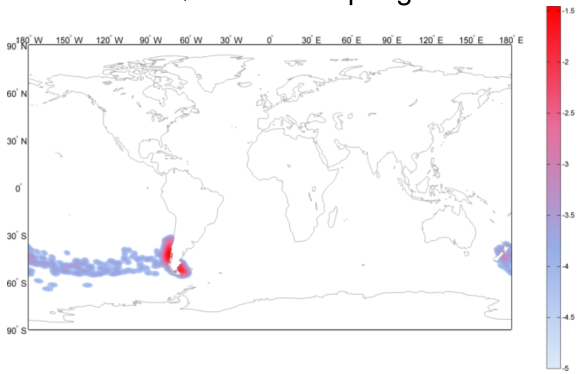
Q3 – austral winter

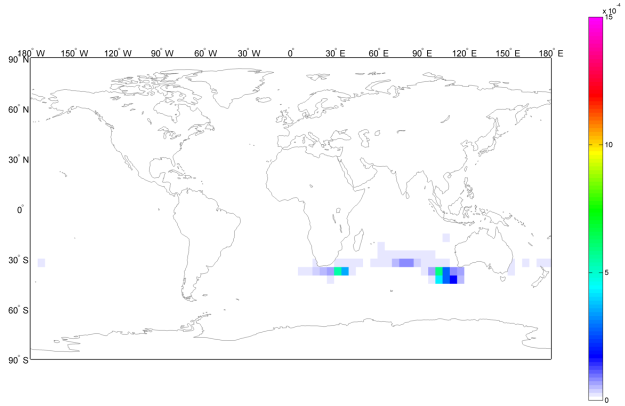






Q4 – austral spring





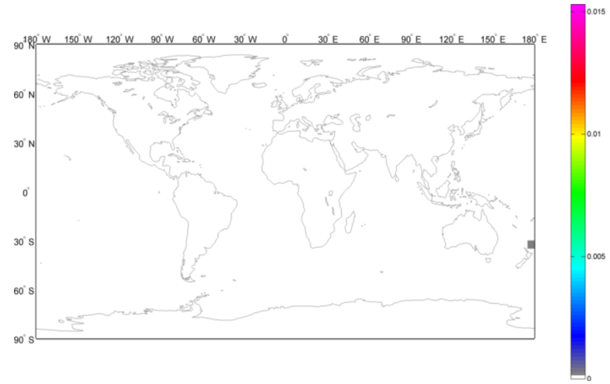


Figure 6. The maximum risk scores for Westland petrel, combining the information for figure 5 above, with the risk score for each cell where fishing effort overlaps with Westland petrel distribution showing the maximum risk score during the year.

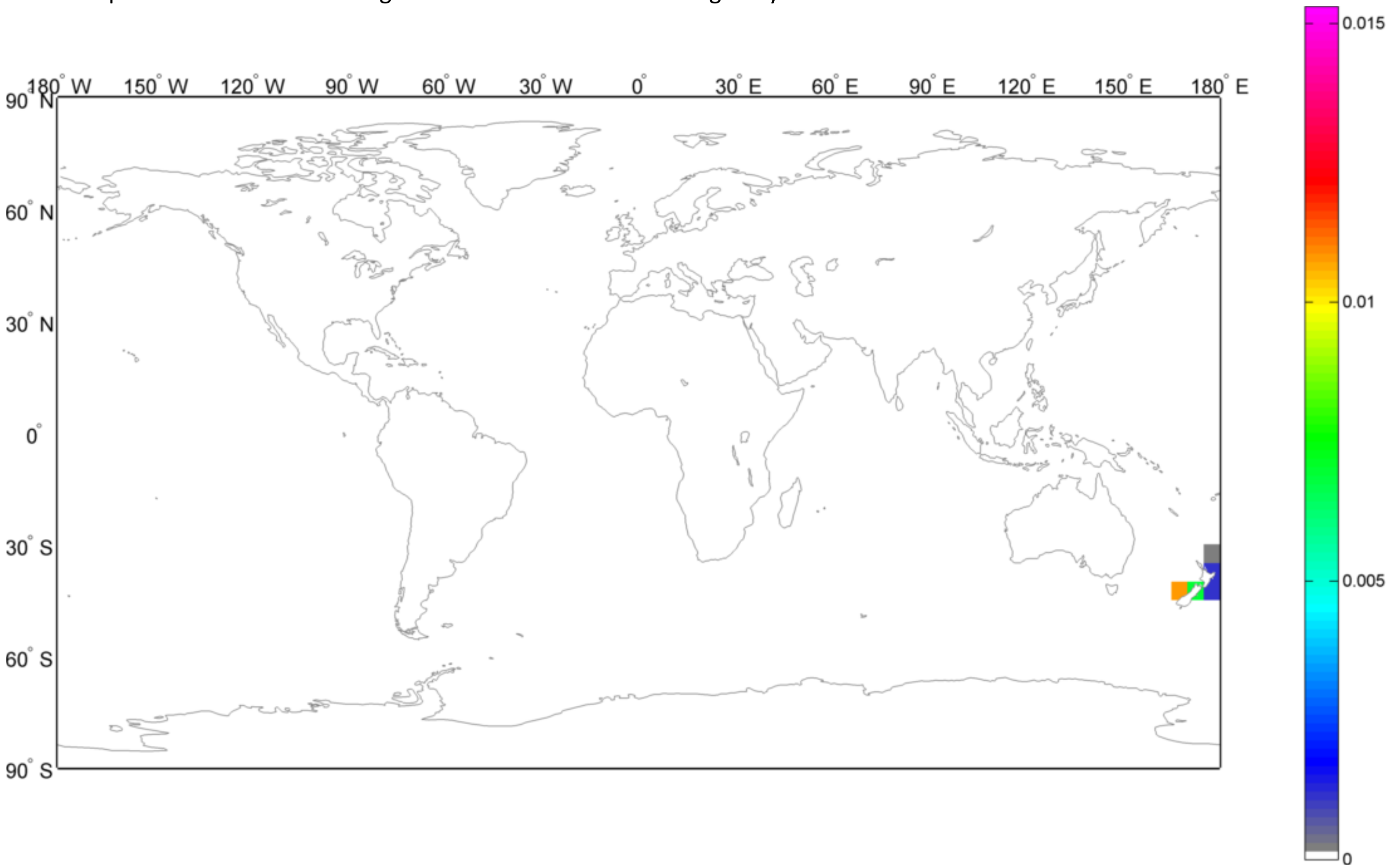
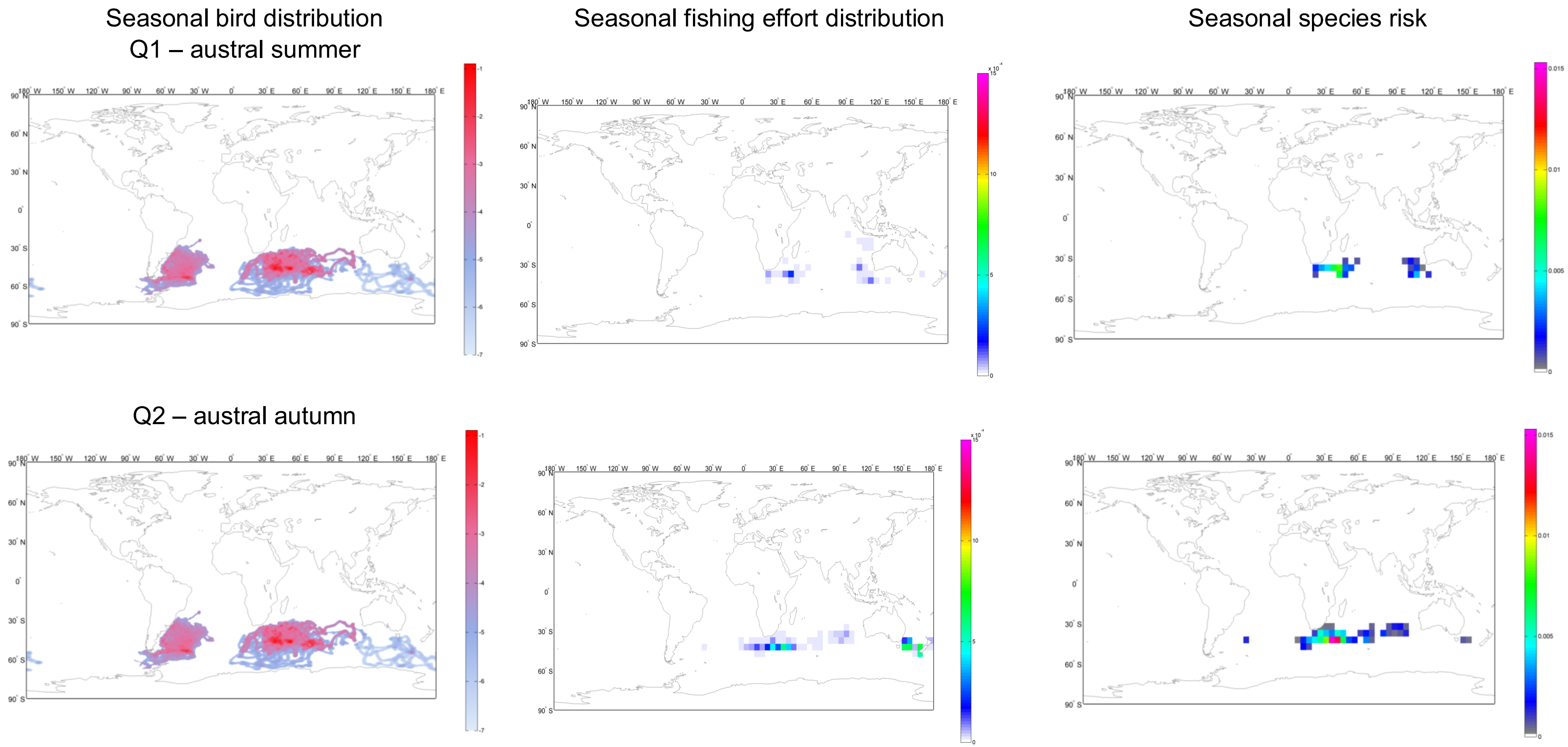
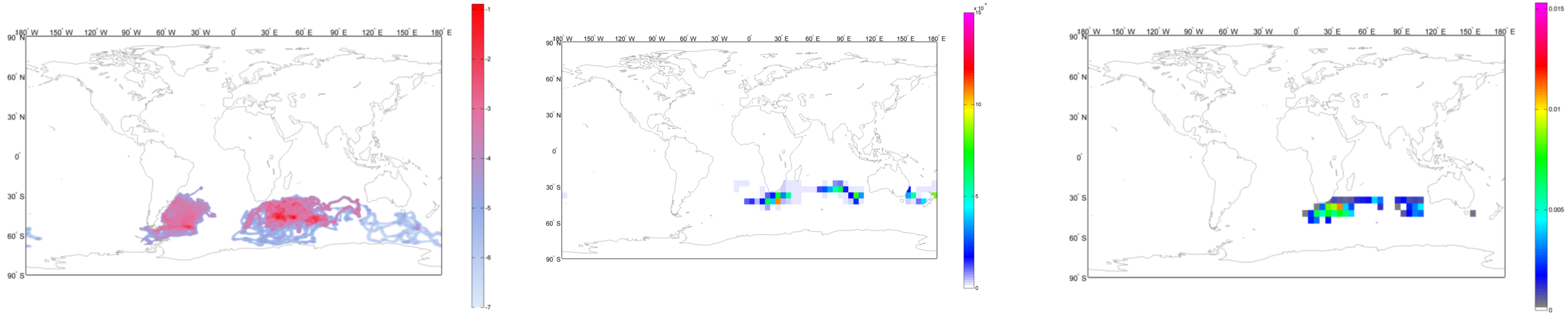


Figure 7. Seasonal spatial bird distribution, effort distribution and seasonal risk map for Wandering albatross, to illustrate the means by which spatial input data layers combine to yield the species risk map. Note that this species has a breeding season that extends throughout the year. The first column contains the bird spatial distribution derived from basic range maps and satellite tracking information for each quarter, the second column is the seasonal distribution of fishing effort using catch of SBT as the index of effort; the third column contains the risk scores for Wandering albatross only.



Q3 – austral winter



Q4 – austral spring

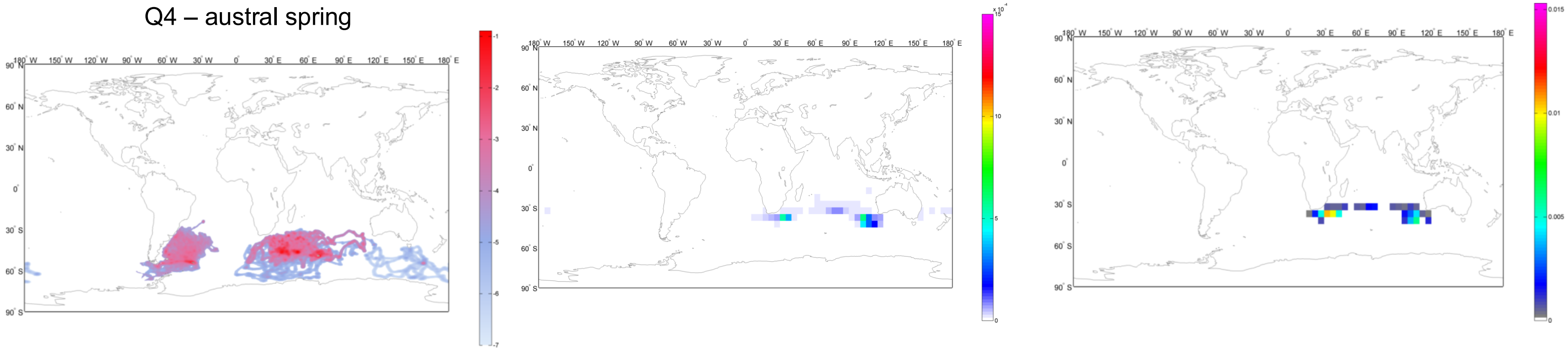


Figure 8.The maximum risk scores for wandering albatross, combining the information for figure 7 above, with the risk score for each cell where fishing effort overlaps with wandering albatross distribution showing the maximum risk score during the year.

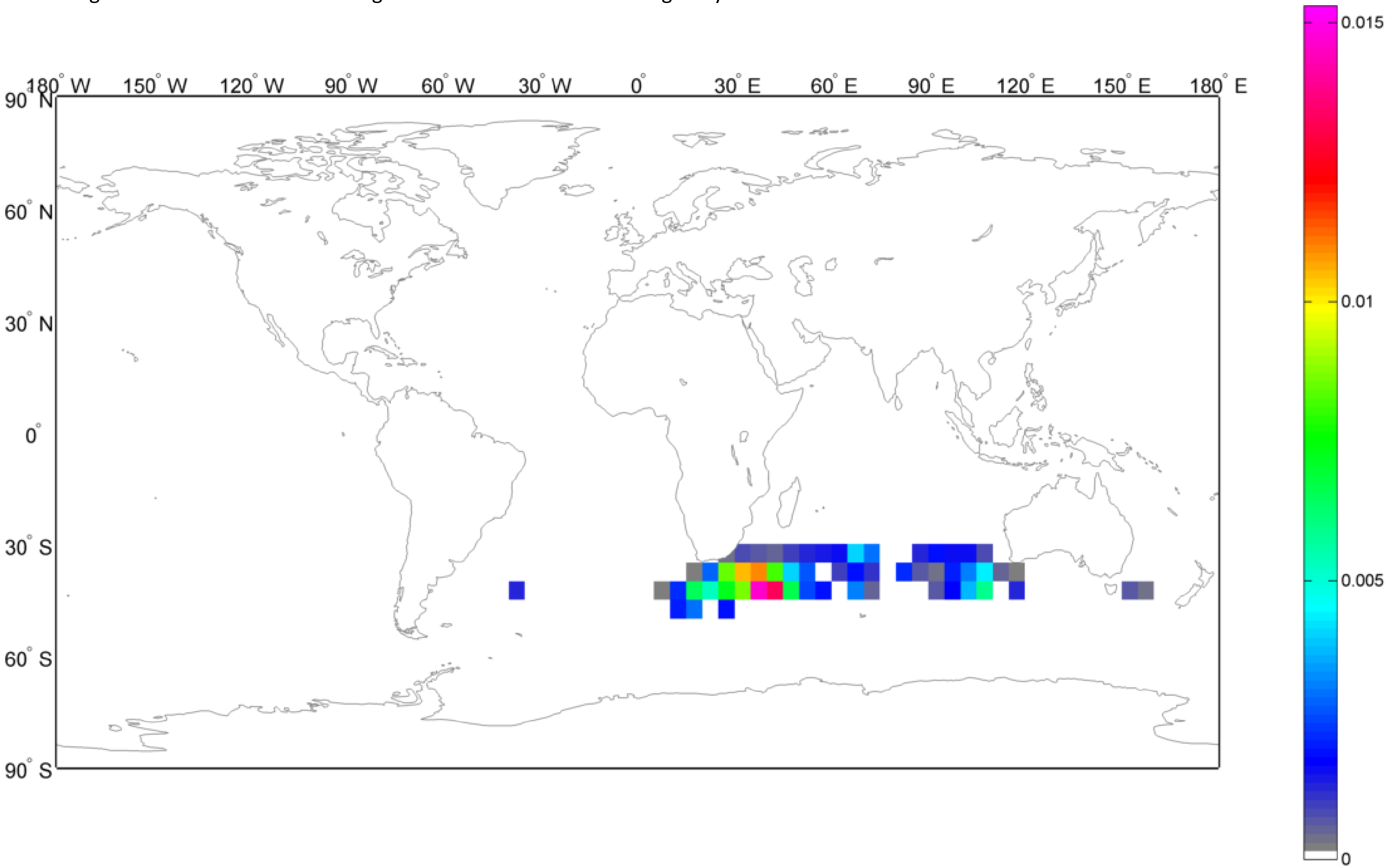
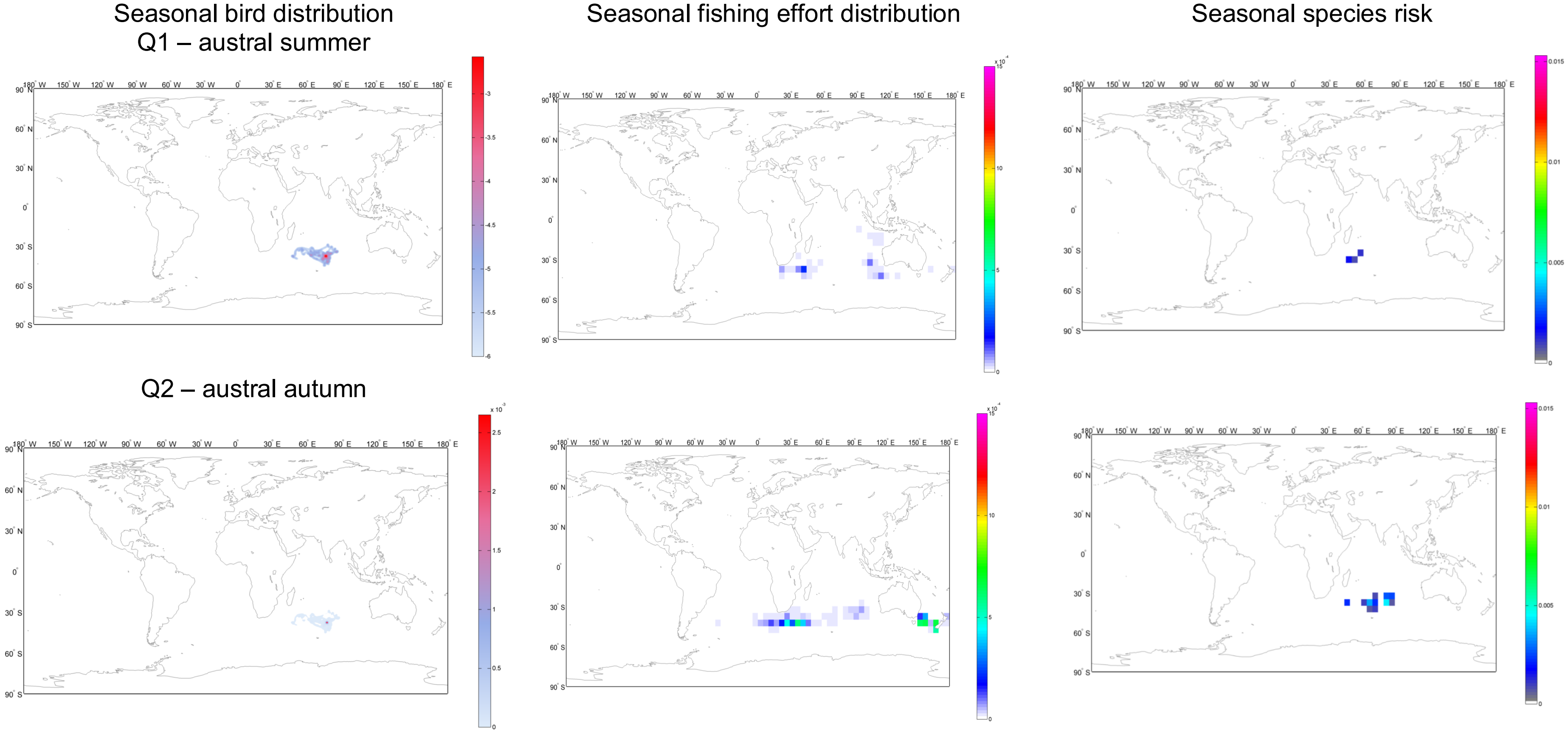
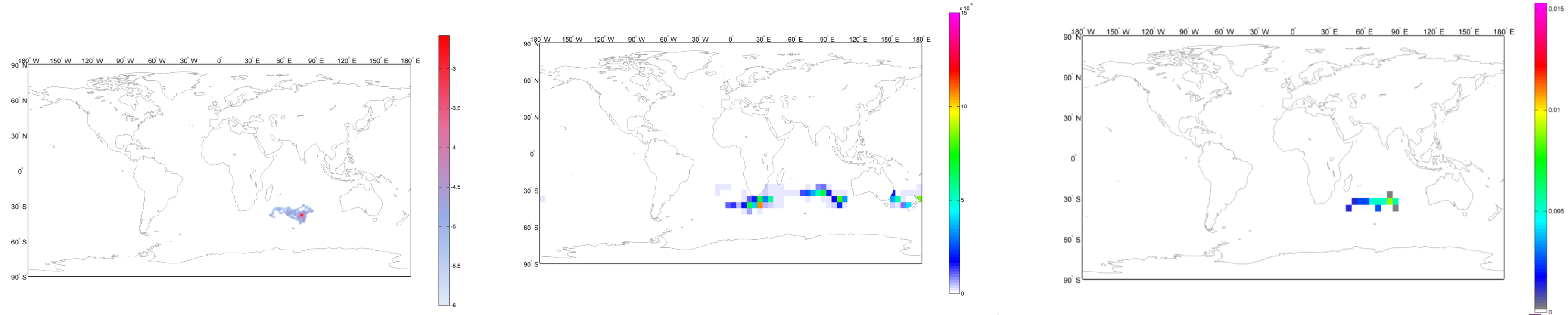


Figure 9. Seasonal spatial bird distribution, effort distribution and seasonal risk map for Amsterdam albatross, to illustrate the means by which spatial input data layers combine to yield the species risk map. Note that this species has a breeding season that extends throughout the year. The first column contains the bird spatial distribution derived from basic range maps and satellite tracking information for each quarter, the second column is the seasonal distribution of fishing effort using catch of SBT as the index of effort; the third column contains the risk scores for Amsterdam albatross only.



Q3 – austral winter



Q4 – austral spring

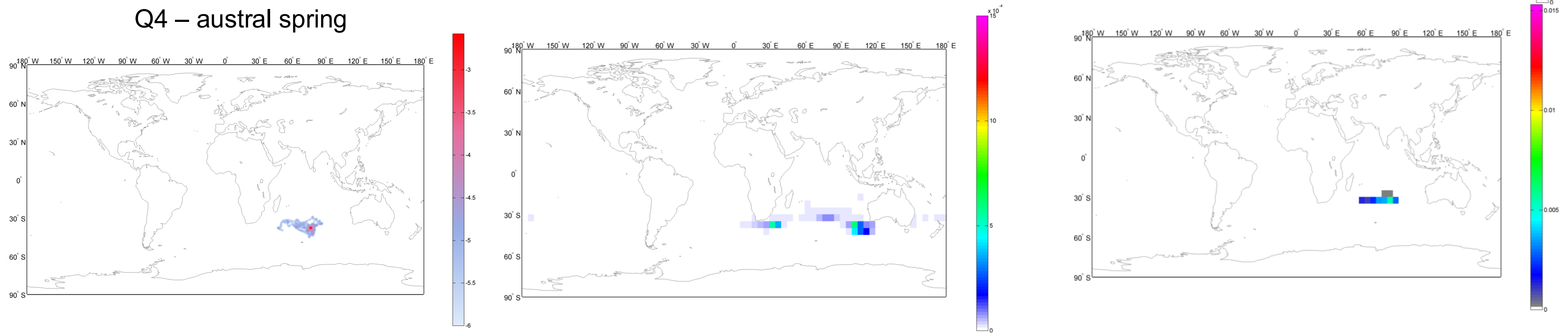


Figure 10. The maximum annual risk map for Amsterdam albatross. The maximum risk scores for Amsterdam albatross, combining the information for figures 9 and 10 above, with the risk score for each cell where fishing effort overlaps with Amsterdam albatross distribution showing the maximum risk score during the year.

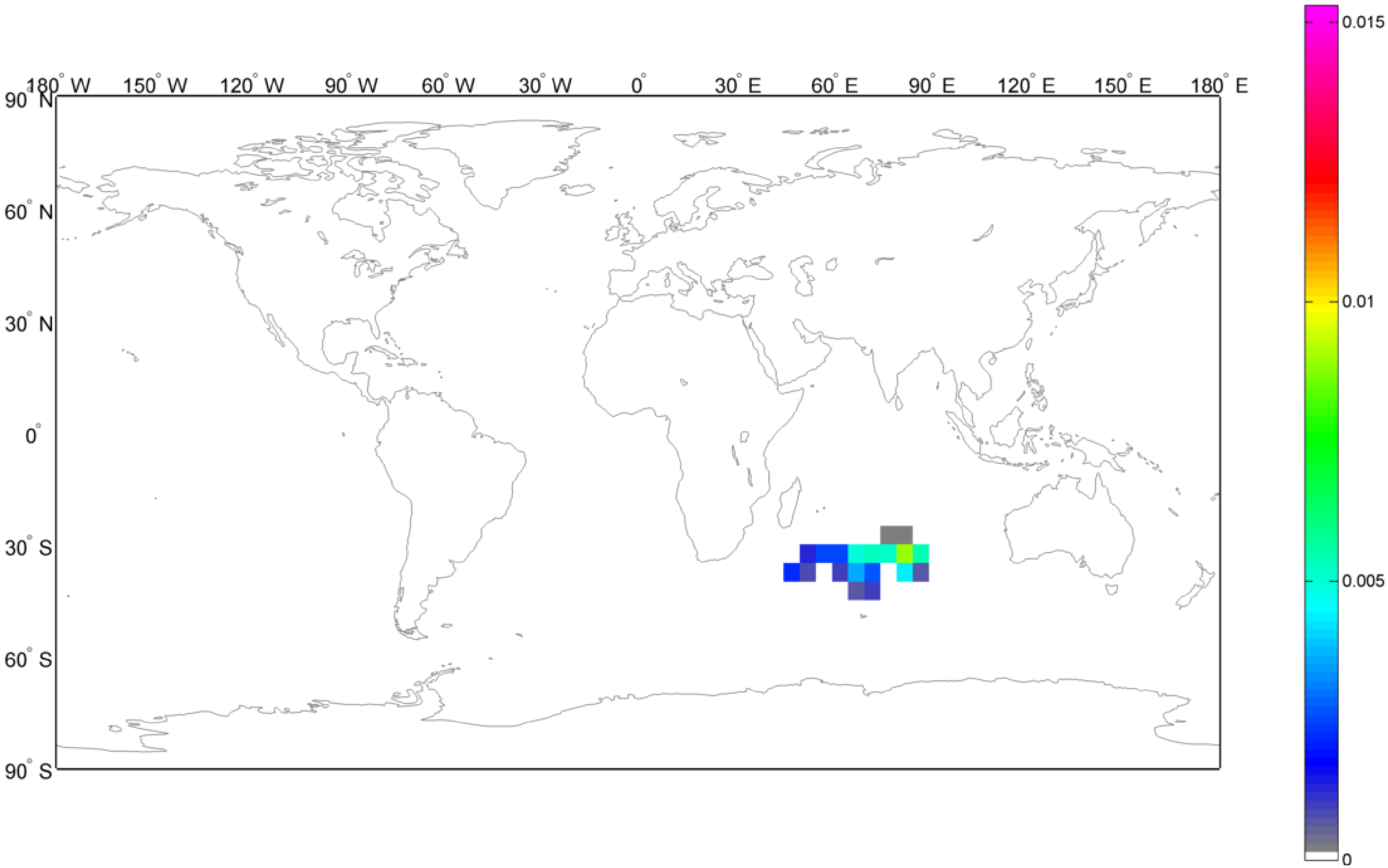


Figure 11. PSA plot showing total annual fisheries-associated risk for each seabird species as a product of their *Susceptibility* index and their inverse *Productivity* index. *Susceptibility* is estimated as a function of spatial overlap and species *Vulnerability (V); Productivity is* a function of the Fecundity Factor Index and age at reproduction as described in the text. *Risk* is proportional to the area of the rectangle formed by plotting the species on these normalized axes (see Table 3 for species codes and index values). Bird species are colour-coded by guild as follows: Large albatrosses = red; small albatrosses = green; giant petrels = magenta; large shearwaters = black; miscellaneous small petrels = blue.

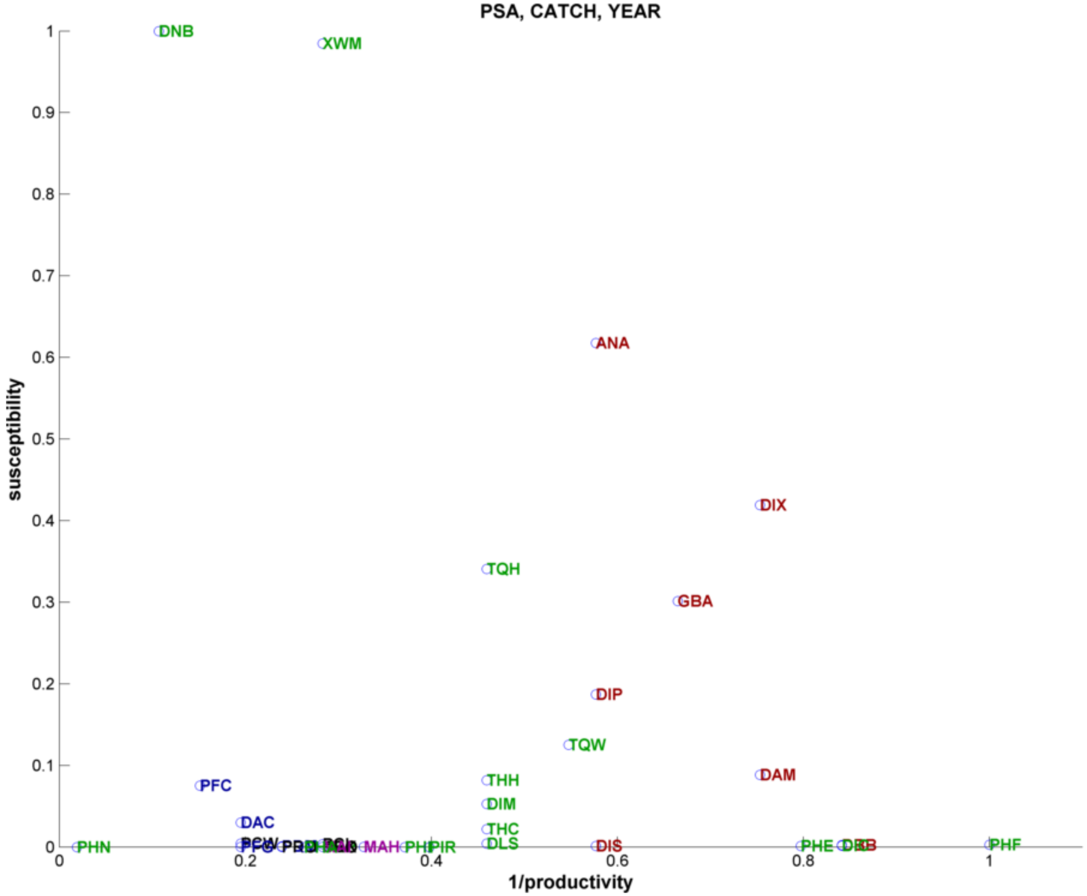
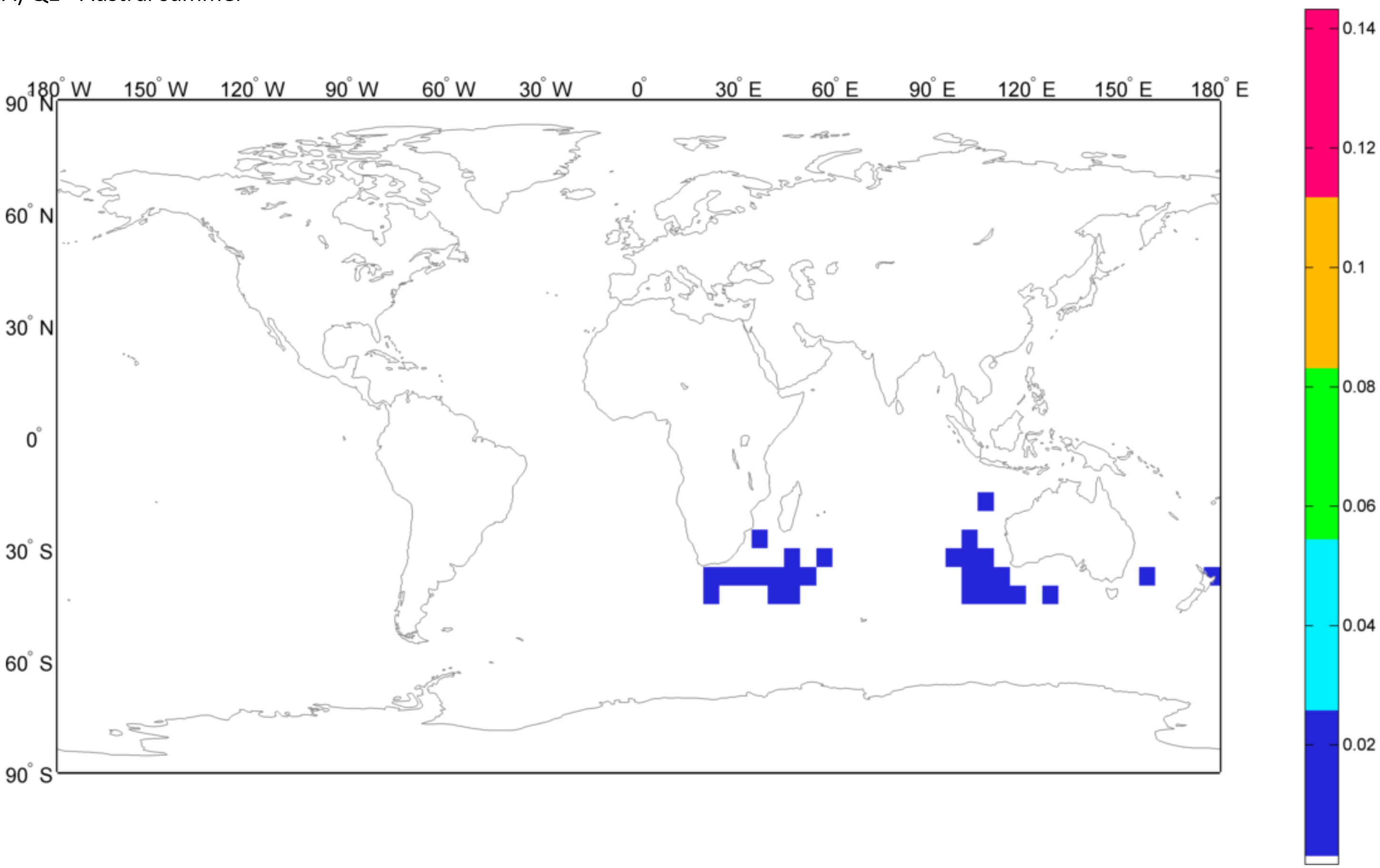
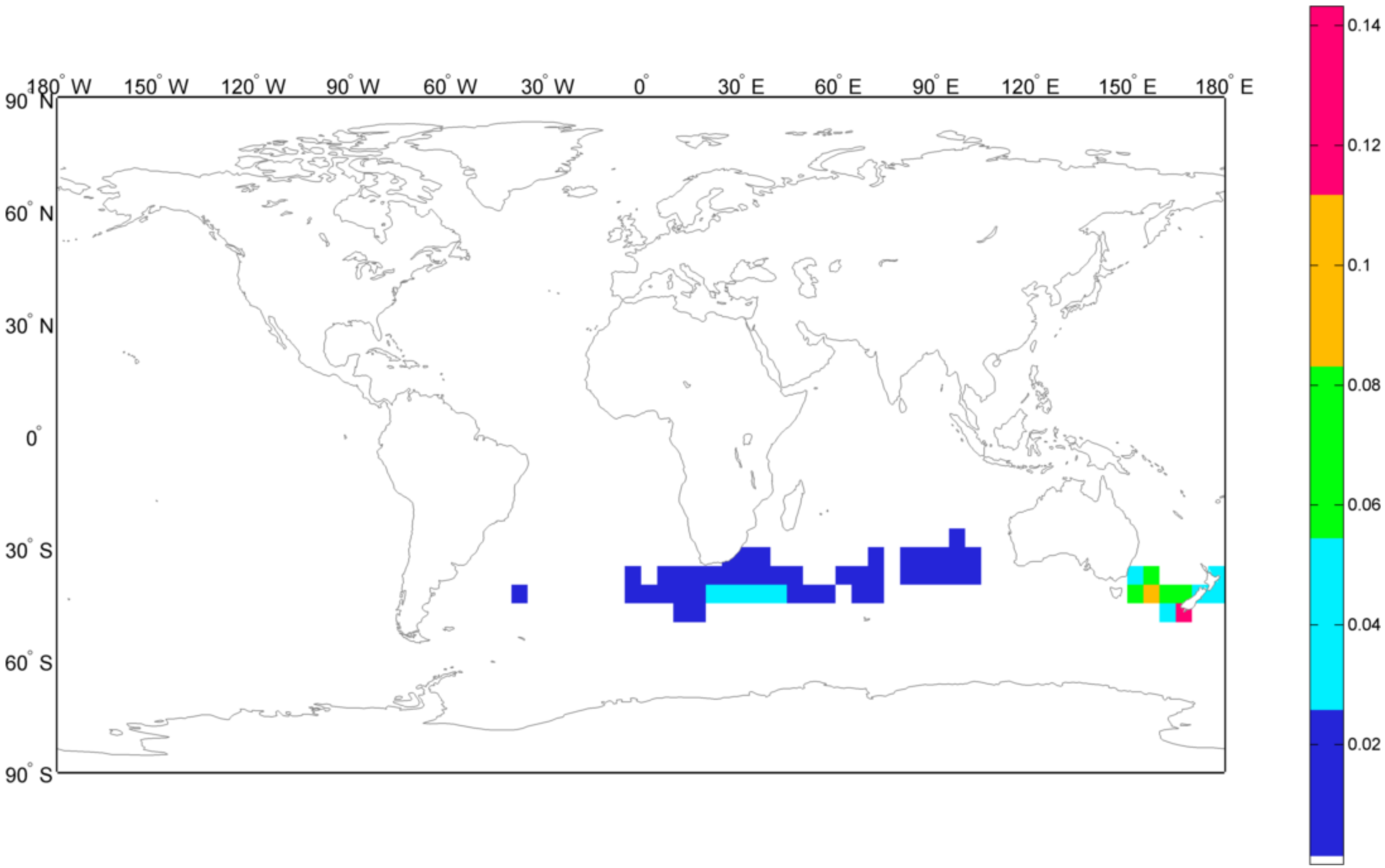


Figure 12. Seasonal risk in each 5 x 5 cell per seasonal quarter, combined for all 34 bird species included in this analysis: a) Q1 - Austral summer, b) Q2 – Austral autumn, c) Q3 - Austral winter, d) Q4 - Austral spring. By summing the untransformed cell values of the species-specific risk maps, the weighted contribution of each species to the combined map is in proportion to the species risk score in Table 3. Data visualisation here includes square-root transformed, giving a value on the scale of the potential catch of birds (~ birds^2/ km^2).

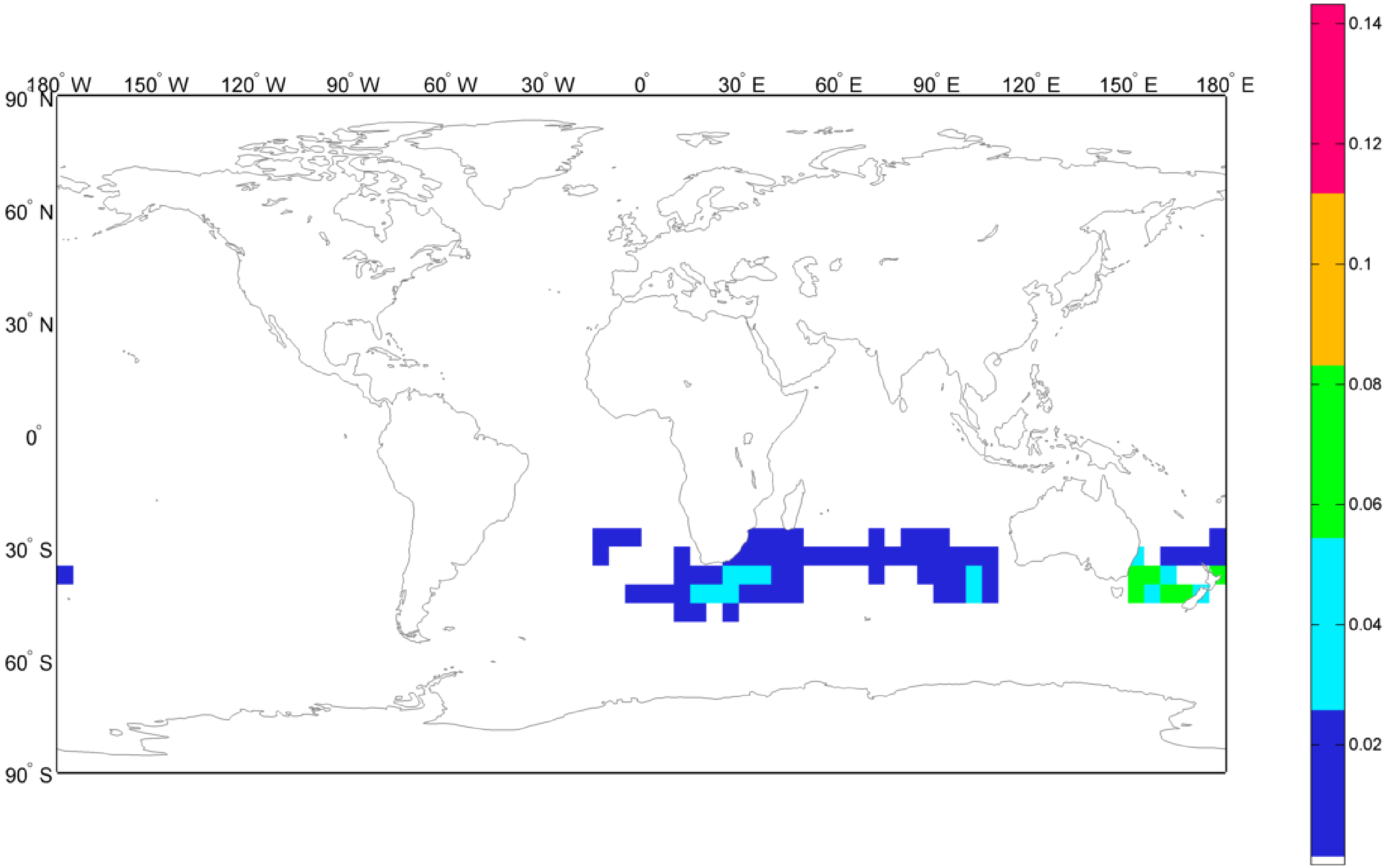
A) Q1 - Austral summer



b) Q2 - Austral autumn



c) Q3 - Austral Winter



d) Q4 - Austral Spring

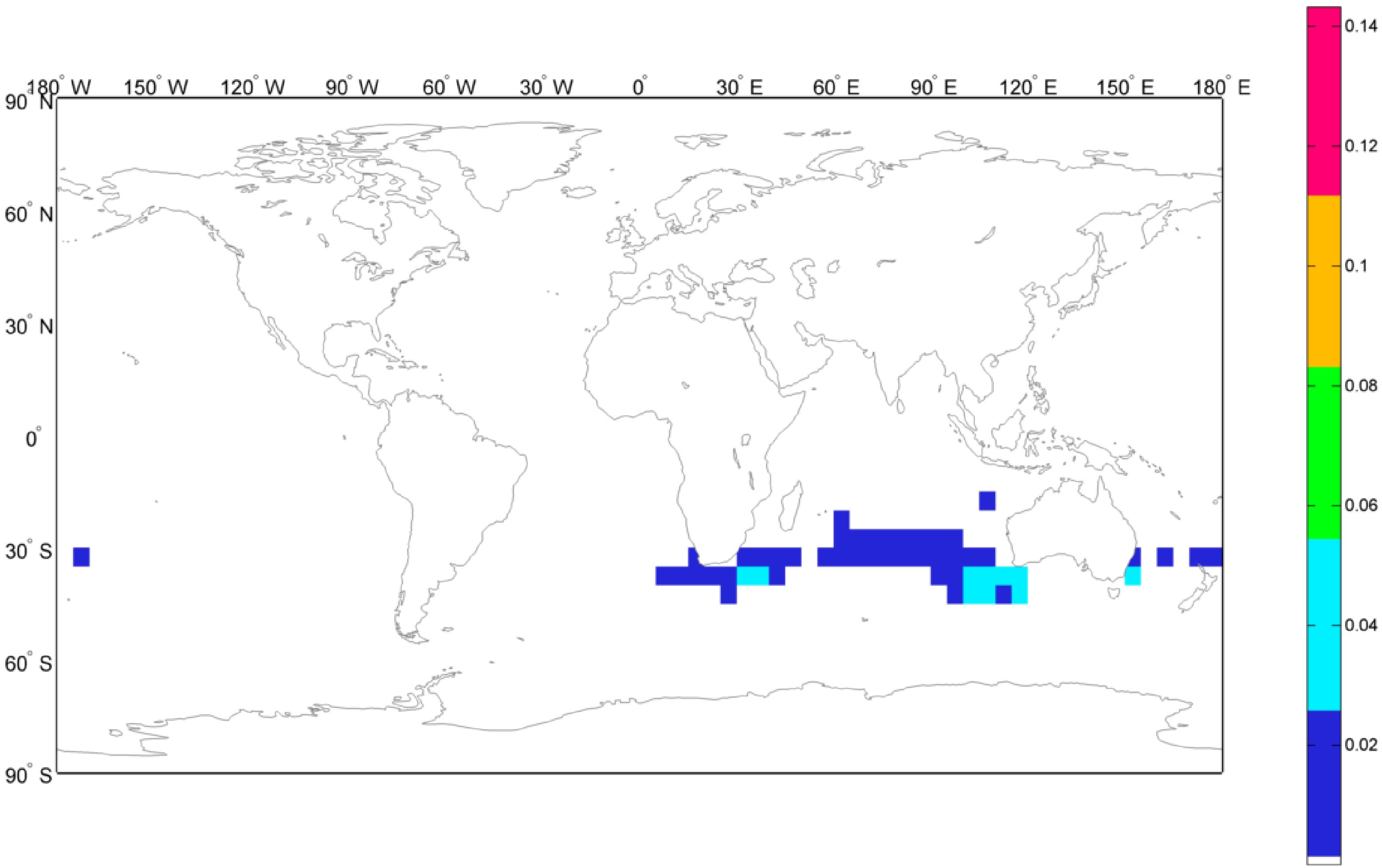


Figure 14. Total annual risk in each 5 x 5 square combined for all 34 species in this analysis, with the maximum value for any cell across all four seasonal quarters in Figure 13. By summing the untransformed cell values of the species-specific risk maps, the weighted contribution of each species to the combined map is in proportion to the species risk score in Table 3. Data visualisation here includes square-root transformed, giving a value on the scale of the potential catch of birds (~ birds^2/ km^2).

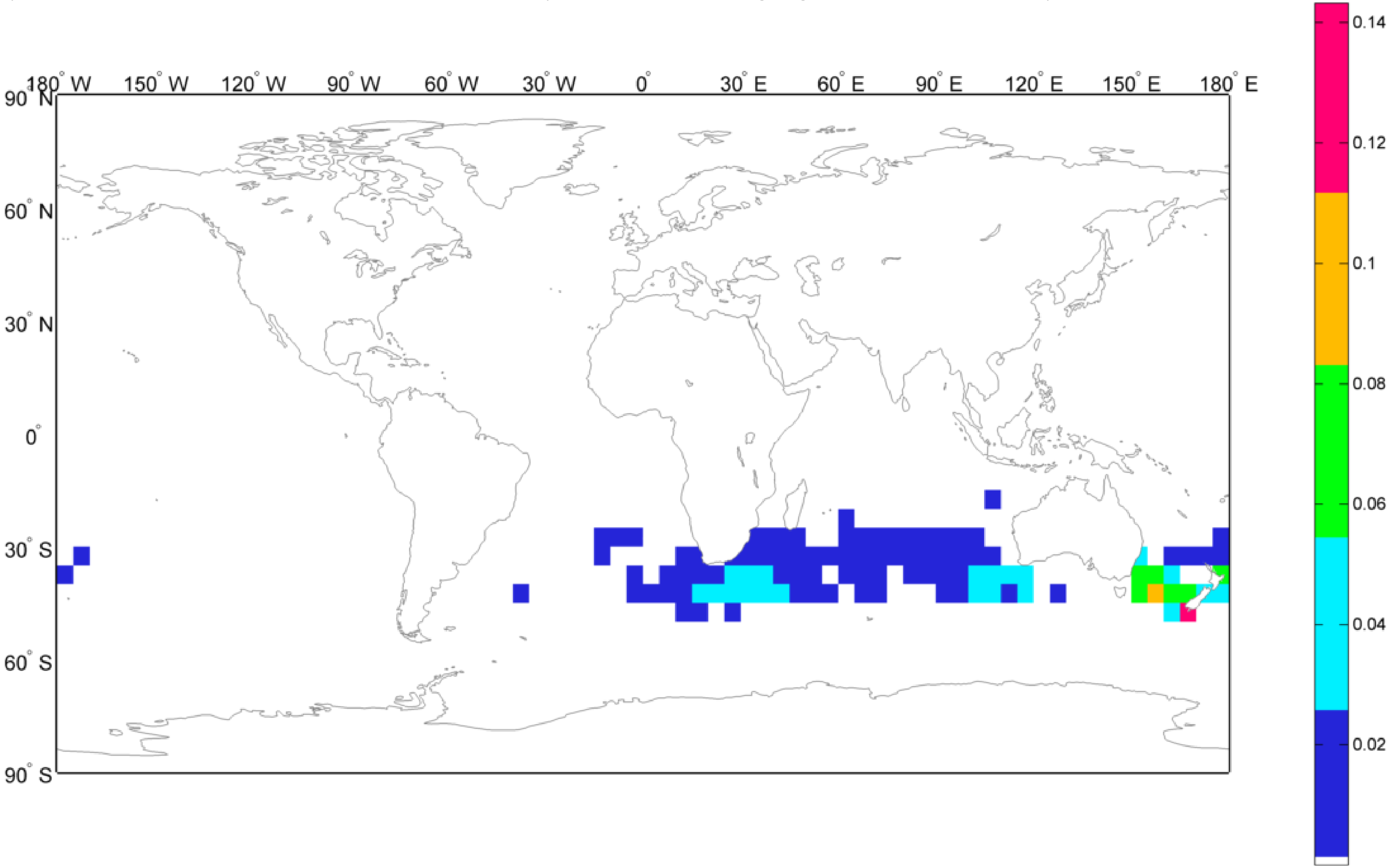


Figure 14. Percentage of the risk to seabird populations likely to be occurring associated with CCSBT longline fishing for tunas, by flag.

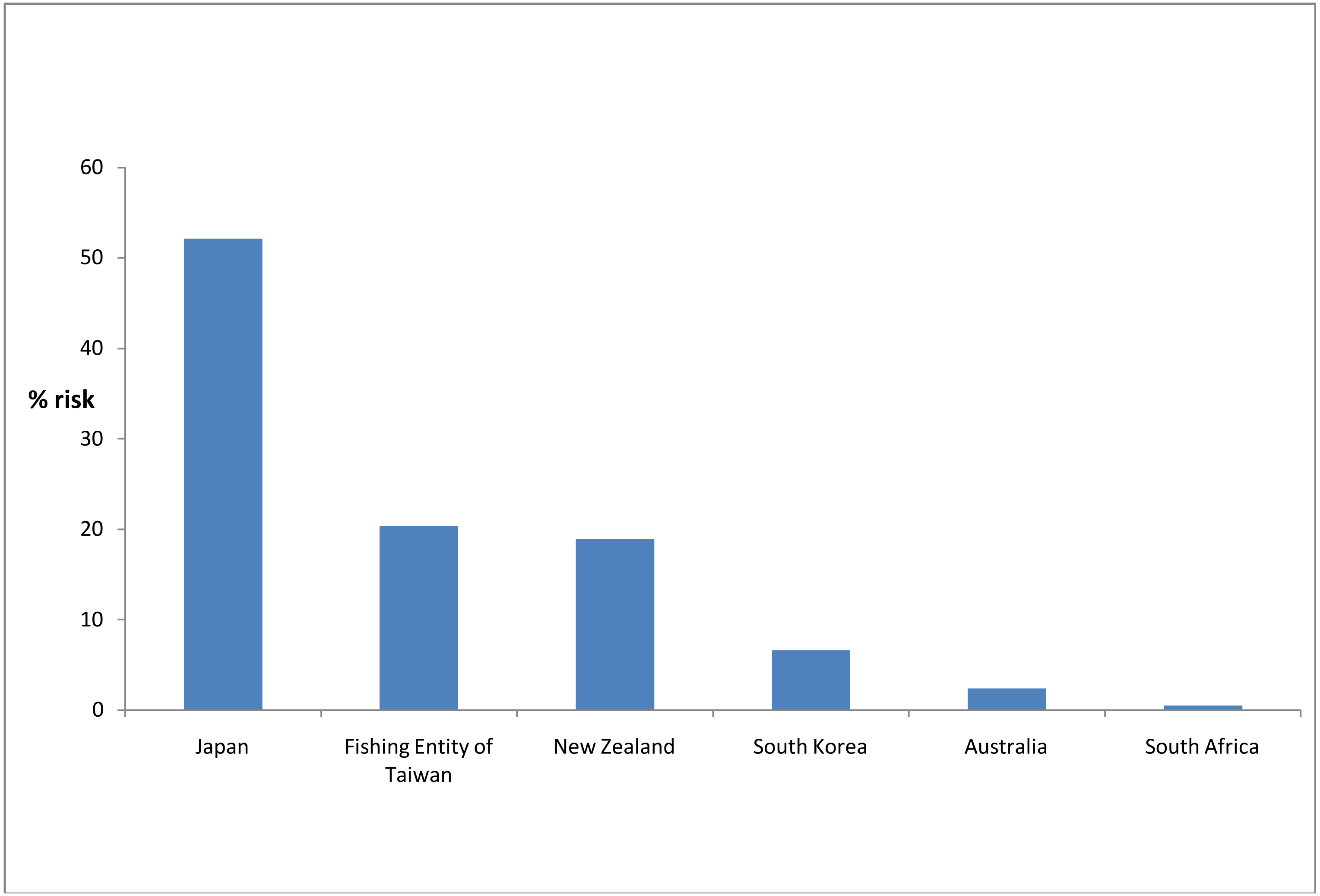


Table 1: Mitigation measures in force in selected fishery Commissions aimed at avoiding incidental capture of seabirds. Symbols ○ = voluntary deployment; ● = mandatory deployment. Note that these measures apply to restricted areas of each Convention area only, not detailed here. ( ) Measures indicated as required due to the mandatory nature of their use in other fisheries commissions. * At least two of the measures indicated with this symbol must be used in WCPFC, and either Streamer lines or a combination of line weighting and night setting in ICCAT for Swordfish fishing, and for IOTC at least one of the items marked * must be deployed

Fishery commission	Mitigation measures in force									
	Streamer lines / tori pole	Night setting	Weighted branch lines / minimum line sink rate	Side-setting and weighted lines	Offal management	Thawed baits	Blue-dyed bait	Underwater setting chute / line shooting device	Bird exclusion device for hauling	Conservation measure(s) referring to mitigation measures
CCSBT	●	( )*	( )*							CCSBT-ERS Recommendation 2011
IOTC	●*	●*	●*		○		○ (squid only)	○		IOTC Recommendation 10/06
WCPFC(South Pacific)	●*	●*	●*							WCPFC Conservation and Management Measure 2012-07
CCAMLR	●	●	●		●				●	CCAMLR Conservation Measure 24-02(2008) CCAMLR Conservation Measure 25-02 (2009)
ICCAT	●*	●*	●*							ICCAT Recommendation 2007-07, ICCAT Recommendation 2011-09

Table 2. Species characteristics for 34 seabird species included in the analysis, including species identifiers (common name, scientific name, species code in analysis); listed by species group by which species *Vulnerability* was assigned; Average age at maturity; Life-history strategy: 3 = biennial breeder, single-clutch egg; 2 = annual breeder, single-clutch egg; FFI – *Fecundity factors index*, calculated by multiplying a normalised LHS and normalised Average age at maturity, and taking the average value of the product. IUCN threat ranking: CR – Critical, EN – Endangered, VU – Vulnerable, NT – Near Threatened, LC – Least concern. World population size used in the analysis; Mean maximum foraging range from breeding colony; Breeding dates (start and end); Relative vulnerability to capture.

Common name	Scientific name	Species group	Code	Average age at maturity	LHS	IUCN 3.1	Spatial data type used	World Population (pairs)	Mean maximum foraging radius from the colony (km)	Breeding start (month)	Breeding end (month)	Relative vulnerability to capture in surface longline fisheries
Amsterdam Albatross	*Diomedea amsterdamensis*	Large albatrosses	DAM	9	3	CR	Remote tracking	26	1200	2	2	3.5539
Antipodean Albatross	*Diomedea antipodensis*	Large albatrosses	ANA	7	3	VU	Remote tracking	6286	2000	1	1	3.5539
Tristan Albatross	*Diomedea dabbenena*	Large albatrosses	DBB	10	3	CR		1700	2500	1	1	3.5539
Southern Royal Albatross	*Diomedea epomophora*	Large albatrosses	DIP	7	3	VU	Colony & Range	7900	1000	10	10	3.5539
Wandering Albatross	*Diomedea exulans*	Large albatrosses	DIX	9	3	VU	Remote tracking	8050	1800	1	1	3.5539
Gibson's Albatross	*Diomedea gibsoni*	Large albatrosses	GBA	8	3	VU	Colony & Range	5271	2000	12	12	3.5539
Northern Royal Albatross	*Diomedea sanfordi*	Large albatrosses	DIS	7	3	EN	Remote tracking	5832	1250	1	1	3.5539
Short-tailed Albatross	*Phoebastria albatrus*	Small albatrosses	PHA	6.77	2	VU	Colony & Range	470	1500	10	6	2.5709

Common name	Scientific name	Species group	Code	Average age at maturity	LHS	IUCN 3.1	Spatial data type used	World Population (pairs)	Mean maximum foraging radius from the colony (km)	Breeding start (month)	Breeding end (month)	Relative vulnerability to capture in surface longline fisheries
Laysan Albatross	*Phoebastria immutabilis*	Small albatrosses	PHI	8	2	NT	Colony & Range	591356	1000	9	7	2.5709
Waved Albatross	*Phoebastria irrorata*	Small albatrosses	PIR	8.3	2	CR	Remote tracking	9620	200	3	12	2.5709
Black-footed Albatross	*Phoebastria nigripes*	Small albatrosses	PHN	4	2	EN	Colony & Range	61307	250	10	6	2.5709
Sooty Albatross	*Phoebetria fusca*	Sooty albatrosses	PHF	11.8	3	EN	Colony & Range	13890	2000	7	5	0.0362
Light-mantled Sooty Albatross	*Phoebetria palpebrata*	Sooty albatrosses	PHE	9.5	3	NT	Remote tracking	22611	1550	9	5	0.0362
Buller's Albatross	*Thalassarche bulleri*	Small albatrosses	DNB	5	2	NT	Remote tracking	30460	450	12	9	6.9264
Indian Yellow-nosed Albatross	*Thalassarche carteri*	Small albatrosses	TQH	9	2	EN	Colony & Range	65000	2600	8	4	2.5709
Shy Albatross	*Thalassarche cauta*	Small albatrosses	THC	9	2	NT	Remote tracking	12585	250	7	7	2.5709
Atlantic Yellow-nosed Albatross	*Thalassarche chlororhynchos*	Small albatrosses	THH	9	2	EN	Remote tracking	69100	2600	8	4	2.5709
Grey-headed Albatross	*Thalassarche chrysostoma*	Small albatrosses	DIC	10	3	VU	Remote tracking	95748	1600	9	5	0.0589

Common name	**Scientific name**	**Species group**	**Code**	**Average age at maturity**	**LHS**	**IUCN 3.1**	**Spatial data type used**	**World Population (pairs)**	**Mean maximum foraging radius from the colony (km)**	**Breeding start (month)**	**Breeding end (month)**	**Relative vulnerability to capture in surface longline fisheries**
Chatham Albatross	*Thalassarche eremita*	Small albatrosses	DER	7	2	VU	Remote tracking	4575	600	7	4	0.0831
Campbell Albatross	*Thalassarche impavida*	Small albatrosses	TQW	10	2	VU	Colony & Range	21000	650	8	5	2.4650
Black-browed Albatross	*Thalassarche melanophrys*	Small albatrosses	DIM	9	2	EN	Remote tracking	601686	1100	9	5	2.4650
Salvin's Albatross	*Thalassarche salvini*	Small albatrosses	DLS	9	2	VU	Remote tracking	31947	1500	8	4	0.3644
White-capped Albatross	*Thalassarche steadi*	Small albatrosses	XWM	7	2	NT	Remote tracking	97111	450	11	11	2.5709
Southern Giant Petrel	*Macronectes giganteus*	Giant petrels	MAI	7	2	LC	Remote tracking	50170	250	6	6	0.1557
Northern Giant Petrel	*Macronectes halli*	Giant petrels	MAH	7.5	2	LC	Remote tracking	11800	550	8	5	0.1557
White-chinned Petrel	*Procellaria aequinoctialis*	Procellaria petrels	PRO	6.5	2	VU	Remote tracking	1241000	1900	10	5	0.1512
Grey Petrel	*Procellaria cinerea*	Procellaria petrels	PCI	7	2	NT	Remote tracking	111684	600	2	12	2.1323
Spectacled Petrel	*Procellaria conspicillata*	Procellaria petrels	PCO	7	2	VU	Colony & Range	10000	1900	9	3	0.0015

Common name	Scientific name	Species group	Code	Average age at maturity	LHS	IUCN 3.1	Spatial data type used	World Population (pairs)	Mean maximum foraging radius from the colony (km)	Breeding start (month)	Breeding end (month)	Relative vulnerability to capture in surface longline fisheries
Parkinson's Petrel	*Procellaria parkinsoni*	Procellaria petrels	PRK	7	2	VU	Remote tracking	3333	550	10	6	0.0015
Westland Petrel	*Procellaria westlandica*	Procellaria petrels	PCW	6	2	VU	Remote tracking	4000	500	2	12	1.0752
Great-winged Petrel	*Pterodroma macroptera*	Large Pterodromas	PDM	6.5	2	LC	Colony & Range	500000	600	6	1	0.0075
Flesh-footed Shearwater	*Puffinus carneipes*	Dark shearwaters	PFC	5.5	2	LC	Remote tracking	216000	250	9	5	1.1660
Sooty Shearwater	*Puffinus griseus*	Dark shearwaters	PFG	6	2	NT	Remote tracking	6000000	100	9	5	0.0030
Cape Pigeon	*Daption capense*	Southern petrels	DAC	6	2	LC	Colony & Range	666000	360	10	1	0.6926

Table 3: Species parameters and indices contributing to the estimate of total species risk (final column) for each of 34 seabird species included in this analysis. Species risk is the product of the normalized *Susceptibility* and inverse *Productivity* indices as illustrated in Figure 6; mathematical derivation of the Susceptibility and Productivity indices as a function of the other parameters is as described in the text.

Species common name	Code	IUCN threat ranking	species spatial overlap	Rank in analysis	species spatial overlap*V/P	Susceptibility S	Productivity P	Score (SxP)
Antipodean Albatross	ANA	VU	1.608747e-006	1	9.281957e-007	0.617754	0.576968	0.356425
Wandering Albatross	DIX	VU	1.091644e-006	2	8.222609e-007	0.419188	0.753231	0.315746
White-capped Albatross	XWM	NT	3.545315e-006	3	1.004020e-006	0.984836	0.283196	0.278902
Gibson's Albatross	GBA	VU	7.849892e-007	4	5.220962e-007	0.301434	0.6651	0.200484
Indian Yellow-nosed Albatross	TQH	EN	1.227256e-006	5	5.638744e-007	0.340914	0.459459	0.156636
Southern Royal Albatross	DIP	VU	4.869468e-007	6	2.809529e-007	0.186986	0.576968	0.107885
Buller's Albatross	DNB	NT	1.336208e-006	7	1.428848e-007	1	0.106933	0.106933
Campbell Albatross	TQW	VU	4.695927e-007	8	2.571448e-007	0.125075	0.547591	0.06849
Amsterdam Albatross	DAM	CR	2.299510e-007	9	1.732063e-007	0.088301	0.753231	0.066511
Atlantic Yellow-nosed Albatross	THH	EN	2.945678e-007	10	1.353420e-007	0.081827	0.459459	0.037596
Black-browed Albatross	DIM	EN	1.981171e-007	11	9.102679e-008	0.052768	0.459459	0.024245
Flesh-footed Shearwater	PFC	LC	5.957785e-007	12	8.996185e-008	0.075058	0.150999	0.011334
Shy Albatross	THC	NT	7.818120e-008	13	3.592109e-008	0.021718	0.459459	0.009978
Cape Pigeon	DAC	LC	3.981066e-007	14	7.765652e-008	0.029794	0.195065	0.005812
Sooty Albatross	PHF	EN	6.743614e-007	15	6.743614e-007	0.002645	1	0.002645
Tristan Albatross	DBB	CR	6.292093e-009	16	5.293935e-009	0.002416	0.841363	0.002033
Salvin's Albatross	DLS	VU	1.041420e-007	17	4.784904e-008	0.004101	0.459459	0.001884
Grey-Headed Albatross	DIC	VU	2.654878e-007	18	2.233716e-007	0.001692	0.841363	0.001423
Light-mantled Sooty Albatross	PHE	NT	3.669920e-007	19	2.926017e-007	0.001439	0.797297	0.001147
Grey Petrel	PCI	NT	1.459872e-008	20	4.134303e-009	0.003364	0.283196	0.000953
Westland Petrel	PCW	VU	3.669321e-008	21	7.157547e-009	0.004263	0.195065	0.000832
Northern Royal Albatross	DIS	EN	3.546213e-009	22	2.046052e-009	0.001362	0.576968	0.000786
Great-winged Petrel	PDM	LC	1.114508e-006	23	2.665128e-007	0.000911	0.23913	0.000218
Northern Giant Petrel	MAH	LC	1.977374e-008	24	6.471195e-009	0.000333	0.327262	0.000109
Chatham Albatross	DER	VU	3.863037e-008	25	1.093998e-008	0.000347	0.283196	0.000098
White-chinned Petrel	PRO	VU	3.772521e-009	26	9.021245e-010	0.000062	0.23913	0.000015

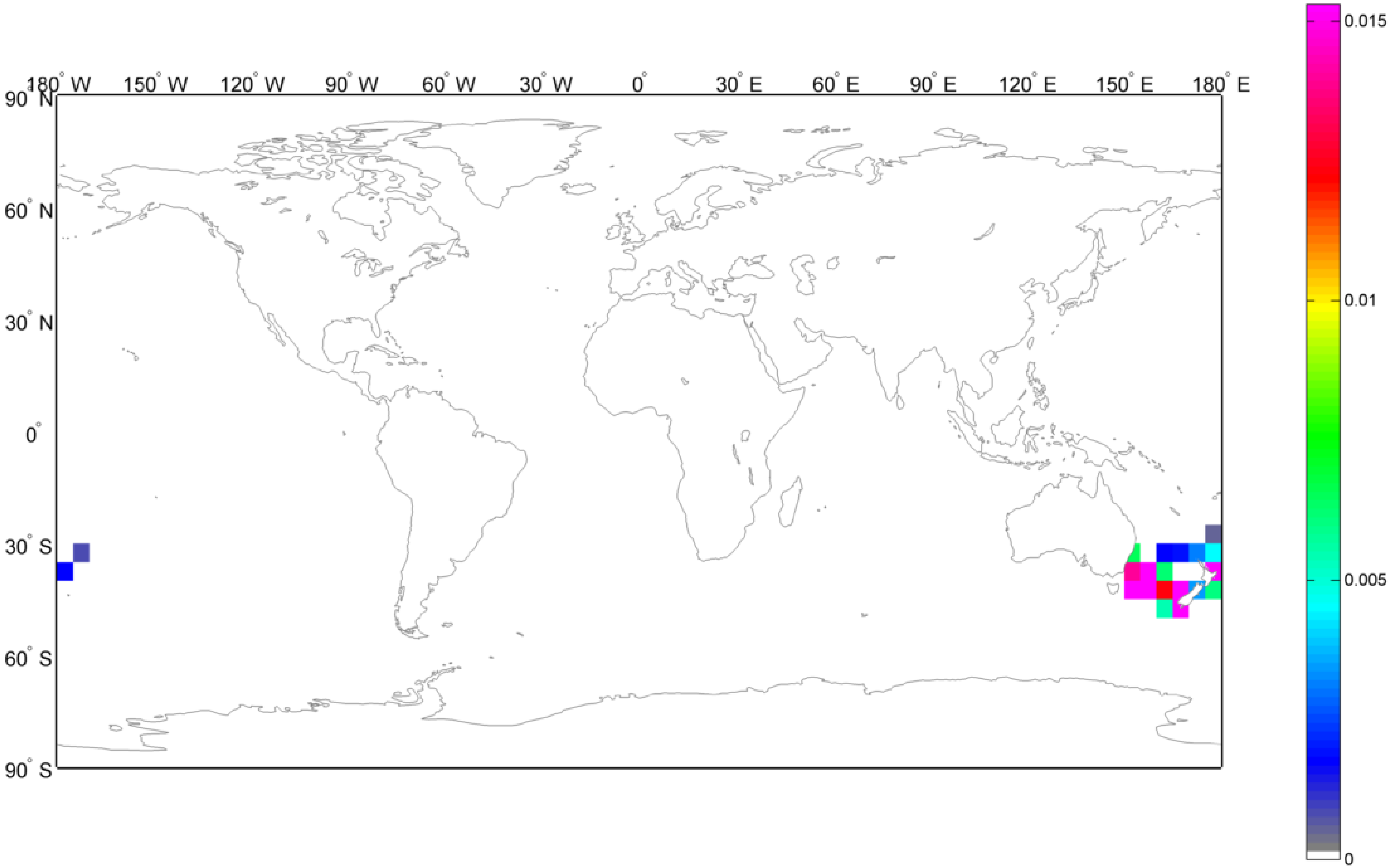
Species common name	Code	IUCN threat ranking	species spatial overlap	Rank in analysis	species spatial overlap*V/P	Susceptibility S	Productivity P	Score (SxP)
Spectacled Petrel	PCO	VU	2.275896e-008	27	6.445253e-009	0.000004	0.283196	0.000001
Parkinson's Petrel	PRK	VU	8.725601e-010	28	2.471058e-010	0	0.283196	0
Southern Giant Petrel	MAI	LC	5.675352e-012	29	1.607238e-012	0	0.283196	0
Sooty Shearwater	PFG	NT	9.471242e-013	30	1.847504e-013	0	0.195065	0
Short-tailed Albatross	PHA	VU	0.000000e+000	31	0.000000e+000	0	0.262926	0
Laysan Albatross	PHI	NT	0.000000e+000	32	0.000000e+000	0	0.371328	0
Black-footed Albatross	PHN	EN	0.000000e+000	33	0.000000e+000	0	0.018801	0
Waved albatross	PIR	CR	28860	34	0.000000e+000	0	0.397767	0

APPENDIX 1

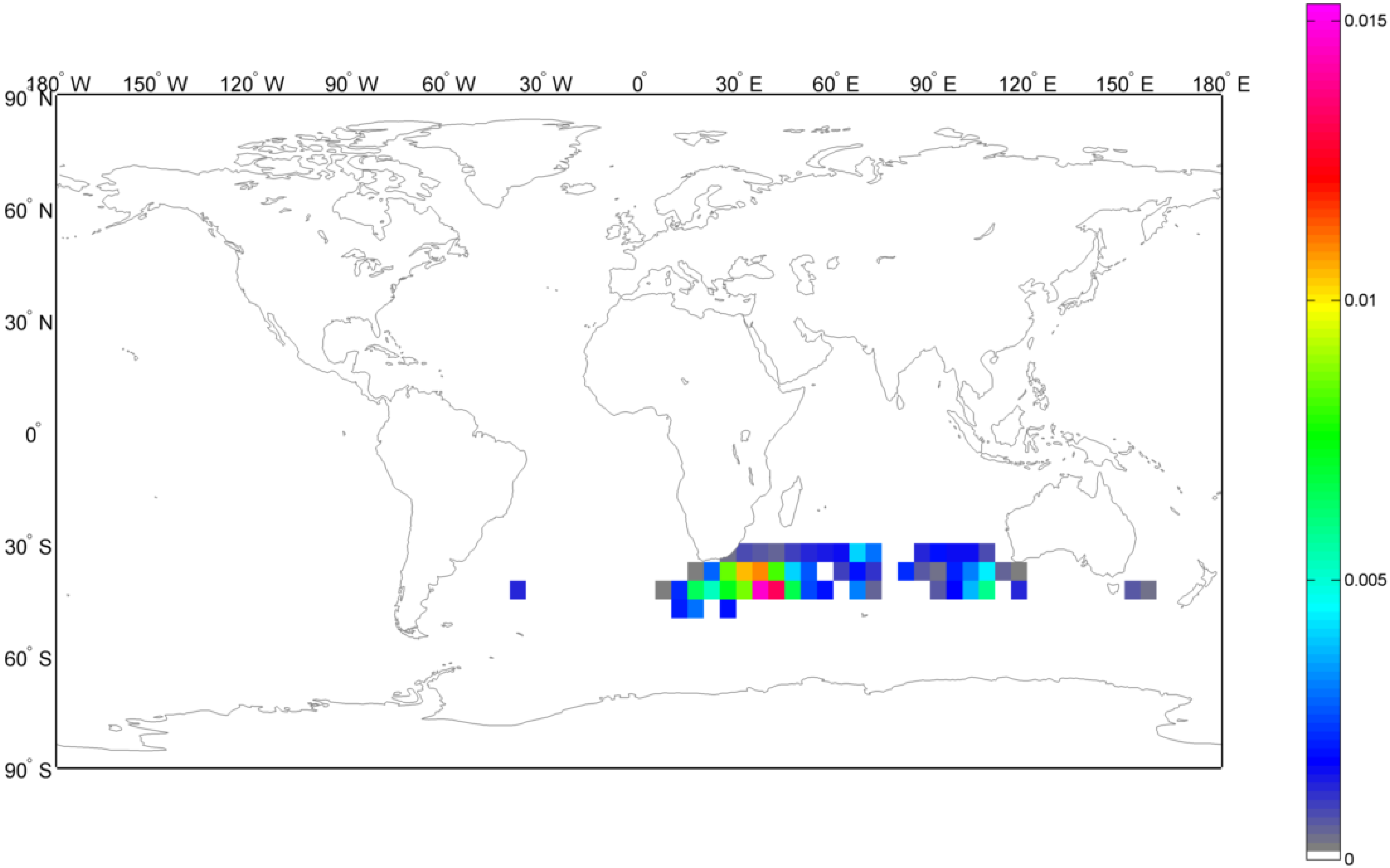
SINGLE SPECIES ANNUAL RISK MAPS (High to Moderate Risk species)

Maps follow in order of risk from highest to lowest (A1 through A11). The risk score for each cell is where fishing effort overlaps with species distribution and the maximum risk score for each cell during the year is shown.

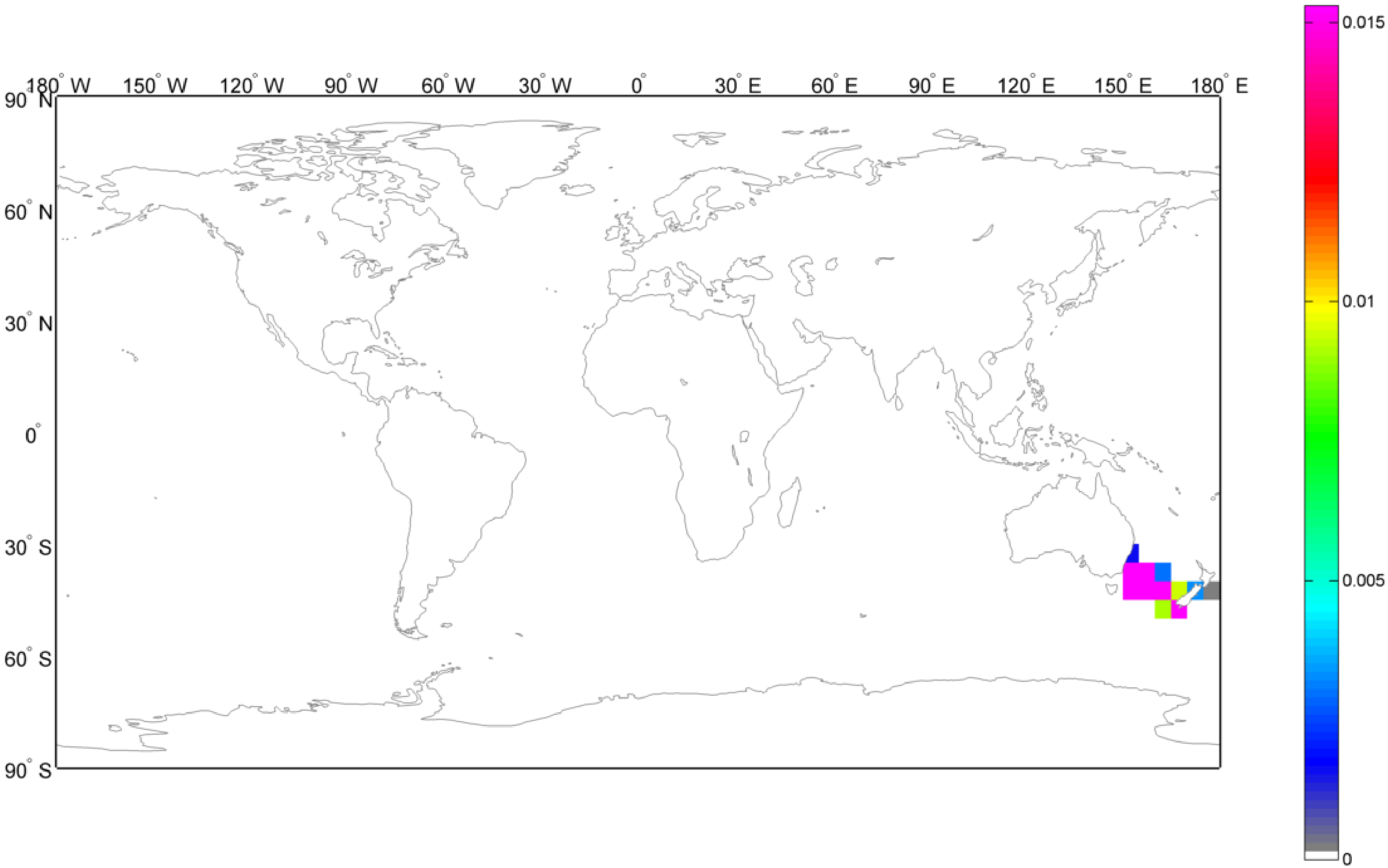
A1.1 Antipodean albatross, Risk rank: 1, score 0.356



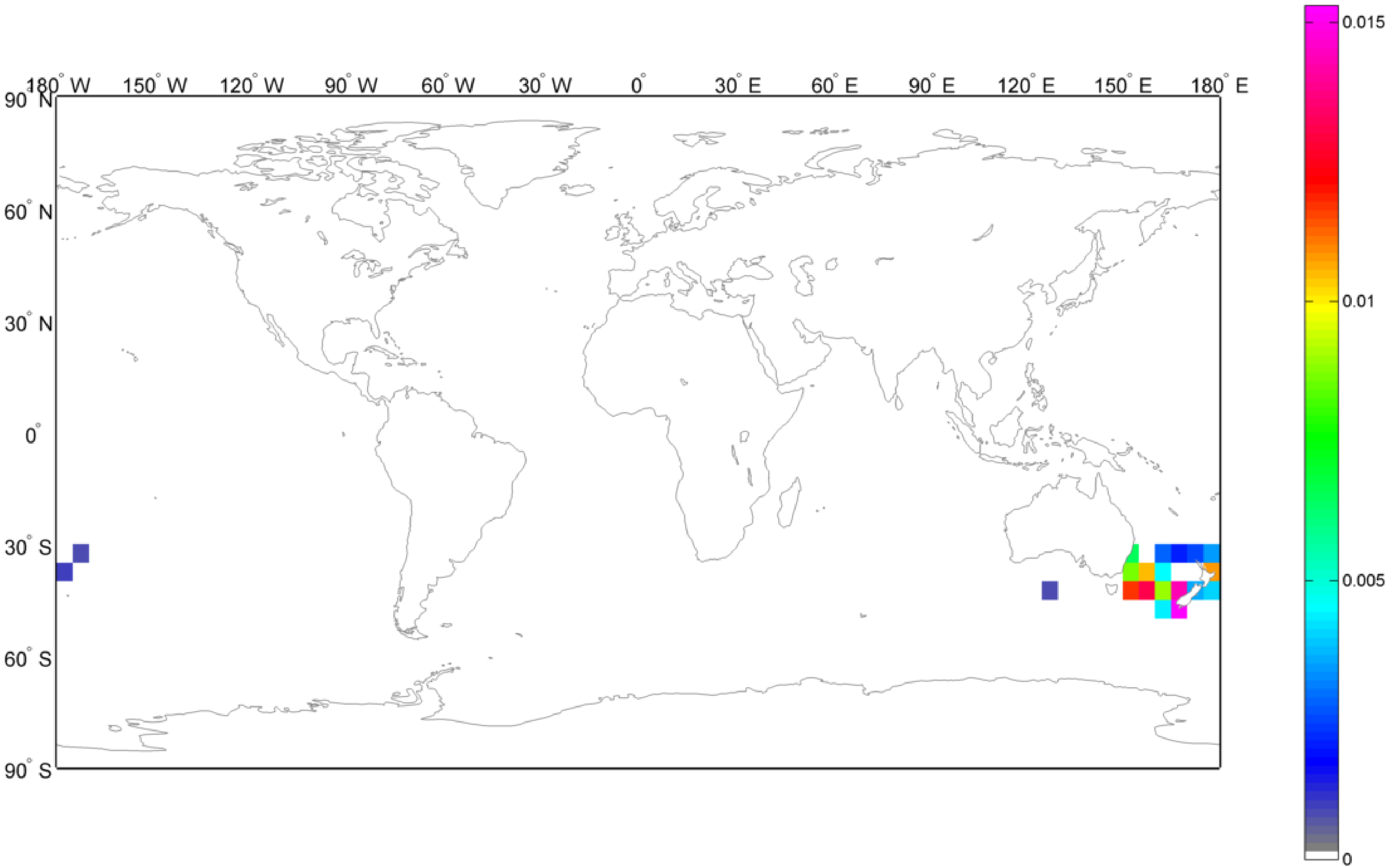
A1.2 Wandering albatross. Risk rank 2. Risk score 0.315.



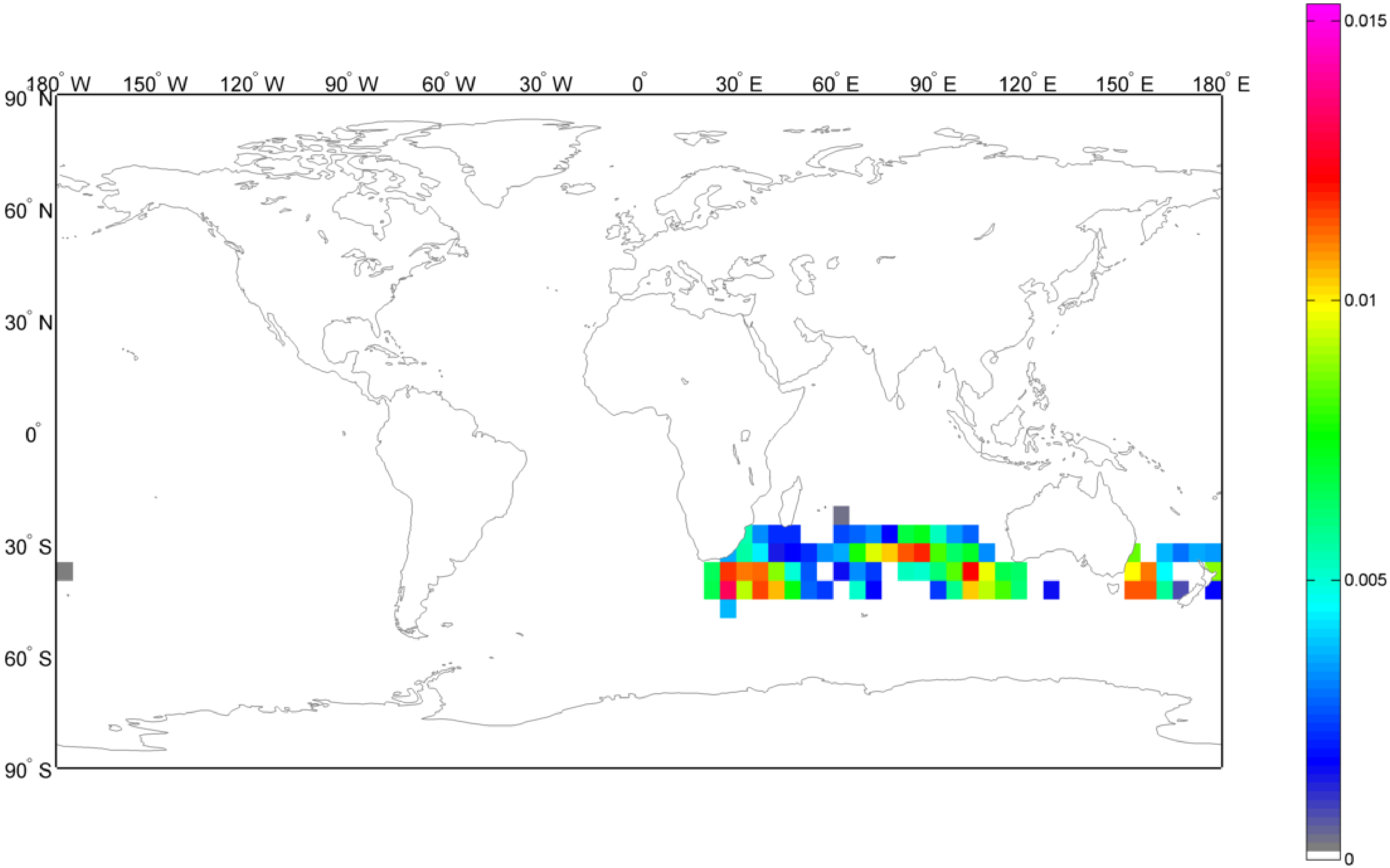
A1.3 White-capped albatross. Risk rank 3. Risk score 0.279



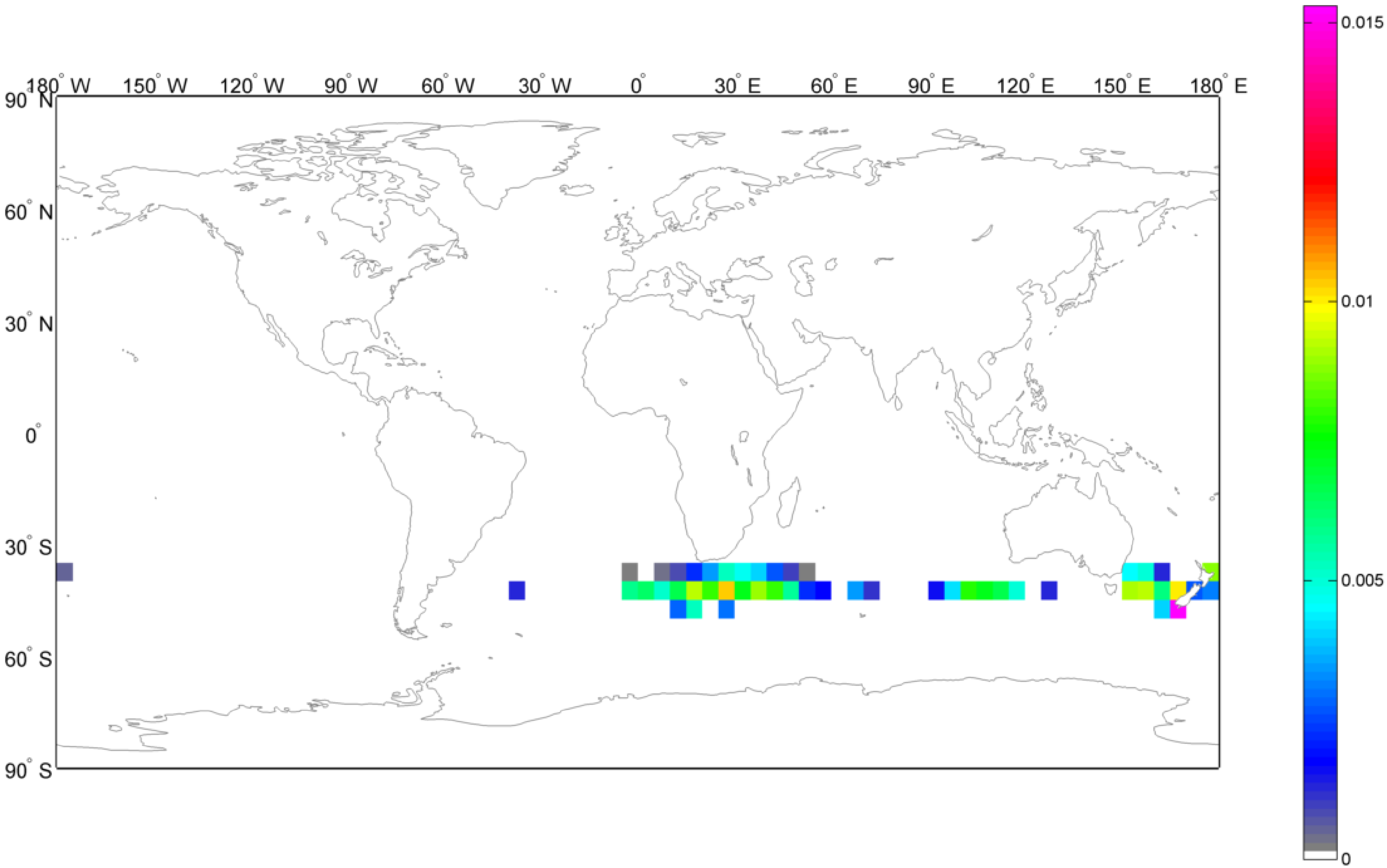
A1.4 Gibson’s albatross. Risk rank 4. Risk score 0.2.



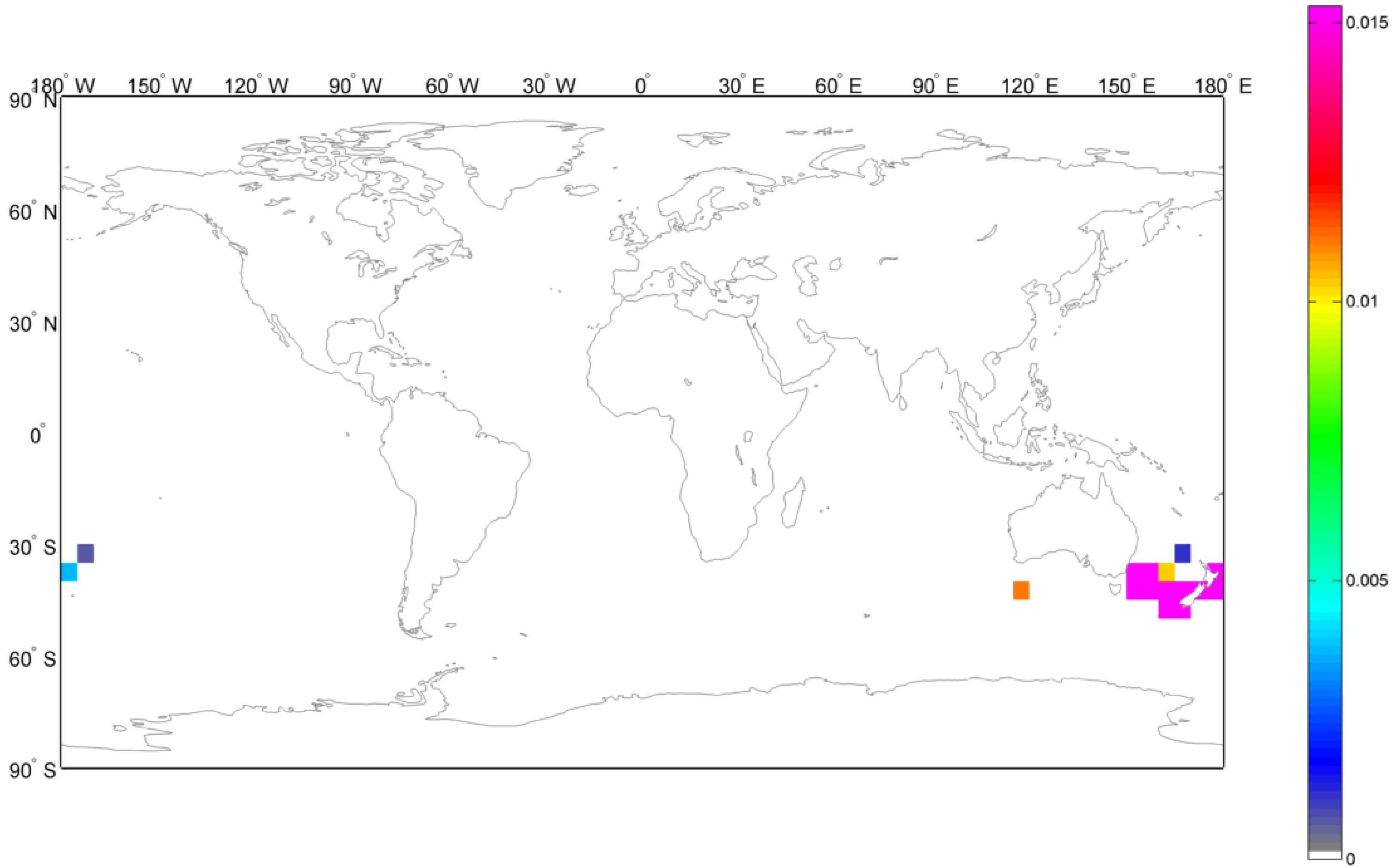
A1.5 Indian yellow-nosed albatross. Risk rank 5. Risk score 0.156.



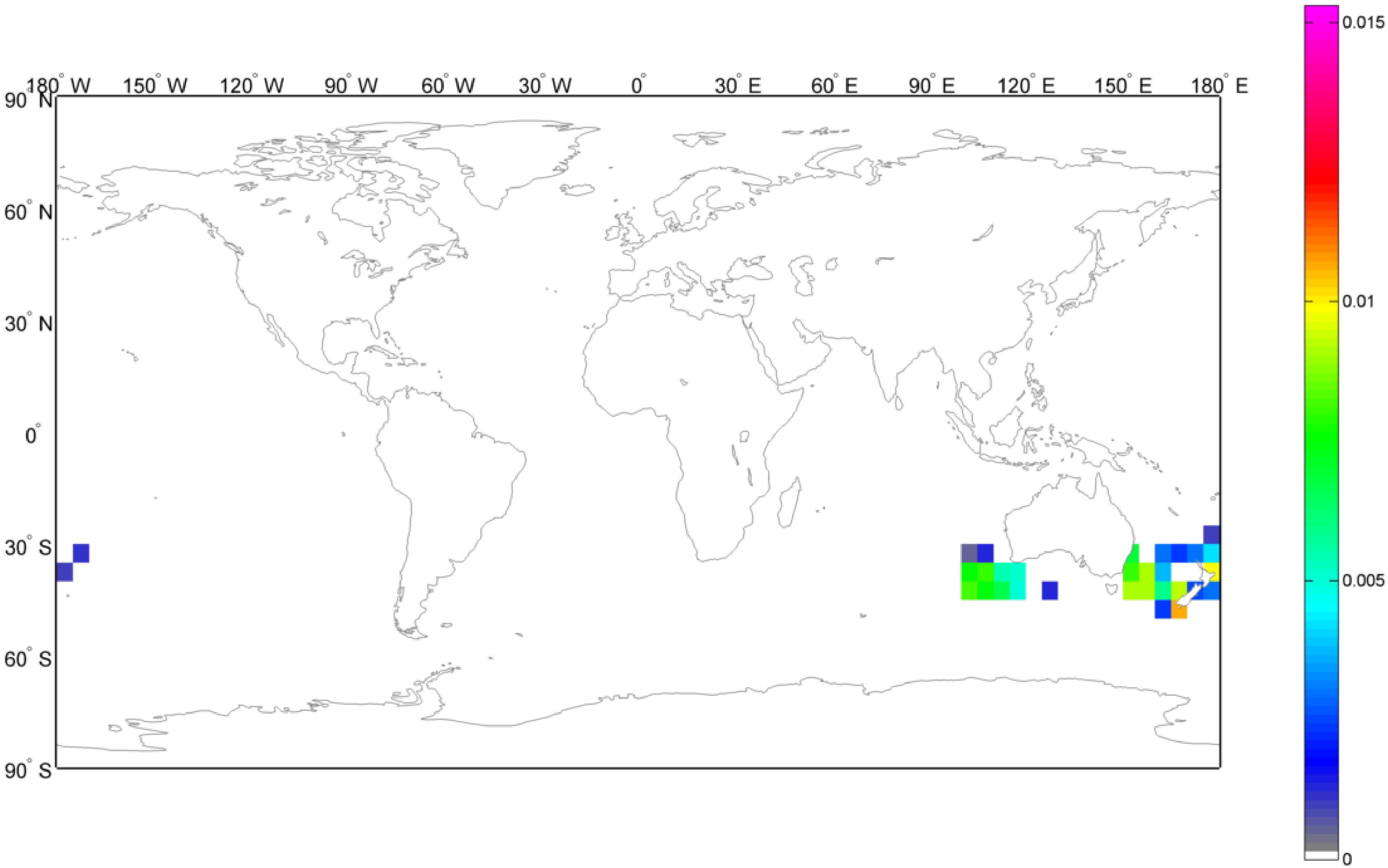
A1.6 Southern royal albatross. Risk rank 6. Risk score 0.108.



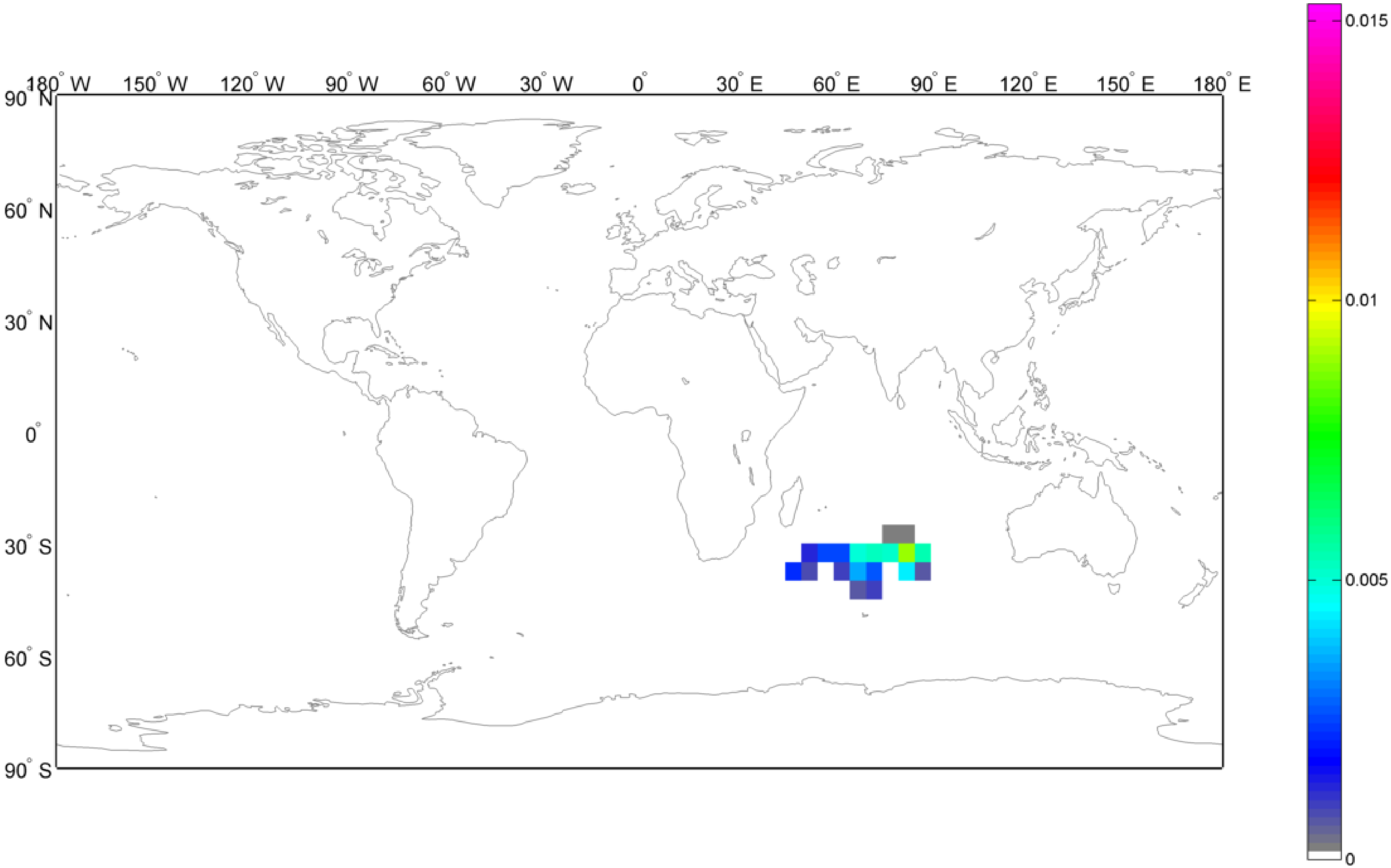
A1.7. Buller's albatross. Risk rank 7. Risk score 0.107.



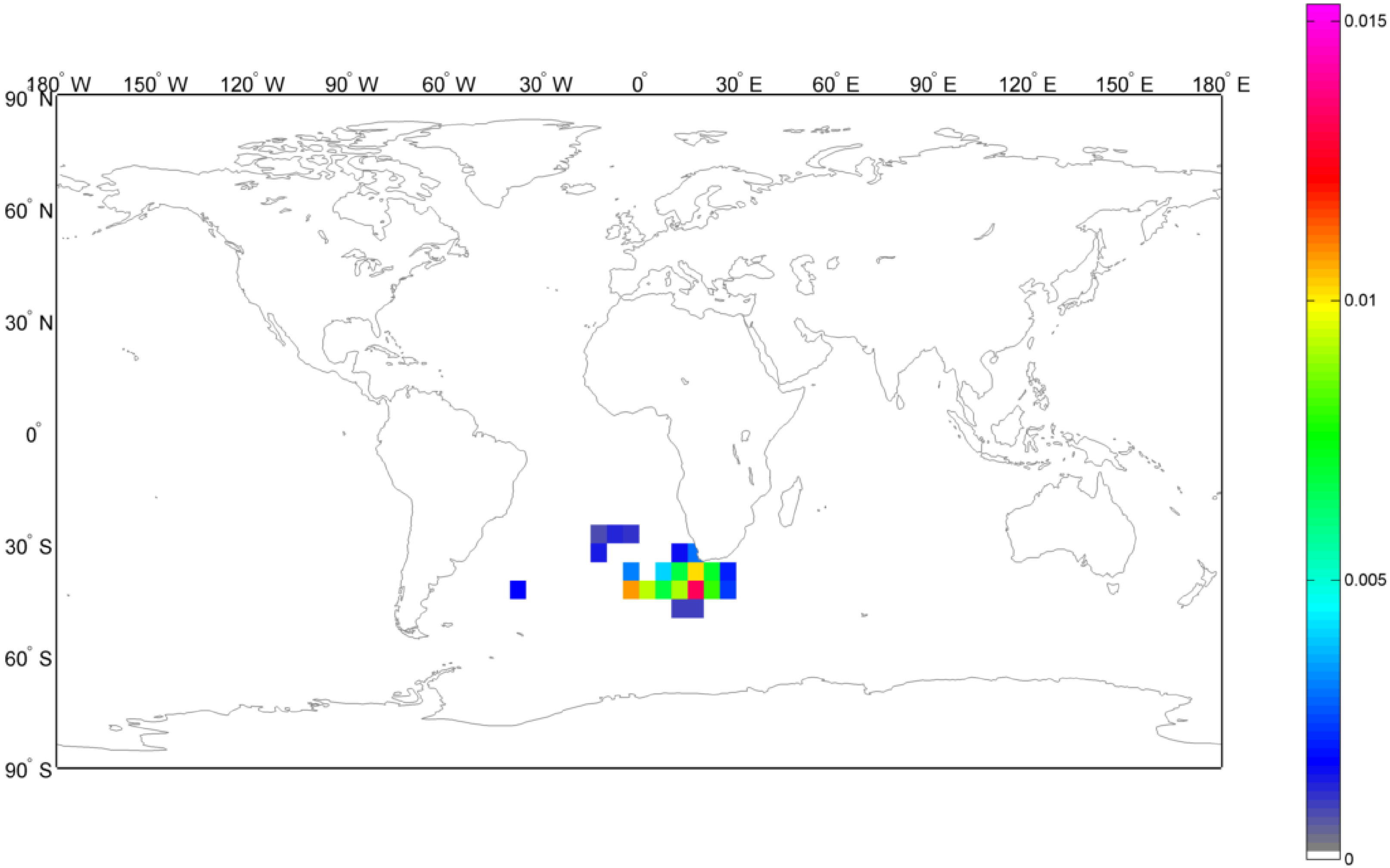
A1.8 Campbell albatross. Risk rank 8. Risk score 0.068.



A1.9 Amsterdam albatross. Risk rank 9. Risk score 0.067.



A1.10 Atlantic yellow-nosed albatross. Risk rank 10. Risk score 0.038.



A1.11 Black-browed albatross. Risk rank 11. Risk score 0.024.